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List of abbreviations

Abbreviation	**Description**
TSS	Total suspended solids
DP	Dissolved pesticides
PP	Particulate pesticides
OM	Organic matter
OC	Organic carbon
HRT	Hydraulic residential time
DOC	Dissolved organic carbon
POC	Particulate organic carbon
Re	Reynolds number
vp	Vapor pressure
FWS	Free water surface
OECD	Organization for Economic Co-operation and Development
PPDB	Pesticide Properties
NER	Non-extractable residue
SET	Settling
RES	Resuspension
VOL	Volatilization
SOR	Sorption
BIO	Biotransformation
PHO	Photolysis
HYD	Hydrolysis

Process formulations and controlling factors of pesticide dissipation in artificial ponds: a critical review

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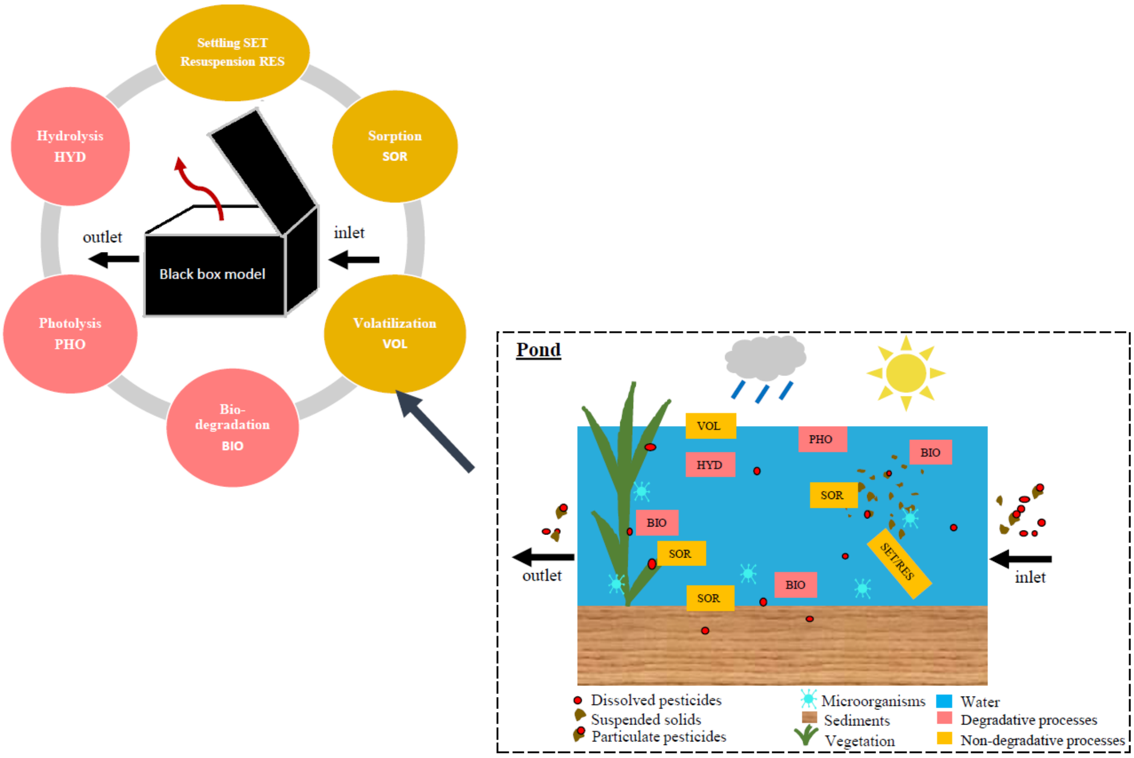
Graphical abstract:



Abstract

Pesticides are a non-point source of pollution affecting the quality of drinking water supplies and aquatic environments. Many facilities are set up to reduce the transfer of pesticides from agricultural parcels into the environment, preventing their input into water bodies. Artificial ponds collecting runoff and erosion fluxes offer a complementary tool to mitigate pesticide transfer. Ponds are free-water surface-constructed wetlands, rarely dried out and not exceeding two meters in depth. Their main compartments include a water column, a sediment layer, living organisms, and suspended particles. The dissipation potential of ponds is due to a combination of physical, chemical, and biological processes that pesticides undergo in an interplay between different pond compartments. Although many studies on the dissipation potential of ponds focus on nitrates and suspended sediments, very little is known about the behavior of pesticides. Also, most studies evaluate a process individually without emphasizing its eventual interaction with other processes. Few papers address and compare mathematical formulations of pesticide transfer and transformation processes for artificial pond modeling. In this review, we investigate the interactions and contribution of multiple processes to artificial pond dissipation function by comparing their rates evaluated at different scales. We also discuss the mathematical formulations of processes and their relationship with pesticide properties. Our evaluation suggests that quantifying the contribution of each process to pesticide dissipation in artificial ponds is a major tool for hierarchizing dissipation processes. We believe that quantification and hierarchization may improve the overall artificial pond efficiency estimation. We anticipate this work to be the basis for developing a process-based model of pesticide fate in artificial ponds. The model will be particularly helpful in setting up the dimensioning criteria to design performant and efficient ponds to mitigate pesticide transfer into the environment.

Keywords:

Artificial pond; pesticide; dissipation; process; model, review.

Highlights:

- Identification and formalisms of dissipation processes of pesticides in artificial ponds
- Sorption as a key process for lowly soluble and hydrophobic pesticide dissipation
- Biotransformation enhanced in water-substrate interfaces favoring microbial activity
- Volatilization and hydrolysis can be mostly neglected
- Processes interplay contributes more significantly to the dissipation of pesticides
- Added value of descriptive process-based model of pesticide behavior in artificial ponds

1. Introduction and definitions

1.1 Wetlands and artificial ponds

In agriculture, various pests (e.g., weeds, insects, rodents, fungi, molds.) can significantly deteriorate the quantity and quality of agricultural products. In order to protect agricultural production, chemicals called "pesticides" are commonly used. Pesticides cover many compounds, including insecticides, fungicides, herbicides, rodenticides, molluscicides, nematicides, plant growth regulators, and others (Aktar et al., 2009). Due to their toxicity, pesticides represent a non-point source of pollution for the ecosystem and affect the quality of drinking water supplies and aquatic environments. Material solutions can be implemented as a complementary tool to safeguard water resources quality and mitigate pesticide input into water bodies, such as edge-of-field and riparian buffer strips, vegetated ditches, wetlands, and artificial ponds (Vymazal et al., 2015). In this paper, we will focus on constructed wetlands, especially artificial ponds, since they have the advantages of needing minimal operations, i.e., being inexpensive to operate and construct and providing wildlife habitat (Sudarsan et al., 2021). A neat definition of wetlands and ponds and a description of their different types can be found in (Kadlec et al., 2008).

Artificial ponds are free water surface constructed wetlands, often kept wet and not exceeding 2m depth (Z. Zhang et al., 2021), and they serve different purposes. Artificial ponds can be used as detention ponds to store water for irrigation purposes (Chaumet et al., 2022; Tournebize et al., 2015) or stormwater ponds to prevent flooding (Maillard et al., 2012). Artificial ponds are also used as complementary tools to treat contaminated agricultural water (Tournebize et al., 2017). These artificial ponds intercept agricultural water through runoff or subsurface drainage from cultivated agricultural plots. Artificial ponds are constructed upon an impermeable layer of sediments. The main artificial pond compartments are a water column, a sediment layer, living organisms (i.e., vegetation and microorganisms), and suspended particles. Since the late '80s, artificial ponds have been generally applied to treat various wastewaters, including agricultural water (Chouinard et al., 2014; Grinham et al., 2018; Wang et al., 2018; Zamora et al., 2019; Zhuang et al., 2019). It was widely reported that artificial ponds are able to reduce the transfer of chemicals (pesticides, organics, and inorganics pollutants), physical components (solid suspended particulates), and biological elements (bacteria) to natural water resources and preserve their quality (Kasak et al., 2018; Zheng et al., 2018). For example, artificial ponds had an important removal efficiency for organic matter, solids, and nitrogen up to 90% (de Oliveira et al., 2020; Tournebize et al., 2015). For pesticides, artificial pond efficiency can reach 97% for the organochlorine chemical group (Matamoros et al., 2006; Rose et al., 2006), 96% for the strobilurin/strobin group, and 84% for pyrethroids (Vymazal et al., 2015).

In that matter, significant research effort has been dedicated to understanding the fate of pesticides in artificial ponds and their link to the pond and pesticide properties (Bloomfield et al., 2006). In artificial ponds, pesticides are dissolved in water, attached to suspended particles, and sorbed on sediments and vegetation. The dissipation potential of artificial ponds results from a combination of transport, transfer, and transformation processes of pesticides. Transport processes carry pesticides following a water flow. Transfer processes allow the pesticide molecule to move from one phase to another (i.e., dissolved phase to particulate phase or dissolved phase to gas phase), and transformation degrades and modifies the original chemical structure of pesticides.

Most studies could not distinguish the extent of removal processes because they depend on each other (Gregoire et al., 2009; Liu et al., 2019; Sahin et al., 2020) and thus could not model their impact on pesticide behavior. Additionally, previous reviews and studies (Stehle et al., 2011; Vymazal et al., 2015) focused on the efficiency of artificial ponds to remove pesticides and the ecological risk assessment of pesticides (Haith, 2010). However, they did not detail the processes behind pesticide dissipation, and they could not identify a direct link between ponds' efficiency and the pesticides' properties. Therefore, this paper investigates the relationship between dissipation processes and the main characteristics of pesticides in artificial ponds. Contrarily to the studies reviewed in this paper, this survey will evaluate plentiful pesticides with contrasting physicochemical properties and present the mathematical formulations of multiple processes evaluated from lab-scale to field-scale. These formulations are useful for modeling that help to estimate artificial pond efficiency and to optimize their implantation in agricultural areas.

1.2 Processes governing the fate of pesticides in artificial ponds

The dissipation of pesticides in artificial ponds results from the synergy between bio-physicochemical processes detailed thereafter. Several studies have proved the possibility of quantifying this dissipation and predicting the fate of pesticides by a black-box model (Braschi et al., 2022). However, a black box approach does not explicit the origin of pesticide dissipation and does not detail the contribution of each process since it only considers a single dissipation rate k_{diss} (d^{-1}) (Yuan et al., 2020).

In this review, we will describe and present different estimation methods of processes behind pesticide dissipation in artificial ponds. The form under which pesticides are introduced to the artificial pond depends on how agricultural water is drained into the artificial pond. If the watershed is drained with significant subsurface flow, pesticides are more likely to be dissolved rather than particulate (Braschi et al., 2022; Le Cor et al., 2021). In contrast, pesticides can be present in dissolved and particulate phases if runoff occurs. Gramlich et al. (2018) assumed that subsurface drainage reduces erosion risk, while runoff may increase it. Dissolved Pesticides (DP) in water bodies are widely studied in the literature, whereas few studies focus on Particulate Pesticides (PP) (Climent et al., 2019). PP are transported within the artificial pond along with TSS following the water flow. According to the hydraulic regime of the artificial pond, PP can settle on the sediment bed or be suspended back to the water. In contrast, DP can be bound to the solid surfaces available in the artificial pond through sorption. Sorbed DP can be released back into water through desorption. DP can also be transported to the atmosphere when volatilized or uptakes by the plants growing in the artificial pond. On the other hand, pesticides can be a subject of transformation processes, whose prevailing drivers are microorganisms, solar radiation, and hydrolysis.

Processes involved in pesticides dissipation in artificial ponds depend on many factors such as the characteristics of the soil (i.e., Organic matter (OM), organic carbon (OC), and clay content), water physicochemical properties (pH, temperature (T) and redox conditions), vegetation, and the hydraulic conditions of the artificial pond (Papaevangelou et al., 2017; Vymazal et al., 2015). The mitigation of pesticides in artificial ponds also depends on the pesticide's physicochemical properties. The main physicochemical properties of a pesticide include mobility, hydrophobicity, the solubility in water (S), and the pesticide half-life (DT_{50})

- the time required for the dissipation of 50% of the concerned substance (Gregoire et al., 2009).

The properties of pesticides and the processes behind their dissipation in artificial ponds will be detailed afterward.

1.3 Hydraulics of artificial ponds

The hydraulic performance is a key factor controlling the pesticide dissipation function of a artificial pond. The hydraulic performance of artificial ponds includes lowering peak flow and limiting sediment export from agricultural parcels to natural water bodies (Passeport et al., 2010; Tournebize et al., 2013). Artificial ponds hydraulic parameters include (i) The hydro period (Bojcevska et al., 2007; Convertino et al., 2013; Liu et al., 2019; Prochaska et al., 2007) and (ii) The hydraulic residence time (HRT) of the water within the artificial pond (J. Holland et al., 2004; Kjellin et al., 2007).

The hydraulic performance of artificial ponds depends on design parameters such as depth, the layout of the inlet and outlet, and plant spacing. In this subsection, we will describe the impact of design parameters on the hydraulic performance of artificial ponds.

Multi tracer experiments showed that shallow water depth was an essential factor enhancing sorption and that pesticide load reduction was more significant in shallow artificial ponds than in deep ones (Lange et al., 2011). A low water depth allows a uniform fluid distribution compared to higher depths. The uniformity of the fluid flow decreases the hydraulic dead zones and increases the effective volume, which is the artificial pond volume used for pesticides dissipation (Rayen et al., 2019).

Globally, an artificial pond volume can be divided into three hydraulic zones (C. J. Martinez et al., 2003; Werner et al., 2000):

1. The main channel with the preferential flow.
2. The mixing zone is a temporary storage zone showing exchanges with the main channel.
3. The dead zone is more isolated from the main channel than the mixing zone.

Tracing studies showed that water is transported mainly by convection in the main channel, while it is transported by dispersion in the isolated zones (mixed and dead zones). The HRT in isolated zones is longer than in the main channel. Thus, isolated zones could enhance dissipation processes such as sorption, settling, and degradation (Gaullier et al., 2018). The main channel is where short circuits in the water flow can occur, causing the transport of water from the inlet to the outlet by preferential flow, which decreases the HRT and effective volume of the pond (Vieira et al., 2018). Another controlling parameter of the artificial pond hydraulic performance is the inlet/outlet design. Shilton et al. (2003) reported that positioning the outlet in an isolated zone from the main channel avoids short-circuiting issues and maintains the maximum hydraulic performance of artificial ponds. In addition, vegetation density slows down water velocity and increases the HRT (Gu et al., 2017). Increasing the hydraulic residence time (HRT) increases the contact time between pesticides and pond substrates, giving pesticides more time to be sorbed, settled, or degraded.

A good hydraulic performance also guarantees an optimistic interception of agricultural water by catching the highest concentrations of pesticides (during the post-application period) in minimum water volumes through off/on stream artificial ponds (Passeport et al., 2013; Passeport et al., 2010).

Usually, the risk of transfer of a pesticide is evaluated by the FOCUS models to decide whether to be authorized for use or not. However, since pesticides are still transferred to the environment after their application and their concentration in water resources than the RAC (regulatory acceptable concentrations) (Gassmann et al., 2015; Halbach et al., 2021; Tang et al., 2012), this theoretical study can be improved. Therefore, this work is based on a detailed bibliographic analysis to present the required knowledge to understand the behavior of pesticides after being applied on agricultural plots and intercepted by artificial ponds. This knowledge is useful for developing a simple and robust conceptual model to simulate the fate of pesticides in artificial ponds and then at the catchment scale, like the models developed for nutrients (Hantush et al., 2013; Kalin et al., 2017; Walker Jr et al., 2011). Modeling can also improve the understanding and hierarchization of the dissipation process and optimize its implantation in agricultural areas. Therefore, the next section will describe the key processes behind pesticide dissipation in artificial ponds and their mathematical formulations used in modeling.

2. Description and modeling of dissipation processes in artificial ponds

Modeling is an efficient, comprehensive tool for assessing and setting up facilities to treat contaminated environments (Warren et al., 2002). There are mainly two types of models that describe pesticide fate in the environment, (i) Physical 3D and 2D models that simulate the hydraulic performance of wetlands by considering water velocity profiles and the vegetation impact of the flow (Carleton et al., 2010; Luo et al., 2016; Christopher J. Martinez et al., 2003; Tsavdaris et al., 2013). (ii) Conceptual models that simulate the biogeochemistry and the dissipation performance of the wetland for a large-scale application (Gobas et al., 2018; Willkommen et al., 2018). The asset of this paragraph is to identify and describe processes behind pesticide dissipation in artificial ponds, along with their mathematical formulations established as part of conceptual models.

Our literature investigation showed that researchers had widely discussed the fate of pesticides in artificial ponds. Although, most studies are generally limited to one pesticide or one specific process evaluation and are rarely supported by mathematical formulae. However, we assume that mathematical formulations allow the integration of processes into a computational model to quantify and predict the fate of contaminants in artificial ponds. The originality of this review lies in the description of the main processes influencing the behavior of multiple pesticides in artificial ponds, namely transport, sorption, and transformation processes. The evaluation and formulation of these processes are necessary to develop predictive models for pesticides dissipation in artificial ponds (Papaevangelou et al., 2017; Saaristo et al., 2018). In this section, we will investigate the behavior of multiple pesticides with contrasting physicochemical proprieties and describe different methods currently employed to evaluate and formulate the processes controlling dissipation in artificial ponds.

2.1 Transport of pesticides

Pesticides reaching artificial ponds can be dissolved in water or associated with dissolved organic carbon (DOC). They can also be bound to the organic fraction of suspended matter (i.e., particulate organic carbon (POC)), featuring pesticides in a particulate form. PP are carried along with suspended matter throughout the pond artificial pond due to water movements. PP can be deposited on the sediment bed due to settling and released back to water through resuspension. Settling and resuspension are transport processes that strongly depend on the particle size and the strength of the water flow in pond artificial ponds.

Pesticides can also be the subject of advection, dispersion, and diffusion processes within the water column and pore water of the sediment layer. According to the Hjulstrom curve, the water velocity in the artificial pond can determine which transport process is prevailing (Hjulstrom, 1935).

2.1.1 Settling

Settling is a physical transport process through which suspended particles in water settle on the sediment bed of artificial ponds. Settling depends on the particle size of pesticides, water velocity and turbulence, temperature, and water column depth (Kadlec et al., 2008).

Settling has been efficient in the water-quality improvement function of artificial ponds, especially in lower flows, since suspended particles, on which pesticides may be attached, can leave the water column while settling on the sediment bed (O'Geen et al., 2010). Settling is enhanced when vegetation covers a significant surface of the artificial pond; emergent macrophytes slow down water flow and amplify settling (Vymazal, 2013). In addition, stems and leaves in the water column obstruct water flow and dissipate the energy required to support particle suspension (Schmid et al., 2005). Rose et al. (2008) pond (0.021 h^{-1}). A small water depth was also reported that the settling rate in a vegetated artificial pond (0.042 h^{-1}) was double the rate estimated in a non-vegetated artificial to promote the settling process; it fosters plant growth and shortens the settling distance for particles (Lange et al., 2011). It is valid for water depths between 20 cm and 50 cm (Braskerud, 2002). Pesticides associated with fine suspended particles may hardly be retained in ponds since fine particles (clay and silts) are highly transported across preferential pathways where the HRT is low (see section 1.3) (Maynard, 2009). The settling of PP is more likely to occur in hydraulically isolated zones with lower water flow and higher HRT (Gaullier et al., 2020; Passeport et al., 2010). Ponds with the highest settling rate are the ones receiving agricultural surface runoff (Fiener et al., 2003; Zhang et al., 2011), where the intercepted agricultural water is heavily loaded with coarser (>200 µm) particles from the fields (Li et al., 2017). It is due to the higher TSS load from erosion that is less available in artificial ponds collecting subsurface drainage water (Koskiaho, 2003; Maillard et al., 2011). It was also suggested that coarse particles settle before the water is distributed across pond compartments, unlike fine particles requiring a longer time to settle. As a result, fine particles become more concentrated downstream of the pond (Gan et al., 2005) because gravity induces the settling of heavy particles (>36 µm) (i.e., sand) more than lighter particles (<2 µm) (i.e., clay and OM) (Fiener et al., 2003). Several studies suggested that a large proportion of highly hydrophobic organic substances in artificial ponds were retained by settling pesticide-laden solids (sorbed on suspended solids) (Luo et al., 2009; Matamoros et al., 2006; Rose et al., 2008). Therefore, settling may not be sufficient to trap pesticides with a stronger affinity to fine suspended particles in ponds. Consequently, a higher HRT and vegetation are required to increase the settling of fine PP (Budd et al., 2009, 2011; Fiener et al., 2003).

Settling was commonly integrated with an accumulation rate expressed empirically in pesticide dissipation models. The empirical formulations of settling rates can be found in Table A.1.The first formulae (i) estimates the settling rate v_s according to Stokes equation for laminar conditions (Reynolds number Re <500). Fiener et al. (2003) computed the settling rates of particles of different sizes in a vegetated wetland. It was observed that particles >63 µm (i.e., sand) were fully sedimented, whereas particles 2 µm< x <63 µm (i.e., silt) were moderately sedimented (up to 26%). In contrast, only 4% of fine particles <2 µm (i.e., clay) settled on the sediment bed. It was also noticed that settling primarily occurred on the side-slopes with small Re. The second formulation (ii) suggested by Kadlec et al. (2008) allows calculating the settling rate in both laminar and turbulent flow based on a drag coefficient C_D that depends on Re. Formula (ii) assumes that settling results from density difference between suspended particles and water, so it expresses the rate of a single and isolated spherical particle. The limit of the first (i) and second method (ii) are the required parameters since in artificial ponds, neither the density nor the particle size is known. In addition, particles are not spheres or discs and can agglomerate to a larger size. The formula (iii) estimates the settling rate of suspended particles in wetlands (Hawley et al., 2014). This formulation required more parameters such as the water kinematic viscosity since it is adapted for also non-spherical particles. However, it was often applied to large size particles.

Overall, the formula (i) could be sufficient and easy to use for a simple simulation of PP settling in laminar flows. If the flow is turbulent with a known Re, formula (ii) would be more suitable for its accuracy. For formula (ii), particle sizes are often approximated according to their nature (i.e., clay, silt, sand, and OM). Otherwise, the settling rate could be calibrated if the required data is available. But still, the majority of settling studies are done on nutrients, and little is known about pesticides (Hantush et al., 2013; Kalin et al., 2017; Walker Jr et al., 2011). Regarding behavior, settling seems to be a key process of pesticide accumulation in pond artificial ponds alimented by surface runoff in erosive areas, especially for hydrophobic pesticides and coarse PP.

2.1.2 Resuspension

Resuspension is the opposite process of settling, through which PP are transported from the sediment bed to water. Resuspension depends mainly on the water flow and the water depth in artificial ponds.

Resuspension is likely to occur when water turbulence is important (Siobhan Fennessy et al., 1994), and high water flow can remobilize settled PP back to the water (Collins, 2004; Jamieson et al., 2005; Vallet, 2011; Wilkes et al., 2009). It was proved that resuspension was highly affected by the type and density of vegetation in artificial ponds (Liu et al., 2019; Schulz et al., 2001). Braskerud (2001) presumed that an increase of 30% in a vegetative cover, initially less than 20%, could reduce resuspension rates from 40% up to 100%. Rose et al. (2008) showed that the resuspension rate was divided in half by switching to a vegetated artificial pond. Some studies suggested that deeper FWS wetlands enhanced settling and burial rather than resuspension (Brueske et al., 1994; Budd et al., 2011). Resuspension is likely to occur in the main channel where the water velocity is fast (see section 1.3) (Gaullier et al., 2020) and is mainly driven by wave action generated by the wind (Hawley et al., 2014). Other agitation factors may be responsible for particle resuspension, such as wildlife movements within the artificial pond (i.e., bioturbation), although quantitative studies on this topic remain scarce. Resuspension was significantly less considered in the literature than settling. However, resuspension occurred with a slow rate of remobilization (v_r ~10^{-6} mm/s) (Hawley et al., 2014; Sharifi et al., 2013), while Rijn (1984) suggested that the resuspension rate v_r reached the quarter of the settling rate v_s.

Globally, significantly lower concentrations of PP measured at the outlet of wetlands than at the inlet suggests that the contribution of resuspension in pesticides behavior might be negligible (Budd et al., 2009; Cryder et al., 2021; Supowit et al., 2016), also resuspension is more dominant in systems with stronger water flows (i.e., rivers) rather than artificial ponds.

In addition, high water flow events are usually short and have a long return period, which decreases the occurrence of resuspension.

2.1.3 Advection, dispersion and diffusion

Among the processes behind pesticide transport within the artificial pond are advection, dispersion including diffusion. Advection carries pesticides in the same direction as the water flow. Dispersion results from heterogeneities in flow patterns and is a macroscopic process. In contrast, diffusion is a microscopic movement of pesticide molecules due to a random molecular motion (i.e., Brownian motion) (Hamdi, 2007). Dispersion and diffusion processes result from a concentration gradient of pesticides in artificial pond water and carry pesticides from highly concentrated areas to low concentrations. Advection, dispersion, and diffusion allow the homogenization of DP concentrations in the artificial pond.

Gaullier et al. (2019) showed that water transport is mainly governed by advection at a high flow rate. While at low flow rates, both water transport and pesticide mitigation are governed by hydrodynamic dispersion, including molecular diffusion, resulting in a longer contact time between pesticides and substrate. Without agitation, pesticide transport is only influenced by diffusion, while agitation enhances advection and dispersion (Gaullier et al., 2018). Thus, advection/dispersion is more important in the main channel, while diffusion is more likely to occur in the dead zones (section 1.3). Moreover, diffusion is slower in the sediment layer pore water because of its tortuosity and the increased length of the water pathway between pores (Beulke et al., 2004). When the hydraulic regime is laminar, the diffusion is on the molecular scale, with a thermal diffusion of the order 1.4 x $10^{-7} m^2 s^{-1}$ in pure water. In river water systems, the water is always turbulent. Thus, the effective diffusion is several orders of magnitude greater. Still, the given thermal diffusion is considered in wetlands, where water disperses horizontally, and the flow rate is lower than in rivers (Damalas et al., 2011). For pesticide molecules, the diffusion coefficients usually range from 10^{-11} to 10^{-10} m^2/s (Kipkoech, 2019). Dispersion, including diffusion, also depend on the internal structure of the artificial pond. For instance, the dispersion was proved to be greater in artificial ponds with a larger width (Gaullier et al., 2019), while macrophytes enhance diffusion and the hydraulic performance of the artificial pond (J. F. Holland et al., 2004).

In a pesticide transport equation, advection is expressed by the term $-\nu\frac{\partial C_w}{\partial x}$ and dispersion is expressed by$D_L\frac{\partial^2 C_w}{\partial x^2}$. Where ν $[L.T^{-1}]$ is the water velocity; C_w $[M.L^{-3}]$ is the pesticide concentration in water; D_L $[L^2.T^{-1}]$ is the hydrodynamic dispersion coefficient; with $D_L=D_e+\alpha_L\nu$, where D_e the effective diffusion coefficient, and $\alpha_L[L]$ the dispersivity.

2.2 Transfer of pesticides

Once pesticides reach the artificial pond, they are distributed among water, sediments, TSS, and vegetation are transferred among artificial pond compartments. DP can be exchanged between the water column and the solid compartments due to “Sorption” and the atmosphere during “Volatilization”. Also, DP can become PP when sorbed to TSS.

2.2.1 Sorption process

Sorption was commonly reported as an important process in the fate of pesticides in artificial ponds (Cryder et al., 2021; Fernández-Pascual et al., 2020; Hand et al., 2001). In the following two subsections, we will first present the sorption theory and analyze study cases of pesticide sorption in artificial ponds. This review will focus on sorption information directly related to pesticides' fate in artificial ponds; more detailed knowledge of sorption can be found in the literature (Huang et al., 2003; M. McBride, 1994; Pignatello, 1989; Sposito, 2008)

2.2.1.1 Sorption theory

Physically, *sorption* refers to all exchange processes between a dissolved substance and solid matter in a solution. Among these exchanges, adsorption and desorption remain the most studied for pesticide fate in the environment. Adsorption is a physical transfer process of retaining dissolved substances in water on solid surfaces. The dissolved substances are the "sorbate," and the solid surface to which they are bound is called "sorbent" or "substrate." *Desorption* is when the substrate releases the sorbate and returns to water (Sposito, 2008). From now on, the term "sorption" will be used to refer to adsorption plus desorption.

Most of the sorption studies are conducted in the laboratory and adapted from OECD 106 guidelines for individual molecules, in which sorption experiments are performed using a batch equilibration technique (OCDE, 2000). The retention and release of the sorbate on a substrate at various concentrations and a constant temperature are described by a curve called "adsorption isotherm" and "desorption isotherm," respectively. The isotherms represent the relationship $C_{se}=f(C_{we})$ during adsorption and desorption equilibrium, where C_{se} ($\mu g.g^{-1}$) and C_{we} ($\mu g.L^{-1}$) are the sorbate concentration in the substrate and the sorbate concentration in the solution at the equilibrium state, respectively.

Based on the thermodynamic equilibria theory that assumes complete reversibility of the chemical reactions, the adsorption and desorption isotherms should be the same (Strawn et al., 1999). However, sorption mechanisms are driven by different kinetically controlled reactions or physical phenomena with a large variability of reaction times going from seconds to years, emphasizing the kinetic character of sorption (Beulke et al., 2004; Caceres-Jensen et al., 2019; Cara et al., 2021; Kaur et al., 2018; Mamy et al., 2007; Sparks, 2000; Villaverde et al., 2009; Wu et al., 2017).

A difference between adsorption and desorption kinetic rates to reach the equilibrium is translated by a mismatch between the adsorption and desorption isotherms curves. This mismatch is called "pseudo-hysteresis" or 'kinetic hysteresis" (Strawn et al., 1999), also referred to as an "apparent irreversibility"(Bladel et al., 1967; M. B. McBride, 1994). It is important to clarify that the term "irreversibility" does not mean that the sorbate will not desorb after its adsorption. Still, it will take a long time to reach the equilibrium state (Limousin et al., 2007). In other terms, hysteresis means that desorption requires a more significant amount of energy than needed by adsorption (Bowman et al., 1985; Koskinen et al., 1979; Wu et al., 1988), which could be caused by experimental artifacts or transformation of the sorbent (Pignatello et al., 1996).

The evaluation of sorption requires the determination of its equilibrium and kinetic properties (Douven et al., 2015). Sorption kinetics are studied with the most applied chemisorption models: Pseudo-first-order (PFO), pseudo-second-order (PSO), and Elovich models (Azizian, 2004; Ho, 2006; Ho et al., 1998, 1999; Yuh-Shan, 2004), to describe the temporal evolution of the sorbed amount of the sorbate. At the same time, sorption equilibrium is commonly modeled by the Freundlich and Langmuir isotherm models (Azizian et al., 2018; Foo et al., 2010; Mckay, 1995). As the equilibrium of kinetic sorption is reached, isotherm models are applied. The corresponding equations of kinetic and isotherm models can be found in Table A.2.

2.2.1.2 Sorption of pesticides in artificial ponds

In the case of artificial ponds, the sorbates are dissolved pesticides (DP), and substrates are total suspended solids (TSS) with organic and inorganic compounds, sediments, and vegetation (Gregoire et al., 2009; Sahin et al., 2020), which represent development support for OM and biofilm. The Freundlich and the linear model are commonly used isotherm models to describe pesticide sorption on artificial pond substrates (Liu et al., 2019; Takagi et al., 2012; Watanabe et al., 2006; Zhang et al., 2018). Pesticide adsorption is widely described by the adsorption coefficient normalized to substrate organic carbon content K_{foc} and K_{oc} for the linear isotherm:

$$K_{foc} = \frac{K_{fads}}{OC} \qquad (1)$$

$$K_{oc} = K_d/OC \qquad (2)$$

Where OC (%) is the organic carbon content of the pond substrates, K_{fads} is the Freundlich adsorption coefficient, and K_d is the distribution coefficient (Table A.2).

Vallée (2015) and Passeport et al. (2011b) studied the sorption of 12 pesticides with contrasting physicochemical properties on pond substrates (i.e., sediments, vegetation, and TSS). Sorption coefficients can be found in Table A.3. Based on the K_{fads} values, epoxiconazole was the highly adsorbed pesticide, followed by prochloraz. Epoxiconazole and prochloraz have a K_{foc} range of 702-1962 $mg^{1-n}{}_{f}.L^{n}{}_{f}.kg^{-1}$and 1222-8654$mg^{1-n}{}_{f}.L^{n}{}_{f}.kg^{-1}$, respectively (Lewis et al., 2016), which, according to the Soil Survey and Land Research Centre (SSLRC) soil persistence classification system, means that they are lowly mobile molecules (PSD, 2005). In contrast, the lowest sorption concerned isoproturon, with the lowest K_{foc} values 55-60 $mg^{1-n}{}_{f}.L^{n}{}_{f}.kg^{-1}$ (Vallée, 2015), followed by MCPA. The low adsorption of MCPA could be due to its anionic form, inducing an electrostatic repulsion of negatively charged particles (i.e., clay and OM), as supported by Thorstensen et al. (2001). Likewise, Poissant et al. (2008) and Wu et al. (2004) reported that pesticides with higher K_{oc} values (K_{oc} > 500 $L.kg^{-1}$) are most likely to be sorbed to pond solid particles, agreeing with previous findings (Papp, 2009; Saripalli, 1997). The low solubility of epoxiconazole and prochloraz also favored their retention on pond substrates. The literature widely reported that low solubility enhances pesticide adsorption (Blachnio et al., 2020; Brusseau et al., 2019; Kah et al., 2006). Sorption also depends on the hydrophobicity of the pesticide expressed by the octanol-water partition coefficient K_{ow}. Hijosa-Valsero et al. (2016) analyzed fifty-nine compounds in water and sediment samples. They found that herbicides in the sediments had an average S=194 $mg.L^{-1}$ and a log K_{ow} = 3.7, while herbicides in the water phase were highly soluble S = 1395 $mg.L^{-1}$ with an average log K_{ow} = 2.24. These results support that lowly soluble, and hydrophobic pesticides (log K_{ow} > 3) are likely to be sorbed on pond sediments, which agrees with O'Geen et al. (2010), that observed a 50% reduction in pesticide concentration with log K_{ow} > 4.2. Likewise, Tang et al. (2017) found that adsorption rates of hydrophobic pesticides were significantly higher on the artificial pond vegetation (5.52–8.84 g m−2 d−1) than hydrophilic pesticides (log K_{ow} <3). These results suggest that sorption may be the primary driving process of removing hydrophobic pesticides from the artificial pond water.

On the other hand, vegetation, biofilm, and the OM and clay content of the substrate have a significant role in pesticide sorption in artificial ponds. Barchanska et al. (2019) observed that the artificial pond remediation of pesticides due to adsorption on vegetation reached 30%. An extensive vegetation cover can enhance adsorption by increasing the HRT Gaullier et al. (2020). A higher HRT increases the contact time between pesticides and pond substrates, favoring adsorption and generally decreasing desorption, as observed by (Mamy et al., 2007; Olvera-Velona et al., 2008).

Passeport et al. (2011b) and Vallée (2015) observed higher K_{fads} on vegetation than in sediments (Table A.3), showing greater retention on OC-rich substrates. Moore et al. (2007) and Sahin et al. (2020) also proved that diazinon preferentially sorbed on vegetation surfaces rather than sediments. In like manner, Rogers et al. (2009) showed that the lowly mobile chlorpyrifos (K_{oc} =5509 $L.kg^{-1}$ (Lewis et al., 2016)) adsorption was ten times higher on vegetation than on sediments, and several other papers demonstrated that adsorption is more dominant in OC-rich substrates, including vegetation (Guo et al., 2000; Mahabali et al., 2014; Moore et al., 2007; Mugni et al., 2011). Biofilm was also proven to enhance adsorption for certain pesticides (Agudelo et al., 2012; Beecraft et al., 2021).

Desorption was also evaluated and was reported to be less dominant in sediments with high clay and OC content, especially for lowly soluble pesticides (Gramlich et al., 2018; Olvera-Velona et al., 2008; Passeport et al., 2011b; Song et al., 2021; Vallée, 2015). In comparison, greater desorption was observed on some highly soluble pesticides (Olvera-Velona et al., 2008). Hence, hysteresis was generally observed on OC and clay-rich substrates from which hydrophobic and lowly soluble pesticides are hardly desorbed (Hand et al., 2001; Li et al., 2003; Vagi et al., 2010).

In addition, kinetic sorption rates showed that desorption was slower than adsorption for six evaluated pesticides, except for picloram. The reviewed kinetic rates of adsorption and desorption can be found in Table A.4. The picloram exception can be explained because its sorption was evaluated in lab conditions. Since, in the experimental sample, there was no consideration of other processes that may compete with desorption (i.e., transformation). The interaction between desorption and transformation was also observed by Sørensen et al. (2006), noticing that slower desorption of MCPA (coincided with higher microbial transformation. In fact, unlike laboratory-batch sorption observations, sorption and transformation processes co-occur under field conditions, favoring the occurrence of hysteresis in the presence of other competitive processes like transformation and diffusion into sediment layer micropores (Mamy et al., 2007; Paszko et al., 2016; Takagi et al., 2012; Uchimiya et al., 2012). In contrast, advection/dispersion enhance sorption by facilitating the access of DP to the pond substrates (Gaullier et al., 2018).

Overall, we assume that both adsorption and desorption depend on the clay and OC content of pond substrates and the properties of pesticides. Adsorption is more dominant for hydrophobic, lowly mobile, and lowly soluble pesticides, and desorption hysteresis is more accentuated in clayey and OC-rich pond substrates, especially for highly soluble pesticides.

2.2.2 Volatilization

Volatilization is a process through which pesticides are transferred outside the pond, from the water to the atmosphere. This non-degradative removal process strongly depends on the pesticide molecule's vapor pressure.

There are two regimes controlling volatilization: the atmospherically controlled regime and the regime controlled by water properties. Most pesticides (80-90%) belong to the atmospherically controlled regime, where the gas exchange between the water and the atmosphere is only sensitive to the parameterization of atmospheric turbulence, and the majority of pesticides belong to it; therefore, we will focus on the atmospherically controlled regime. In the atmospherically controlled regime, at a very low dimensionless Henry constant ($<10^{-7}$) (equivalent to a vapor pressure vp $<10^{-5}$ Pa), the volatilization rate is so slow that the volatilization half-life is tens to hundreds of years. In that case, volatilization becomes insignificant compared to other processes (Jacobs et al., 2012). In that respect, Höllrigl-Rosta (2017) classified of volatilization according to the vapor pressure of the pesticide molecule (Table A.5).

In modeling, volatilization is often described by a first-order kinetic equation (Barchanska et al., 2019; Comoretto et al., 2008; Gobas et al., 2018; Inao et al., 1999; Singh et al., 1991; Watanabe et al., 2006) :

$$\frac{dMw}{dt} = -k_v * Mw \quad (3)$$

Where Mw is the mass of the DP, and k_v (d^{-1}) is the pesticide rate constant of volatilization to the atmosphere. Volatilization can also be represented by its corresponding half-life DT_{50v} (d):

$$DT_{50v} = \frac{\ln{(2)}}{k_v} \quad (4)$$

Since this transfer process depends on other factors, the volatilization rate may be expressed empirically as a function of both pesticide properties and climate conditions. Table A.6 are simplified form of the rate of volatilization k_v used in descriptive models of the fate of pesticides. In other studies, there are many complicated formulations (Bunyakan et al., 2006; Comoretto et al., 2008), in which k_v depends on diffusion coefficients, air-water side, water-air side mass transfer coefficient, and friction. However, they are usually left for volatilization-specific studies and do not model the fate of pesticides in artificial ponds.

A bibliographic review of calculated volatilization half-lives DT_{50v} (d) of certain pesticides can be found in Table A.7. For most pesticides in artificial ponds, the vapor pressure is lower than 10^{-5} Pa, which enables to neglect the contribution of volatilization to the artificial pond dissipation function according to the classification in Table A.5. However, it was demonstrated that some pesticides were efficiently removed from water by volatilization (Wang et al., 2014; Wang et al., 2019). In an aquatic microcosm under semi-field conditions, Laabs et al. (2007) showed that volatilization was a primary reason for high pesticide losses from the water shortly after application, especially for non-polar pesticides (i.e., endosulfan, chlorpyrifos, and trifluralin). Similarly, Comoretto et al. (2008) reported that oxadiazon had a DT_{50v} of 23 days in a rice field, making it more vulnerable to volatilization when compared to stable MPCA and pretilachlor. k_v rates were either calculated using the empirical formulas in Table A.6 or estimated in laboratory experiments. Empirical formulations of k_v show that volatilization rates are mainly dependent on water depth, a mass water-air transfer coefficient that is a function of the wind speed, the Henry constant (or vapor pressure and solubility), and the molecular weight of the pesticide. Pesticides with a vp lover than 10^{-5} require years to be volatilized according to volatilization half-lives in Table A.7, making them virtually non-volatile. In contrast, other pesticides such as alphamethrin, molinate, endosulfan, and dieldrin need a few days to volatilize, while oxadiazon requires more days to volatilize. With careful consideration of the vp of these substances, we assume that these assumptions agree with the classification in Table A.5. These observations showed that most pesticides are non-volatile due to their low vapor pressure. However, there are still some for which volatilization takes part in their dissipation in artificial ponds.

2.3 Transformation of pesticides

In addition to the transport and transfer processes described in sections 2.1 & 2.2, pesticides face other processes that transform their original chemical structure. They are also designated as “degradative” processes when attempting to degrade pesticides to non-organic products (Cessna et al., 2015; Imfeld et al., 2021). The importance of transformation processes in the artificial pond dissipation efficiency was widely proved. Pesticides can be transformed into less toxic products or, in the ideal case, wholly degraded to simple mineral products under the action of the chemical, photochemical, and biological processes (Tournebize et al., 2017). This section describes the main biotic and abiotic transformation processes to which pesticides are exposed in the different compartments of the artificial pond.

2.3.1 Biotransformation

Biotransformation is a process by which organic substances are transformed by microorganisms (bacteria, archaea, and fungi) into chemical substances called metabolites, resulting in some cases in their complete mineralization (Garcia-Rodríguez et al., 2014). If the microbial transformation is complete, it becomes a transformation, and pesticides are mineralized to CO_2 and CH_4. Otherwise, co-metabolism occurs, during which microorganisms degrade OM substrates, the primary source of nutrients of the bacteria, primarily to produce enzymes that will break down sequentially the pesticide molecule, representing a secondary source of nutriments of the bacteria (Arcangeli et al., 1997), resulting in a metabolite.

Recently, in a artificial pond system context, some metabolites were less toxic than their parent pesticides (Kang et al., 2020). However, it remains unknown if other metabolites are less or more toxic than the mother compounds. Nonetheless, biotransformation has been extensively shown as a significant removal process for micro-organic pollutants (Hijosa-Valsero et al., 2010; Langergraber, 2008; Llorens et al., 2011; Lyu et al., 2018; Samso et al., 2013; Yang et al., 2017; Zhang et al., 2017).

Biotransformation occurs in both aerobic and anaerobic conditions. Aerobic biotransformation occurs in oxic conditions, mainly near the plant roots or the water surface, where the amount of dissolved oxygen is high enough. Biotransformation is anaerobic in deep water and sediment depths with generally dissolved oxygen concentrations below 0.5 $mg.L^{-1}$ (U.S.G.S, 2020). The rhizosphere can also be considered as an aerobic zone since oxygen can be transported from the plant body to the roots (Maltais-Landry et al., 2009), but researchers showed that the amount of oxygen released by the plant was too little to be considered in artificial pond fate models (Bezbaruah et al., 2005; Llorens et al., 2011). Artificial ponds are mostly aerobic in the water column surface, with anaerobic conditions occurring at the bottom sediment layer and deep water (Vymazal et al., 2015). The sediment layer was proved to be characterized by a sharp decrease in oxygen (Martínez-Lavanchy et al., 2015; Stottmeister et al., 2003; Thomas et al., 2011; Zhang et al., 2010). Karpuzcu et al. (2013), Paszko et al. (2016) and (Mulligan et al., 2016); Paszko et al. (2016) demonstrated that anaerobic conditions would result in lower biotransformation rates than under aerobic conditions. It agrees with the recent study of commonly used herbicides in which Droz et al. (2021) indicated that acetochlor and S-metolachlor transformation prevailed in the water phase and seemed insignificant in the sediment.

Microbial transformation also depends on temperature and radiation (Burrows et al., 2002), pH, OM content, and microbial population (Vink et al., 1997). Sahin et al. (2020) reported that humic substances increase biotransformation rates since they can carry microorganisms to the pesticide. This effect has been widely investigated (Cai et al., 2017; Ortega-Calvo et al., 1998; Smith et al., 2009; Tejeda-Agredano et al., 2014), mainly for hydrophobic pesticides. For instance, atrazine was highly biodegraded when sorbed onto clay-humic acid complexes (Besse-Hoggan et al., 2009). Likewise, Rong et al. (2019) found that the interfacial reactions on clay surfaces increased the biotransformation rates of methyl parathion. The exposure history to pesticides of artificial ponds also influences biotransformation. Johnson et al. (2004) demonstrated that sites with a higher pesticide exposure history had higher biotransformation rates due to microbial acclimation and microbial community adaptation.

The water level is another controlling factor of biotransformation in the sediment compartment of artificial ponds. Ma et al. (2018) speculated that a decline in the water level enhances microbial growth and activity and would modify the structure and function of microbial communities. The study also emphasized that a water volume drop-off would enhance aerobic conditions, induce plant succession to the artificial pond center, and stimulate OM accumulation and mineralization.

The aging of sediments also intensifies the sequestration of PP after their settling, thus decreasing their bioavailability and hindering microbial transformation (Ahmad et al., 2004). Budd et al. (2011) proved that once pyrethroids were settled, they were stable in dry and aerobic sediments between irrigation seasons. They were moderately persistent under flooded and anaerobic conditions, with a DT_{50} between 106-353 days. This agrees with the observation of Cryder et al. (2021), showing that adsorption and subsequent settling were the dominant processes governing the fate of pyrethroids in ponds instead of biotransformation.

Vegetation has been widely recognized as an important stimulator of biotransformation (Moore et al., 2007; Rai, 2009; Stang et al., 2016; Sur et al., 2003; Wang et al., 2017). For instance, chlorpyrifos removal rate reached 0.1 $mg.m^{-1}.day^{-1}$ due to biotransformation in vegetated mesocosms (Tang et al., 2019). Submerged stems and leaves of macrophytes are an essential habitat for biofilms (Brix, 2003; C. Zhang et al., 2021). These biofilms, along with those on all other immersed solid surfaces in the artificial pond system - including dead macrophytes – are stimulators of microbial processing in artificial ponds (Brix, 2003; Dash et al., 2020; Lv et al., 2017). Several kinds of phototrophic bacteria, algae, and biofilms in the water-sediment interface and on the surface of macrophytes were proved to be important agents of pesticides transformation in artificial ponds (Katagi et al., 2016). Although, biotransformation in the biofilm can be limited sometimes due to the slow kinetics of mass transfer or enzymatic reactions (Chen et al., 2015).

In modeling, biotransformation is often described by a first-order kinetic model (Beringer et al., 2021; Green et al., 1990; Mamy et al., 2007; Pignatello, 1999; Scow et al., 1996; Torabi et al., 2020). Where k_{bio} (d^{-1}) is the biotransformation kinetic constant in the compartment in question. Biotransformation can be represented either by the kinetic constant k_{bio} (d^{-1}) or by the corresponding half-life DT_{50bio} (d) (see Eq.(3) and Eq.(4)).

Several studies managed to estimate the biotransformation rate constants in artificial ponds in laboratory conditions, through incubation manipulations. Incubation experiments aims to calculate pesticide concentrations in the different compartments of a contaminated sample (water and sediments). Then, the concentrations are fitted to a first-order kinetic decay model to estimate the rate constants k_{bio} and half-lives DT_{50bio}. Table A.8 presents a literature review of estimated biotransformation half-lives DT_{50bio}.

The values of biotransformation half-lives (Table A.8) show that hydrophobic pesticides pretilachlor and fenthion were rapidly biodegraded in sediments; we suggest that it may be related to the fact that they were mainly sorbed on substrates easily accessible to microorganisms. Pretilachlor was more rapidly degraded in water ($DT_{50bio,w}$ = 9.8 d). Even though being hydrophobic and lowly mobile (K_{oc} = 5509 $L.kg^{-1}$), chlorpyrifos was more persistent in sediments than in water columns, and was favorable to aerobic conditions (O'Geen et al., 2010). Comparing the behavior of pretilachlor, fenthion, and chlorpyrifos, that hydrophobicity, and low mobility may not necessarily favor biotransformation in sediments rather than water. On the other hand, biotransformation in sediments seemed relatively slower for more mobile pesticides (i.e., 2, 4-D, clothianidin, acetochlor, metazachlor, and S-metolachlor), except for the MCPA. We suggest that MCPA was not expected to sorb on TSS and sediments due to its low K_{oc}. Its transformation might be more favorable under aerobic conditions (on the water surface) (PubChem, 2021), explaining its short half-life in the water phase. S-metolachlor was moderately persistent in both water and sediments; this may be explained by its complex chemical structure that forbids access to enzymes and prevents its transformation under aerobic and anaerobic conditions (Droz et al., 2021). Finally, DT_{50bio} values showed that biotransformation was faster in oxic areas, except for acetochlor ($DT_{50bio,w}$ = 47 d) and clothianidin ($DT_{50bio,w}$ = 27.2 d) that were rapidly transformed under anaerobic conditions. Although, it is not possible to infer any direct link between the OC content and the biotransformation rate since sites were in the same range of OC.

Compared to the values of the Pesticide Properties Database (PPDB) (Lewis et al., 2016), biotransformation half-lives were faster when estimated in artificial ponds, and for now, we cannot make any conclusion about this difference since the conditions to estimate DT_{50bio} in PPDB have not been detailed.

Overall, biotransformation is commonly expressed by a first-kinetic order model and enhanced by increased vegetation within the artificial pond providing OM that supports microbes and increases the HRT. Therefore, we suggest introducing a modulation coefficient according to vegetation cover and OC content in water and sediments of a specific site will increase k_{bio} representability. At this time, no rule has been set up for pesticides biotransformation in artificial ponds: neither a direct link with the pesticide properties has been highlighted because biotransformation does not act alone, but simultaneously with other processes. In the future, a more profound exam of biotransformation rates of different pesticides accounting for their interaction with other processes (i.e., adsorption and settling, adsorption/desorption, and dispersion/diffusion) is needed. There is a lack of information concerning biotransformation in vegetation/biofilm compartments. Therefore further studies should be carried out to estimate corresponding transformation rates and account for a possible lag phase reflecting microbial acclimation, as shown by Passeport et al. (2011a). In pesticides databases (i.e., PPDB) there is no distinction between the different compartments (water, sediments, and vegetation) for biotransformation half-lives, making it hard to compare and confront the rates measured as a part of artificial pond surveys with those in pesticides databases.

Globally, biotransformation remains a highly changing phenomenon that depends on both the pesticide properties and the site conditions in which microorganisms can act. Defining specific biotransformation half-life for artificial ponds, would be more informative than the PPDB DT_{50bio} values to estimate the contribution of biotransformation to overall dissipation in artificial ponds.

2.3.2 Photodegradation

Photodegradation or photolysis transforms pesticides exposed to sunlight, especially at wavelengths below 400 nm. The products formed by sunlight-induced transformations are known as photoproducts (Gerecke et al., 2001). Photodegradation is one of the significant abiotic degradation processes in artificial ponds (Lin et al., 2005; Papaevangelou et al., 2017; Stangroom et al., 2000).

Photolysis can be direct or indirect. During direct photolysis, pesticides absorb light energy and become excited; they can be transformed depending on the availability of the activation energy. In indirect photolysis, the pesticide reacts with other species produced photochemically (i.e., $CO3\bullet^{-}$, hydroxyl radical $OH\bullet$, singlet oxygen $O^{1}{}_{2}$, and triplet-excited state dissolved OM (DOM)), causing degradation or conversion into other products (Katagi et al., 2016; Miller et al., 2005; Pohlman et al., 1983; Zeng et al., 2013; Zepp et al., 1987). $OH\bullet$ can be produced by nitrate photolysis, even at a concentration as low as 0.02 $mg.L^{-1}$ (Vione et al., 2014). DOM generates DOM*, the precursor of singlet molecular oxygen ($O^{1}{}_{2}$) in surface waters. DOM* and $O^{1}{}_{2}$ can react with pesticides (Drouin et al., 2021). The most reactive species is $OH\bullet$ attacking C-H bonds and mainly controls pesticide photodegradation.

Because of the non-ability of solar radiation to penetrate deeper into the water, direct photolysis remains efficient only in the first centimeters of the artificial pond (Frank et al., 2002). The effectiveness of photodegradation strongly depends on the shading capacity of the vegetation canopy and turbidity within the water column, which may attenuate the penetration of solar radiation (MacIntyre et al., 2006; Rose et al., 2008). This transformation process was also reported to be positively correlated to the artificial pond water pH (Le Person et al., 2016). In natural waters, DOM and inorganic compounds are essential absorbents of sunlight and influence the photochemistry of pesticides in artificial pond (Burrows et al., 2002). In addition, particulate matter such as sediment particles and microorganisms suspended in the water column may scatter incident light and prevent the light from penetrating the artificial pond. This phenomenon is called the quenching effect of OM (Larson et al., 1994; Pattanayak et al., 2018; Ruiz-Medina et al., 2018). Photochemical degradation would be more efficient in an open water artificial pond with low turbidity (O'Geen et al., 2010; Vymazal et al., 2015). In DOC-rich ponds (DOC > 20 $mg.L^{-1}$), photolysis is unlikely to be operative in pesticide dissipation since DOC content massively attenuates UV light transmission to DP (Waiser et al., 2004). Lagunas-Allué et al. (2010) observed less than an 8% decrease of boscalid caused by direct photolysis, while the main removal processes were adsorption on OM and plant uptake. This observation can be explained by the high OM content that prevents light penetration to pesticides and limits direct photolysis.

On the other hand, several herbicide were efficiently removed by indirect photolysis, such as atrazine, diuron, and chlorpyrifos in a study conducted by Zeng et al. (2013). Mathon et al. (2019) evaluated the photodegradation of 23 pesticides using in-situ photo-reactors. They found that 15 pesticides were photodegraded in the summer against 12 in winter, which showed that photolysis was not directly linked to the season. However, when Mathon et al. (2019) compared each of the 23 pesticides photolysis direct and direct+indirect experimental half-lives, they found that indirect photolysis enhances the degradation compared to direct photolysis. Additionally, the contribution of direct and indirect photolysis was studied for 60 pesticides. Direct photolysis predominated for dinitroaniline herbicides, although more than 50% of the total photodegradation rate could be attributed to indirect photolysis for the other pesticides. This observation supports the relevance of indirect photolysis in artificial ponds when light attenuation is minimal (Katagi et al., 2016; Zeng et al., 2013).

As with most transformation processes, photodegradation was commonly computed with a first-order kinetic model (Green et al., 1990; Lam et al., 2003; Rose et al., 2008; Watanabe et al., 2006; Zeng et al., 2013), where k_{photo} (d^{-1}) is the photodegradation kinetic constant, assumed to be the sum of direct and indirect photolysis kinetic rates (Mathon et al., 2019), and $DT_{50photo}$ (d^{-1}) is the photodegradation half-life. Photolysis half-lives that fitted a first-order kinetic model in pesticide fate models are presented in Table A.9.

Photodegradation half-lives show that the slowest pesticides to be photodegraded were mobile and moderately mobile pesticides, according to their K_{oc} values. In contrast, while lowly mobile pesticides were rapidly photodegraded (Table A.9). Since lowly pesticides are likely to be sorbed (see section 2.2.1), we propose that they might be indirectly photodegraded once sorbed on sediments, suspended solids, or plants through chemical reactions with free radicals in the water-substrate interface.

Overall, since the penetration of solar radiation to deeper water is limited, direct photolysis is considered only in the first centimeters of the artificial pond, depending on the DOC content. In contrast, indirect photolysis involving excited radicals may be more relevant through photochemical degradation and is well described by a first-order kinetic rate. We suggest that introducing a modulating factor taking into consideration vegetation cover and OM limit light penetration will increase the accuracy of photolysis modeling.

2.3.3 Hydrolysis

Another abiotic transformation process is the hydrolysis of DP, which depends on the pH and temperature of artificial ponds. Hydrolysis was commonly considered a minor dissipation process when compared to adsorption, microbial and photochemical degradation. However, in some cases, hydrolysis played a significant role in the dissipation function of artificial ponds, especially under alkaline conditions (Bondarenko et al., 2004; Liu et al., 2001; Papaevangelou et al., 2017; Yu et al., 2019).

For instance, chlorpyrifos was proved sensitive to hydrolysis (Liu et al., 2001; Wu et al., 2002). Experimental studies showed that higher pH (> 8) increased the hydrolysis rate of chlorpyrifos (Agudelo et al., 2010; Baskaran et al., 2003; Sardar et al., 2005), corresponding to a half-life of 50 days (Liu et al., 2019; Mackay et al., 2006). Similarly, carbaryl insecticide was mainly degraded by photolysis and hydrolysis in a pond (Wolfe et al., 1978). Hydrolysis mainly contributed to endosulfan dissipation (Walse et al., 2002) and dichlorvos had a hydrolysis half-life of 3 days (Tomlin, 2009). In contrast, hydrolysis contributed to less than 2% of boscalid degradation (Lagunas-Allué et al., 2010). Similarly, the hydrolysis half-life of dieldrin exceeded three months (Singh et al., 1991) and reached 115 and 138 days for diazinon and fenthion, respectively (Ibanez et al., 2006; Mackay et al., 2006; Sakellarides et al., 2003). S-Metolachlor and butachlor were also proved to be persistent to hydrolysis with a half-life of 200-1155 days at different pH (Masbou et al., 2018; Zheng et al., 2001). Sometimes, change in diurnal pH due to photosynthesis by aquatic algae and macrophytes may result in photo-synthetically-driven alkaline hydrolysis of some pesticides (Katagi, 2006, 2013).

Likewise, hydrolysis can be estimated with a first-order kinetic model (Rose et al., 2008; Wang et al., 2019), where $k_h(d^{-1})$ is the hydrolysis kinetic constant and DT_{50h} (d^{-1}) is the hydrolysis half-life. It was reported that the main controlling factor of hydrolysis is pH (Deer et al., 2001; Farran et al., 1988; Sereshti et al., 2021; Stangroom et al., 2000).

Overall, most pesticides that threaten water quality were considered insensitive to hydrolysis. It may be due to their chemical structure, such as the lack of methyl ester linkage for sulfosulfuron herbicide (Sabadie, 2002; Saha et al., 2002) and phosphodiester linkages for organophosphate pesticides (Sahin et al., 2020). In addition, artificial pond water might generally be beyond the range of alkaline or acidic conditions, making hydrolysis a very slow process (Rose et al., 2008). Therefore, we assume that hydrolysis could be neglected in modeling pesticide dissipation in artificial ponds.

3. Discussion

After reviewing various research works about pesticide dissipation in artificial ponds, it appears that some studied different processes, while others only focused on one specific process. Similarly, some surveys investigated the behavior of multiple pesticides while others only evaluated one or multiple pesticides from the same chemical group (see section 2). Papers evaluating multiple processes often studied one process. Therefore for a better understanding of artificial pond processing, this review combines the investigation of multiple pesticides behavior with contrasting physicochemical properties and evaluates transport, transfer, and transformation processes occurring in the artificial pond compartments (water, sediments, and vegetation). In the following subsections, we discuss this critical review's main assumptions regarding the processes' controlling factors, the estimation of parameters required to model processes, and the hierarchization of processes.

3.1 Controlling factors of the processes

As described in section 2, the surveyed processes depend on several factors:

- **The environmental conditions**: hydraulic parameters, temperature, sediments and TSS composition (OC, clay content), vegetation cover, solar radiation, pH, and redox conditions.
- **The pesticide physicochemical properties**: mobility K_{oc}, hydrophobicity K_{ow} and vapor pressure vp.

The hydraulic regime is responsible for the residence time. A long HRT gives time for the pesticide to settle, sorb, desorb, or be transformed (Liu et al., 2019; Sahin et al., 2020). Another hydraulic parameter is the water column depth; a shallow water depth enhances photolysis and biotransformation by increasing the oxygen concentration in artificial ponds.

Sediments and TSS composition interfere in the sorption process. As presented in section 2.2.1, OC content adsorption and clay content increase the trapping of pesticides in sediments by favoring desorption hysteresis. Thus, OC and clay content increase the removal of pesticides from the artificial pond water. There was no direct link between biotransformation half-lives DT_{50bio} in substrates and their OC content (Table A.11). $DT_{50bio,s}$ values show that the extent of the studied pesticides was not resistant to biotransformation in the sediment regardless of the OC content.

However, biotransformation half-lives in the water $DT50_{bio,w}$, and sediments $DT50_{bio,s}$ varied according to pesticide properties. However, no direct link was observed between half-life ($DT50_{bio}$) and pesticide hydrophobicity (Koc) (Table A.8). This observation supports our hypothesis of process synergy, and that dissipation is explained by the interaction between processes (i.e., sorption and transformation) and not a single process.

For pesticides properties, the most affecting ones are the K_{oc} and K_{ow}, since they control the distribution, mobility, and availability of pesticides in the different artificial pond compartments. We observed that K_{oc} and K_{ow} were determining factors for sorption, biotransformation, and photodegradation.

We have also noticed that biotransformation half-lives were faster in artificial ponds than half-lives in pesticide databases (i.e., PPDB). It should also be noticed that biotransformation half-lives are faster in artificial ponds sediments than in agricultural soils (Bolan et al., 1996; Passeport et al., 2011a), which support the efficiency and usefulness of artificial ponds.

Besides biotransformation, the K_{oc} and log K_{ow} also affect the sorption process. Once lowly mobile and hydrophobic pesticides are sorbed onto a substrate ($K_{oc} > 1000$ L.kg^{-1} and log $K_{ow} > 3$), they become harder to release back into the water. In comparison, more mobile pesticides ($K_{oc} < 1000$ L.kg^{-1}) are continuously transported within the artificial pond. However, it is unknown whether sorbed pesticides will be trapped in the sediment, suspended solid or vegetation, or desorbed afterward.

Numerous studies proved that vegetation is a determining agent in the fate of pesticides as it intervenes implicitly in the removal function of artificial ponds. It provides habitat to microorganisms responsible for pesticide transformation, decreases water velocity to increase settling, increases the OC content and composition diversity to enhance adsorption, and brings oxygen to sediments to promote aerobic transformation. In rare cases, vegetation was also responsible for trapping pesticides due to plant uptake. Nevertheless, it remains complex to estimate and quantify the effect of plant uptake on the fate of pesticides (Jorda et al., 2021). Another phytoremediation function of vegetation is through entophytic bacteria living in the interior tissue of plants that can potentially degrade pesticides. However, the fraction of pesticides collated in plants were often insignificant (< 10%) (Butkovskyi et al., 2021; Singh et al., 2021).

Temperature is a stimulator for transformation that increases the microbial activity behind biotransformation, solar radiation is responsible for photodegradation and enhances hydrolysis.

We suggest that process interactions affect their contribution to pesticide fate. The strong relationship between a process and pesticide properties may explain the choice of studying a specific process and pesticide. For instance, sorption is more involved in hydrophobic pesticide dissipation. At the same time, volatilization contributes more to the behavior of pesticides with a significant vapor pressure (Table A.7). On the other hand, PPDB half-life values can be used to model the behavior of some pesticides if they are close to the half-lives estimated in laboratory conditions or give satisfactory simulation results close to the observations.

3.2 Parameter estimation for modeling

After the analysis of different formulations, we suggest that processes can be quantified and hierarchized according to:

- Kinetic rates k (d^{-1}) for the transport and transfer processes: adsorption-desorption; settling-resuspension; and volatilization.
- Half-lives DT_{50} (d) for the transformation processes: biotransformation, photolysis, and hydrolysis.

The different methods used to evaluate the previously described processes and their corresponding parameters are summarized in Table 1.

Table 1

Commonly established methods in the literature to quantify and parametrize a process for a pesticide dissipation modeling approach in artificial ponds.

Process	Method	Parameter	Units
Transport			
Settling-Resuspension	Empirical formulas	v_s; v_r	$m.day^{-1}$
Transfer			
Adsorption-Desorption	Batch experiments + Kinetic model fitting	k_{ads}; k_{des}	day^{-1}
Volatilization	Empirical formulas	k_v	day^{-1}
Transformation			
Biotransformation	Incubation experiments + Kinetic model fitting	DT_{50bio}	day^{-1}
Photolysis	Lab experiments	$DT_{50photo}$	day^{-1}
Hydrolysis	Empirical formulas	DT_{50h}	day^{-1}

Most studies were conducted on a laboratory scale (batch, microcosms, and mesocosms). However, laboratory experiments are less advantageous than the rarely conducted in-situ manipulations (Fig. 1). More representative; in-situ manipulations imply a better description of the processes and the fate of pesticides while considering the interactions and links between the different processes (i.e., between sorption and transformation, sorption and dispersion/diffusion), which is not easy to bring out in laboratory conditions. In fact, in laboratory experiments, only one process is emphasized and examined independently of other processes that usually co-occur in field conditions. This often results in the overestimation of dissipation in lab-scale studies.

Additionally, not all estimated half-lives are compatible with those available in the PPDB, especially biotransformation (Table A.8). This incompatibility may be explained by the different sites where parameters were estimated, whereas, for the accessible PPDB data, there is no site specification. This observation questions the validity of parameters in the free PPDB data and their applicability in a process-based model that aims to describe what is happening in the field accurately. Using PPDB values in models increases the incertitude; therefore, we suggest defining a DT50 specific to ponds for each process to use in models. The fact that DT50 values in the PPDB are higher than those calculated in-situ shows a greater dissipation in pond artificial ponds. Because certain photodegradation half-lives in the literature present better compatibility with the PPDB values (Table A.8), they could be a good model fit.

Table 2 summarizes the studied processes for each pesticide described in this review and the equation used to describe it. We noticed a lack of studies quantifying and modeling dissipation processes in artificial ponds because studies are often limited to observing the process without a modeling approach. A considerable number of pesticides (55) were studied to evaluate their behavior according to previously described processes. However, many of these pesticides have only been evaluated once. Hence, more studies are needed on these pesticides to compare their behavior and draw more robust conclusions.

Some pesticides (i.e., boscalid, chlorpyrifos, and pretilachlor) were evaluated according to multiple processes, but not necessarily with a modeling perspective (Agudelo et al., 2010; Laabs et al., 2007; Lagunas-Allué et al., 2010; Rogers et al., 2009; Tang et al., 2019; Karpuzcu et al., 2013; Mackay et al., 2006; Yu et al., 2019). Only, 57 of the 252 reviewed articles concern modeling.

Little is known about the estimation of biotransformation half-life in artificial pond compartments, except that macrophytes and algae enhance both adsorption and biotransformation of DP (Katagi et al., 2016). Similarly, anaerobic biotransformation may occur in the sediment layer yet is still under-researched (Maillard et al., 2014; Vandermaesen et al., 2016). Globally, it remains challenging to determine a DT_{50bio} in each compartment with lab experiments due to the different behavior a pesticide may have in each of them. It is also challenging because laboratory experiments can hardly mimic reproducibly process interactions occurring in the field. For sorption, vegetation remains the least exploited compartment since there were few surveys to estimate sorption kinetics on vegetation, its biotransformation rates, or its uptake rates.

We propose that the previously mentioned controlling factors (temperature, OM, and vegetation cover) and their interactions are considered for better accuracy by integrating site-specific modulating factors in the development of pesticide fate models. Modulating factors also highlight the temporal variation of process efficiency, such as biotransformation prevailing during the plant growth period (Vandermaesen et al., 2016). So then the question is whether it is necessary to include this temporal variation in a process-based model to improve the accuracy or if, by doing so, the complexity increases.

3.3 Hierarchization of dissipation processes

This review provides a primary idea about the hierarchization of processes according to their contribution to the removal function of artificial ponds.

As adsorption can only trap pesticides in artificial ponds, degradation (i.e., complete biotransformation, photolysis, and hydrolysis) contributes more to the artificial pond removal function since it can permanently eliminate toxic substances from the environment. Although little is known about whether sorbed pesticides will be degraded or released back into the environment over time, we assume that sorption is still a key process, as it determines the distribution and availability of pesticides in artificial pond compartments.

Among the transformation processes, the most rapid was biotransformation compared to photodegradation for some pesticides. By comparing the different half-lives, volatilization was the slowest process, which is in line with the fact that most pesticides are non-volatile due to their low vapor pressure ($<10^{-5}$ Pa). Lastly, hydrolysis appears negligible of most pesticides.

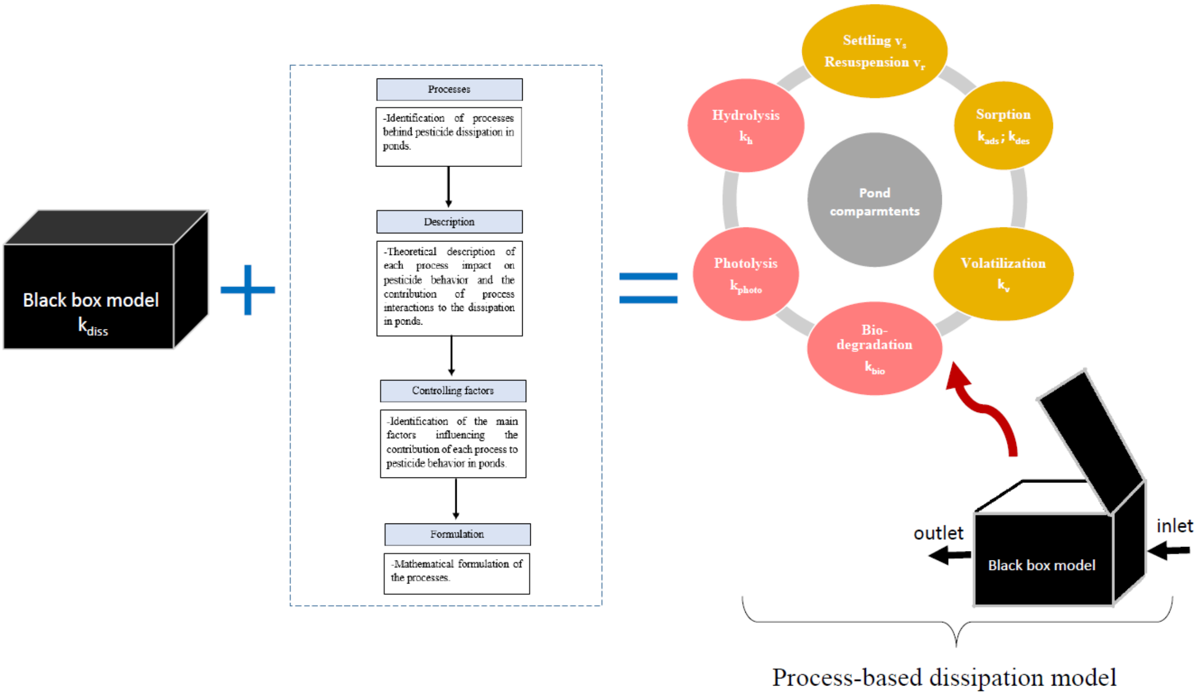
Previous reviews, such as Stehle et al. (2011) and Vymazal et al. (2015), did not find a direct link between pesticide dissipation or accumulation in artificial ponds and pesticide properties (i.e., K_{oc} vs sorption). We suggest that it is because the fate of pesticides in artificial ponds does not result from a single process but a combination of different transport, transfer, and transformation processes involving multiple artificial pond compartments. This combination depends on different conditions (i.e., hydrology, climate, physico-chemical factors, and vegetation cover). We assume that the most dependent processes are sorption and biotransformation. For some moderately mobile and hydrophilic pesticides, biotransformation can be the main dissipation process in a artificial pond. After 42 days, up to 86% of diuron (K_{oc} = 600 L.kg^{-1}), atrazine (K_{oc} = 100 L.kg^{-1}and terbuthylazine (K_{oc} = 318 L.kg^{-1}) were degraded (Anderson et al., 2002; Gregoire et al., 2009; Hijosa-Valsero et al., 2016). Also, the removal of the moderately mobile triazophos (K_{oc} = 308 L.kg^{-1}) was mostly achieved by biotransformation rather than adsorption (Cheng et al., 2007; Imfeld et al., 2009; Meng et al., 2014; Rani et al., 2001). In contrast, for the highly hydrophobic butachlor (log K_{ow} = 4.5), adsorption was the main process contributing to its dissipation in microcosm experiments. Many researchers ascertained that strong adsorption to sediments and limited desorption (also known as "aging") might decrease the bioavailability of pesticides and thus limit their biotransformation (Ahmad et al., 2004; Budd et al., 2011; Chaumet et al., 2021; Lee et al., 2004; Mulligan et al., 2016). All of these results emphasize the dependency between sorption and biotransformation.

When examining citation frequency, some processes (i.e., sorption, biotransformation) were more cited than others (i.e., settling, resuspension, and volatilization). When a process is not frequently cited, this could be explained by the difficulty level of its evaluation in both laboratory and field conditions.

The previous conclusions were based on observing removal rates of different processes and pesticides. The limitations of this approach are the relevance of the statements made based on a single case study. Nevertheless, describing one survey means that no other case study was found. The robustness of the conclusions made in this review will be stronger if the same processes are tested on multiple pesticides and sites with contrasting properties to highlight the link between the impact of processes and the properties of the pesticides and artificial ponds.

Identifying the processes with a higher impact can be used to hierarchize processes based on their contribution to the dissipation function of artificial ponds. We believe that a hierarchization of relevant processes may improve the estimation of the overall efficiency of artificial ponds in reducing pesticide transfer into the environment.

In addition, the black box model does not consider the processes mentioned above and describes pesticide behavior with an overall dissipation rate k_{diss}. Although easy to apply, the black box approach does not provide enough information on the behavior of pesticides and the contribution of each process to the dissipative function of artificial ponds. Therefore, this paper discusses the required elements to develop a simple process-based model involving the transfer, transport, and transformation processes in an interplay of artificial pond compartments to open the black box (Fig. 3). We assume that a model with multiple process rates shows the complexity of modeling the fate of pesticides in artificial ponds; however, it can describe and predict pesticide behavior more realistically. We believe this work will contribute to a better understanding of artificial pond processing and optimize pesticide fate models.



Processes

-Identification of processes behind pesticide dissipation in ponds.

Description

-Theoretical description of each process impact on pesticide behavior and the contribution of process interactions to the dissipation in ponds.

Controlling factors

-Identification of the main factors influencing the contribution of each process to pesticide behavior in ponds.

Formulation

-Mathematical formulation of the processes.

Fig. 3. Summarized elements and steps detailed in this work in order to develop a process-based model, that contrarily to black models, involves multiple processes in an interplay between different artificial pond compartments.

Table 2

List of the surveyed processes for each cited pesticide in the reviewed papers, and the equation used to estimate the rate of processes. NA means that the process was evaluated without a modeling approach.

	Processes							
Pesticides	**SET**	**RES**	**SOR**	**VOL**	**BIO**	**PHO**	**HYD**	**Source**
Acetochlor					Eq. (13)			(Droz et al., 2021)
Aclonifen					NA			(Passeport et al., 2013)
Alphamethrin				Eq. (11)				(Comoretto et al., 2008)
Atrazine			NA		NA			(Besse-Hoggan et al., 2009; Moore et al., 2013; Zeng et al., 2013)
Bifenthrin	NA							(Cryder et al., 2021)
Boscalid			NA		NA	NA	Eq. (17)	(Lagunas-Allué et al., 2010; Vallée, 2015)
Butachlor			NA		NA		NA	(Masbou et al., 2018; Torabi et al., 2020; Zheng et al., 2001)
Carbaryl							NA	(Wolfe et al., 1978)
Chlorothalonil					NA			(Passeport et al., 2013)
Chlorotoluron					NA			(Passeport et al., 2013)
Chlorpyrifos			NA	NA	Eq. (13)		NA	(Agudelo et al., 2010; Laabs et al., 2007; Sardar et al., 2005 ; Rogers et al., 2009; Tang et al., 2019; Wang et al., 2019 ; Karpuzcu et al., 2013; Liu et al., 2019; Mackay et al., 2006; Yu et al., 2019)
Clothianidin					Eq. (13)			(Beringer et al., 2021)
Cyfluthrin	NA							(Cryder et al., 2021)
Cyproconazole					NA			(Passeport et al., 2013)
Diazinon			NA		Eq. (13)		Eq. (17)	(Ibáñez et al., 2006; Mackay et al., 2006; Sahin et al., 2020; Sakellarides et al., 2003)
Dichlorvos					NA		Eq. (17)	(Sahin et al., 2020; Tomlin, 2009)
Dieldrin				Eq. (11)			Eq. (17)	(Singh et al., 1991)
Diuron					NA			(Zeng et al., 2013)
Endosulfan				Eq. (11)		Eq. (15)	Eq. (17)	(Rose et al., 2008; Singh et al., 1991; Walse et al., 2002)
Epoxiconazole			NA		NA			(Passeport et al., 2011; Passeport et al., 2013)
Ethofumesate					NA			(Passeport et al., 2013)
Fenitrothion						Eq. (15)		(Sakellarides et al., 2003)
Fenpropidine					Eq. (13)			(Passeport et al., 2013)
Fenthion					NA	Eq. (15)	NA	(Sahin et al., 2020; Sakellarides et al., 2003)
Fipronil	NA							(Cryder et al., 2021)
Fludioxonil			NA			Eq. (15)		(Katagi et al., 2016; Maillard et al., 2014)
Fluometuron			NA			Eq. (15)	Eq. (17)	(Hand et al., 2001; Rose et al., 2008)
Glyphosate			NA					(Maillard et al., 2014)
Imazosulfuron			Eq. (8)					(Takagi et al., 2012)
Imidacloprid			NA					(Mahabali et al., 2014)
Isoproturon			NA		NA			(Passeport et al., 2011; Passeport et al., 2013; Vallée, 2015)
Kresoxim-methyl				Eq. (11)				(Gobas et al., 2018)
Lindane			Eq. (8)					(Krishna et al., 2008)
MCPA			NA		Eq. (13)			(Comoretto et al., 2008; Sørensen et al., 2006; Thorstensen et al., 2001)
Mefenacet			Eq. (8)	Eq. (11)				(Bolan et al., 1996; Watanabe et al., 2006; Yoshida et al., 2000)
Mefenpyr-diethyl				NA				(Passeport et al., 2013)
Metazachlor			NA		Eq. (13)			(Passeport et al., 2011; Passeport et al., 2013)

Table 2 *(continued)*

	Processes							
Pesticides	**SET**	**RES**	**SOR**	**VOL**	**BIO**	**PHO**	**HYD**	**Source**
Methyl parathion		NA	NA					(Rong et al., 2019; Schulz et al., 2003)
Molinate			Eq. (8)	Eq. (11)		Eq. (15)		(Inao et al., 1999)
Napropamide			NA		NA			(Passeport et al., 2013; Vallée, 2015)
Oxadiazon				Eq. (11)				(Comoretto et al., 2008)
Parathion-Methyl						Eq. (15)		(Sakellarides et al., 2003)
Picloram			Eq. (8)					(McCall et al., 1985)
Pretilachlor			Eq. (8)	Eq. (11)	Eq. (13)	Eq. (15)		(Vallée, 2015)
Prochloraz			NA		NA			(Passeport et al., 2013)
Prosulfocarb					NA			(Gobas et al., 2018)
Pyraclostrobin				Eq. (11)				(Maillard et al., 2014)
Pyrimethanil			NA					(Droz et al., 2021; Masbou et al., 2018; Passeport et al., 2013)
S-Metolachlor					Eq. (13)	Eq. (15)	NA	(Comoretto et al., 2008; Nakano et al., 2004; Watanabe et al., 2006; Yoshida et al., 2000)
Spiroxamine			NA					(Maillard et al., 2014)
Tebuconazole			NA		NA			(Maillard et al., 2014; Passeport et al., 2011; Passeport et al., 2013; Vallée, 2015)
Terbutylazine					NA			(Hijosa-Valsero et al., 2016)
Triazophos					NA			(Cheng et al., 2007)
Trifluralin				NA				(Laabs et al., 2007)

4. Conclusion

The scientific community has widely proved the role and efficiency of artificial ponds in reducing pesticide transfer into the environment. However, few papers have identified the set of processes behind the behavior of pesticides in artificial ponds and opened the black box. The relationship between the dissipation potential of processes and the properties of artificial ponds and pesticides has yet to receive little attention.

When reviewing the literature, we assumed that pesticide behavior is due to multiple processes acting to degrade or trap them in artificial ponds. From this, we can conclude that the fate of pesticides results from the interaction of different processes. These processes depend on several hydraulic, climatic, and physicochemical factors and vegetation cover. Since previous studies have generally focused on a single pesticide or dissipation process, this review combines research work on pesticides with contrasting properties. This extends to pesticide behavior, reflecting several processes involved in the dissipation of pesticides in a large variety of artificial ponds. We also describe and compare the methods used to quantify processes behind pesticide dissipation or accumulation in artificial ponds.

Based on the bibliometric analysis, we explain the citation frequency of a process by the difficulties of evaluating its contribution in the field or the laboratory. Moreover, investigating pesticide behavior in an interplay between transfer, transport, and transformation processes can be used to hierarchize processes based on their contribution to the dissipation function of pond artificial ponds. We suggest that this hierarchization may also improve the estimation of the overall efficiency of pond artificial ponds in pesticide dissipation.

This work is a detailed analysis of the literature to improve our understanding of the potential of artificial ponds to dissipate incoming pesticide loads and thus lead to better management and, therefore, water quality improvement. This work paves the way for the scientific community to develop generic process-based models that, contrary to black-box models, will predict pesticide behavior at the artificial pond scale while integrating the key dissipation processes.

Author Contributions:

Conceptualization: A.B, J.T, S.S, S.P, G.I.; Writing - Original Draft: A.B; Writing - Review & Editing: J.T, S.S, S.P, G.I, J.P, B.C; Supervision: J.T, S.S; Project administration: G.I; Funding acquisition: G.I, J.T, S.S, S.P, J.P.

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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.

Appendix A.

Table A.1

Formulation of settling rates of suspended particles in wetlands and associated parameters.

Settling rate v_s (m.s^{-1})	**Parameters**	**Conditions of application**	**Source**
$\frac{2}{9} * \frac{r_p^2 * (\rho_p - \rho_w) * g}{\mu} \quad (i)$	• v_s (m.s^{-1}): Settling rate. • r_p (m): Average radius of the particle. • ρ_p (kg.m^{-3}) : Particles density. • ρ_w (kg.m^{-3}) : Water density. • g (m.day^{-2}) : Gravity. • µ (kg.m^{-1}.day^{-1}) : Water viscosity.	Reynolds number < 500	(Fiener et al., 2003; Marois et al., 2016; Reddy et al., 2008)
$(\frac{4}{3} * \frac{g * d}{C_D} * \frac{(\rho_s - \rho_w)}{\rho_w})^{\frac{1}{2}} \quad (ii)$	• v_s (m.s^{-1}): Settling rate. • d (m): Particles diameter. • ρ_s (kg.m^{-3}) : Solid density. • ρ_w (kg.m^{-3}) : Water density. • g (m.j^{-2}) : Gravity. • C_D = f (Rep) , Rep : Reynolds number of the particle.	Spherical particles in water at 20°C	(Kadlec et al., 2008)
$\mu * d^{-1} * ((25 + 1{,}2 * d^{*2})^{0{,}5} - 5))^{1.5} \quad (iii)$ $d^* = (d * (g * \mu^{-2} * (s - 1))^{1/3}$	• v_s (m.s^{-1}): Settling rate. • d (m): Particles diameter. • g (m.j^{-2}) : Gravity. • µ (kg.m^{-1}.day^{-1}) : Water viscosity. • s : Relative density= f(ρ_s, $\rho_{w)}$	Natural sand particles	(Hawley et al., 2014)

Table A.2

Integrated sorption kinetic and isotherm models. Kinetic models describe the temporal evolution of the sorbed amount of a dissolved substance. Kinetic rates are estimated from fitting laboratory measurements to the kinetic models. Isotherm models describe adsorption and desorption equilibrium. Likewise, isotherm constants are calculated from fitting laboratory measurements to isotherm curves.

Kinetic models

Equations	Parameters	Source
Pseudo first order (PFO): $C_s = C_{se} - e^{-k_1 * t}$	• C_s (µg.g^{-1}): Sorbate concentration in the substrate at instant t. • C_{se} (µg.g^{-1}): Sorbate concentration in the substrate at the equilibrium. • k_1 (min^{-1}): Kinetic constant.	(Lin et al., 2009)
Pseudo second order (PSO): $C_s = \frac{k_2 * {C_{se}}^2 * t}{1 + k_2 * C_{se} * t}$	• C_s (µg.g^{-1}): Sorbate concentration in the substrate at instant t. • C_{se} (µg.g^{-1}): Sorbate concentration in the substrate at the equilibrium. • k_2 (g.mg^{-1}.min^{-1}): Kinetic constant rate.	(Boparai et al., 2011)
Elovich: $C_s = \frac{1}{\beta} * \ln(1 + \alpha * \beta * t)$	• C_s (µg.g^{-1}): Sorbate concentration in the substrate at instant t. • α (g.mg^{-1}.min^{-1}) , β (g.mg^{-1}): Kinetic constants.	(Chien et al., 1980; Ho et al., 2002)

Isotherm models

Equations	Parameters	Source
Freundlich: $C_{se} = K_f * {C_{we}}^{n_f}$	• C_{se} (µg.g^{-1}): Sorbate concentration in the substrate at the equilibrium. • C_{we} (µg.g^{-1}): Sorbate concentration in the water at the equilibrium. • K_f ($mg^{1-n}{}_{f}.L^{n}{}_{f}.kg^{-1}$): Freundlich constant expressing the molecule-substrate affinity. During adsorption K_f is called K_{fads} and K_{fdes} during desorption. • n_f (unitless): Freundlich constant. When n_f = 1, the kinetic model becomes linear and K_f becomes the classical distribution coefficient K_d.	(Mckay, 1995)
Langmuir: $C_{se} = \frac{K_L * C_{we}}{1 + n_L * C_{we}}$	• C_{se} (µg.g^{-1}): Sorbate concentration in the substrate at the equilibrium. • C_{we} (µg.g^{-1}): Sorbate concentration in the water at the equilibrium. • K_L (L.g^{-1}), n_L (unitless): Langmuir constants.	(Mckay, 1995)

Table A.3

Non-linear Freundlich isotherm adsorption K_{fads} (mg^{1-n} kg^{-1} L^{n}) and desorption K_{fdes} (mg^{1-n} kg^{-1} L^{n}) coefficients. Linear isotherm coefficient K_d ($L.kg^{-1}$) for metazachlor, isoproturon, epoxiconazole, napropamide, boscalid, prochloraz, tebuconazole, glyphosate, pyrimethanil, tetraconazole, fludioxonil and spiroxamine. The sorption coefficients were measured in laboratory conditions with batch experiments on different pond substrates: Sediments, vegetation, and total suspended solids (TSS) (Maillard et al., 2014; Passeport et al., 2011b; Vallée, 2015).

Pesticide	K_d/K_f*	Substrates	K_d	K_{fads}	K_{fdes}	Source
	K_d = 0.78	Sediments	3.65	4.5	0.11	
Metazachlor	K_f = -	Vegetation	53	60	3,35	
	K_d= -	Sediments	5.85	3.5	0.1	
Isoproturon	K_f = 2.83	Vegetation	34	44	26.5	
	K_d = -	Sediments	-	281.5	0.2	
Epoxiconazole	K_f = 12.2	Vegetation	546	902	2.34	(Passeport et al., 2011)
	K_d = 21.5	Sediments		5	4.8	
Napropamide	K_f = 21,6	Vegetation		77.4	75.2	
	K_d= -	Sediments		9.05	8.9	
Boscalid	K_f = 12.6	Vegetation	-	191.8	129.5	
	K_d= -	Sediments		63.3	10.3	
Prochloraz	K_f = 38	Vegetation		442.2	110	
	K_d= -	Sediments		12.3	9	
Tebuconazole	K_f = 12.69	Vegetation		127.8	98.1	(Vallée, 2015)
	K_d= 209.4		Inlet: 87.06			
Glyphosate	K_f = 226.3	TSS	Outlet: 18067			
	K_d= -		Inlet: 833.3			
Pyrimethanil	K_f = 8.38	TSS	Outlet: 414			
	K_d= -		Inlet: 524.8	-		
Tetraconazole	K_f = 145	TSS	Outlet: 1533			
	K_d= -		Inlet: 944.4			
Fludioxonil	K_f = 3312	TSS	Outlet: 977			
	K_d= -		Inlet: 1461			
Spiroxamine	K_f = 142.3	TSS	Outlet: 1681			(Maillard et al., 2014)

(*): Sorption coefficients (K_d, K_f) were extracted from the pesticide database PPDB (Lewis et al., 2016).

(-): Not measured.

Table A.4

Adsorption and desorption kinetics (k_{ads}, k_{des}) (d^{-1}) of five selected pesticides (imazosulfuron, lindane, mefenacet, molinate and picloram) in different ponds, along with their physicochemical properties S ($mg.L^{-1}$) and K_{oc}($L.kg^{-1}$) (Inao et al., 1999; Krishna et al., 2008; Mccall et al., 1985; Takagi et al., 2012; Yoshida et al., 2000). Pesticide properties were extracted from pesticide databases (ChemSpider; Kim et al., 2021; Lewis et al., 2016).

Pesticide	Physicochemical properties	Adsorption k_{ads}(d^{-1})	Desorption k_{des}(d^{-1})	Site	Source
Imazosulfuron	S (mg/l) = 1400 K_{oc}(L/kg) = 754	$1.64*10^{-1}$	$7.59*10^{-3}$	Paddy field OC = 1% OM = 1.83%	(Takagi et al., 2012)
Lindane	S (mg/l) = 8.5 K_{oc}(L/kg) = 1081	$1.7*10^{-4}$	$8.54*10^{-6}$	Agricultural pond OC=2% Clay=56%	(Krishna et al., 2008)
Mefenacet	S (mg/l) = 4 K_{oc}(L/kg) = 2964	1.2	$2*10^{-1}$	Paddy field OC = 16% OM = 27%	(Yoshida et al., 2000)
Molinate	S (mg/l) = 1100 K_{oc}(L/kg) = 190	$4.87*10^{-1}$	$2.9*10^{-1}$	Paddy field OC = 2%, OM=3.44% Clay = 47%	(Inao et al., 1999)
Picloram	S (mg/l) = 430 K_{oc}(L/kg) = 100	$1.78*10^{-1}$	3.7	Agricultural pond OC= 3%	(McCall et al., 1985)
Pretilachlor	S (mg/l) = 50 K_{oc}(L/kg) = 1159	$1.4*10^{-5}$	$2.3*10^{-6}$	Paddy field OC = 16% OM = 27%	(Yoshida et al., 2000)

Table A.5

Volatilization classes according to the vapor pressure values (Höllrigl-Rosta, 2017).

Vapor pressure (Pa) at 20°C	Volatilization class
$< 10^{-5}$	Non-volatile
$< 10^{-4}$	Non-volatile from soil surfaces, no volatilization from greenhouses
$10^{-5} < \_\_\_ < 10^{-4}$	Semi volatile
$10^{-4} < \_\_\_ < 5.10^{-3}$	Semi volatile
$\geq 10^{-3}$	Volatile

Table A.6

Empirical formulations of volatilization rate $k_v(d^{-1})$ along with their corresponding parameters, based on a 1^{st} order kinetic volatilization model.

Volatilization rate k_v	Parameters	Source
$\frac{1}{h_w} * \frac{\frac{H}{RT}}{\frac{1}{720 * \sqrt{\frac{18}{MW}}} + \frac{1}{4.8 * \sqrt{\frac{44}{MW}}}}$	• H ($Pa.m^3.mol^{-1}$): Henry coefficient at 25°C. • R ($Pa.m^3.mol^{-1}\ K^{-1}$): Universal gas constant. • T (K): Absolute temperature. • MW ($g.mol^{-1}$): Molecular Weight of the chemical compound. • hw (m): Water depth.	(Jacobs et al., 2012)
$\frac{1}{\frac{1}{0,0036 + 0,01 * u^{1,2}} + \frac{8.3T}{H * (3,6 + 5 * u^{1,2})}}$	• u ($m.s^{-1}$): Wind speed. • H ($Pa\ m^3\ g^{-1}$): Henry constant. • T(°C) : Temperature.	(Rose et al., 2008)

Table A.7

Review of the volatilization half-lives DT_{50v} (d) and the corresponding volatilization rates $k_v(d^{-1})$ in ponds, of different pesticides along with their corresponding vapor pressure vp (Pa) and their volatilization classification according to the table A.5.

Pesticide	vp (Pa)*	Classification	DT_{50v} (d)	k_v (d^{-1})	Source
Alphamethrin	$2.3*10^{-3}$	Volatile	0.2	3.01	
Oxadiazon	$1.49*10^{-5}$	Semi volatile	24.8	0.03	(Comoretto et al., 2008)
Kresoxim-methyl	$2.29*10^{-6}$	Non-volatile	96924	0.00	
Pyraclostrobin	$2.60*10^{-8}$	Non-volatile	6850343	0.00	(Gobas et al., 2018)
Molinate	1.3	Volatile	2.6	0.27	(Inao et al., 1999)
Endosulfan	$7.59*10^{-1}$	Volatile	3	0.23	
Dieldrin	$3.20*10^{-3}$	Volatile	4.4	0.16	(Singh et al., 1991)
Pretilachlor	$1.2*10^{-4}$	Semi volatile	346.5	0.00	
Mefenacet	$6.39*10^{-7}$	Non-volatile	5941	0.00	(Watanabe et al., 2006)

*: vp (Pa) is the vapor pressure of the pesticide at 20°C. The values were extracted from pesticides properties databases PPDB, PubChem and ChemSpider.

Table A.8

Biotransformation half-lives in the water column $DT_{50bio,w}$ (d) and sediment surface $DT_{50bio,s}$ (d) of nine pesticides in different wetlands, along with their mobility K_{oc}(L.kg^{-1}) (Beringer et al., 2021; Bondarenko et al., 2004; Comoretto et al., 2008; Droz et al., 2021; Nakano et al., 2004; Passeport et al., 2011a; Passeport et al., 2013; Sahin et al., 2020). These half-lives were confronted to the PPDB values.

Pesticide	**K_{oc} $^+$ (L/kg)**	**$DT_{50bio,w}$ (d)**	**$DT_{50bio,s}$ (d)**	**Site**	**$DT_{50bio,w}$ (d) PPDB**	**$DT_{50bio,s}$ (d) PPDB**	**Source**
Acetochlor	156	Anaerobic: 59 Aerobic: 31	Anaerobic: 47 Aerobic: 53	Wetland OC=2.3%	40.5	19.7	(Droz et al., 2021)
Chlorpyrifos	5509	-	Anaerobic: 125-746 Aerobic: 1.8 - 4.9	Urban stream sediment OC=1.8%	-	36.5	(Bondarenko et al., 2004)
Clothianidin	123	-	Anaerobic: 27.2 Aerobic: 115	Wetland OC=0.2%	-	-	(Beringer et al., 2020)
Diazinon	165-189	-	Aerobic: 53.32	Lake sediment OC= 2.22%	4.3	10.4	(Sahin et al., 2020)
Fenthion	1500	-	Aerobic: 5.5		-	92	
MCPA	100	-	5.1*	Paddy field OC=0.9%, Clay=47%	-	25	(Comoretto et al., 2008)
Metazachlor	134	216*	20.6*	Wetland OC = -	-	-	(Passeport et al., 2013)
Pretilachlor	1159	9.8*	19.0*	Paddy field OC=2.3%	-	-	(Nakano et al., 2004)
S-Metolachlor	300	Anaerobic: 199 Aerobic: 29	Anaerobic: 182 Aerobic: 151	Wetland OC=2.3%	88	365	(Droz et al., 2021)
Epoxiconazole	300		170	Wetland OC=2%	1000	103.6	(Passeport et al., 2011)

*: It was not specified under which conditions (aerobic or anaerobic) biotransformation half-lives were measured.

+: The K_{oc} values were extracted from pesticides databases (ChemSpider; Kim et al., 2021; Lewis et al., 2016; Zambonin et al., 2002).

- : Not available.

Table A.9

Photodegradation half-lives in the water column $DT_{50photo}$ (d) of eight selected pesticides with different mobilities K_{oc} (L.kg^{-1}). The DT_{50} values were estimated in (Inao et al., 1999; Katagi et al., 2016; Rose et al., 2008; Sakellarides et al., 2003).

Pesticide	**K_{oc}* (L/kg)**	**$DT_{50photo}$ (d)**	**PPDB**	**Source**
Atrazine	100	385	2.6	(Mathon et al. 2019)
Dimethoate	5.2-50	495	175	(Mathon et al. 2019)
Diuron	680	93.7	43	(Mathon et al. 2019)
Endosulfan	11500	12	-	(Rose et al., 2008)
Fenitrothion	2000	0.5	3.5	(Sakellarides et al., 2003)
Fenthion	1500	1.6	0.4	(Sakellarides et al., 2003)
Fludioxonil	145 600	9	10	(Katagi et al., 2016)
Fluometuron	202	1	stable	(Rose et al., 2008)
Isoproturon	126	41.3	48	(Mathon et al. 2019)
Molinate	190	54	stable	(Inao et al., 1999)
Parathion-Methyl	1500	25	9	(Sakellarides et al., 2003)
Simazine	130	173	1.9	(Mathon et al. 2019)

*: The K_{oc} values were extracted from pesticides databases (ChemSpider; Kim et al., 2021; Lewis et al., 2016; Zambonin et al., 2002).

Table A.10

Modulating factors applied to the photolysis rate k_{photo} to account for controlling factors (i.e., Temperature, depth, and radiation).

Modulating factors of k_{photo}	**Parameters**	**Source**
$\frac{1}{h_w} * \frac{I_0}{I_{max}} * \frac{1}{25 * TSS * e^{-25 * TSS * z}}$	• z (m): Depth. • I_0 (MJ m^{-2} h^{-1}): Actual radiation. • I_{max} (MJ m^{-2} h^{-1}) : Maximum radiation. • TSS (mg.g^{-1}): Amount of TSS.	(Rose et al., 2008)
R_{UVB-b}	• $R_{UVB-b:}$ Daily UV-B radiation that penetrates the pond.	(Watanabe et al., 2006)

Table A.11

Biotransformation half-lives $DT_{50bio,s}$ (d) of five pesticides with different mobilities K_{oc} and hydrophilicities log K_{ow}, measured in laboratory studies on sediment substrates with different organic carbon content OC (%) (Vallee et al., 2016). Along with, biotransformation half-lives at the water-sediment interface in wetlands, extracted from the PPDB.

Pesticides	**K_{oc} (L/kg)***	**log (K_{ow})***	**OC (%)**	**$DT_{50bio,s}$ (d)**	**$DT_{50bio,s}$ (d) PPDB**
Boscalid	9500	2.96	1.23	39.4	
			1.5	29.8	545
			1.7	9.4	
Isoproturon	122	2.5	1.2	100	
			1.5	61	149
			1.7	63.9	
Napropamide	885	3.3	1.23	82.7	
			1.5	80.1	316
			1.7	54	
Prochloraz	378	4.6	1.2	100	
			1.5	17.2	359
			1.7	10.2	
Tebuconazole	769	3.7	1.2	30.5	
			1.5	30.6	365
			1.7	1.4	

(*): The physicochemical properties K_{oc} and log (K_{ow}) were extracted from the Pesticide Properties DataBase PPDB and PubChem.

Appendix B.

Methodology

In this paragraph, we will present the methodology adopted to realize this critical review of process formulations and controlling factors behind pesticide dissipation in ponds. This work started with a bibliometric research. We used the listed research equations in different bibliographic research tools*:

- (constructed wetland) AND ((wastewater OR agricultural) AND treatment)

- pesticide AND (dissipation OR mitigation) AND (wetland OR pond)

- (pesticide OR "Organic pollutant") AND wetland AND model

- (pesticide OR "Organic pollutant") AND wetland AND fate

- pesticide AND *process* AND (pond OR wetland)

- pesticide AND transfer AND (wetland OR pond)

- *process* AND (rate OR model OR kinetic) AND pesticide AND pond

*: Web of Science (1), Scopus (2), Google Scholar (3), EndNote (4), and Science direct (5).

(1):https://www.webofscience.com/wos/woscc/basic-search

(2):https://www.scopus.com/search/form.uri?display=basic#basic

(3): https://scholar.google.com/schhp?hl=fr

(4): https://www.myendnoteweb.com/EndNoteWeb.html

Italic represents the name of the processes (i.e., sorption/adsorption/desorption, biodegradation/degradation, settling/sedimentation/burial, resuspension, photolysis/photodegradation, volatilization, and hydrolysis).

First, we looked for general studies about the fate of pesticides in wetlands to record every potential process behind their dissipation. Afterward, we searched for each process whose effect was investigated in ponds on single or multiple pesticides. We selected theoretical papers for the descriptive part of processes and modeling papers to extract the equations for the formulation part in section 3. In each paper, we identified the pesticides targeted, processes investigated, site, and the scale of their evaluation (i.e., laboratory, in-situ) by focusing more on recent work. Given this, three categories of processes in ponds can be distinguished:

1. Transport is responsible for the fate of pesticides within a compartment of the pond.
2. Transfer controls redistribution of pesticides from a compartment to another.
3. Transformation degrades and modifies the original chemical structure of pesticides.

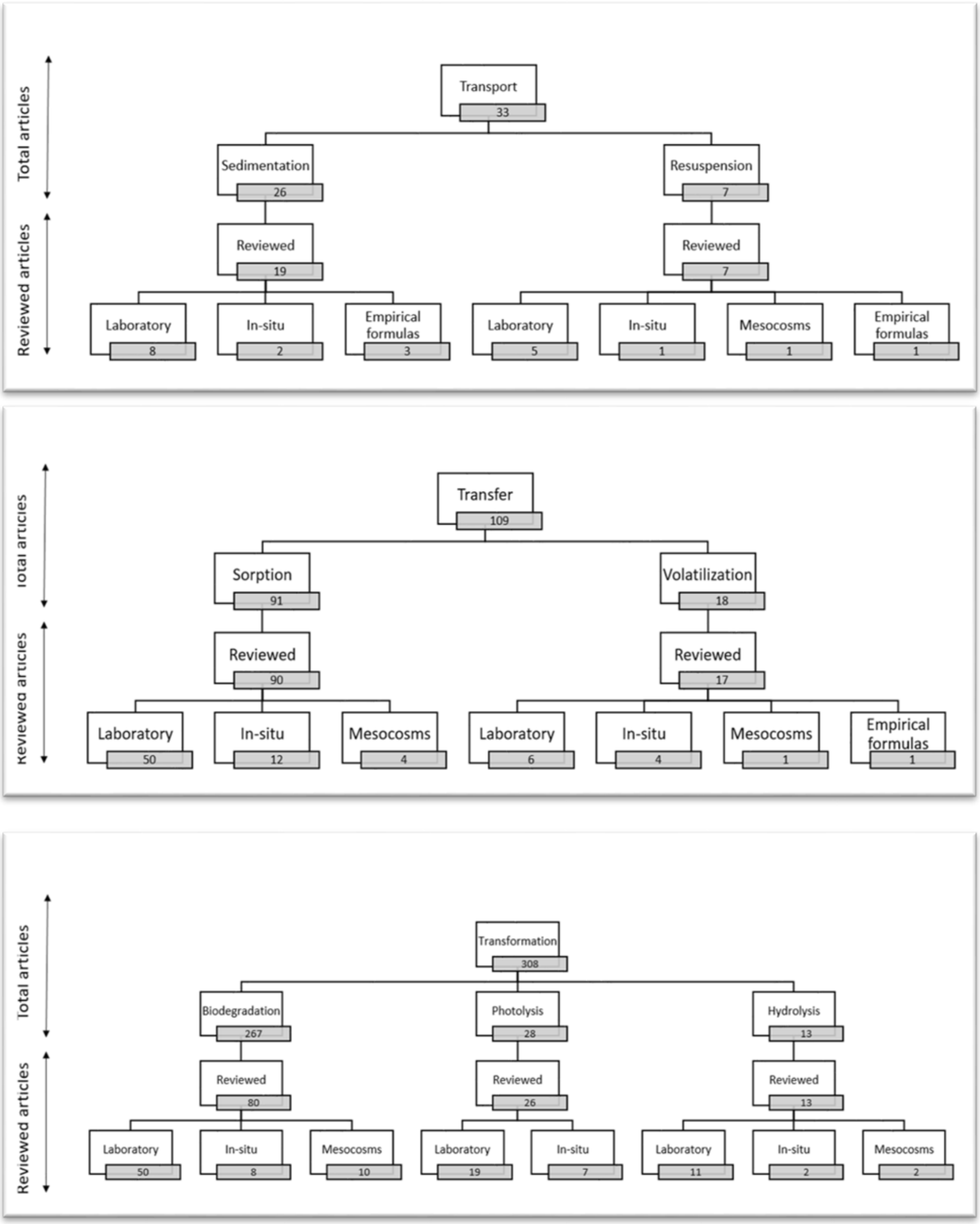

Fig A. 1. Diagram showing the number of searched articles according to the type of processes, and the number of papers we reviewed. We reviewed papers with the most information about pesticides and processes in ponds. We did not analyze papers with little information, out of context, or redundant. The bottom of the diagram refers to the enumeration of papers according to study scale.

Among the set of processes, according to transformation processes, biodegradation was the most one surveyed with a total of 267 papers, followed by 91 studies of sorption (Fig. 1). Photolysis and settling were less evaluated; only 28 and 26 articles cited them, respectively. Finally, resuspension and volatilization had a minor presence in search results. As for processes formulations, it was necessary to look for modeling papers that either presented a process equation within the framework of a fate model or studied just that specific process. Among the 273 reviewed papers, only 20% were about descriptive models of pesticides in ponds, and over 80% evaluated more than one pesticide. Although, most of the studies investigated a single process for a group of 2-6 pesticides. Most studies aiming to quantify a process rate were conducted at a laboratory scale in controlled conditions. They were few studies conducted in mesocosms* and even less in the field.

*: Mesocosms are experimental water enclosures designed to provide a limited body of water with close to natural conditions, in which environmental factors can be realistically manipulated (Berger et al., 2016).

References

Agudelo, C. R., Jaramillo, M. L., & Penuela, G. (2012). Comparison of the removal of chlorpyrifos and dissolved organic carbon in horizontal sub-surface and surface flow wetlands. *Science of the Total Environment, 431*, 271-277. doi:10.1016/j.scitotenv.2012.05.045

Agudelo, R. M., Penuela, G., Aguirre, N. J., Morato, J., & Jaramillo, M. L. (2010). Simultaneous removal of chlorpyrifos and dissolved organic carbon using horizontal sub-surface flow pilot wetlands. *Ecological Engineering, 36*(10), 1401-1408. doi:10.1016/j.ecoleng.2010.06.019

Ahmad, R., Kookana, R. S., Megharaj, M., & Alston, A. M. (2004). Aging reduces the bioavailability of even a weakly sorbed pesticide (carbaryl) in soil. *Environ Toxicol Chem, 23*(9), 2084-2089. doi:10.1897/03-569

Aktar, M. W., Sengupta, D., & Chowdhury, A. (2009). Impact of pesticides use in agriculture: their benefits and hazards. *Interdisciplinary toxicology, 2*(1), 1-12. doi:10.2478/v10102-009-0001-7

Anderson, K. L., Wheeler, K. A., Robinson, J. B., & Tuovinen, O. H. (2002). Atrazine mineralization potential in two wetlands. *Water Res, 36*(19), 4785-4794. doi:10.1016/s0043-1354(02)00209-9

Arcangeli, J. P., & Arvin, E. (1997). Modeling of the cometabolic biodegradation of trichloroethylene by toluene oxidizing bacteria in a biofilm system. *Environmental Science & Technology, 31*(11), 3044-3052. doi:DOI 10.1021/es9609112

Azizian, S. (2004). Kinetic models of sorption: a theoretical analysis. *J Colloid Interface Sci, 276*(1), 47-52. doi:10.1016/j.jcis.2004.03.048

Azizian, S., Eris, S., & Wilson, L. D. (2018). Re-evaluation of the century-old Langmuir isotherm for modeling adsorption phenomena in solution. *Chemical Physics, 513*, 99-104. doi:10.1016/j.chemphys.2018.06.022

Barchanska, H., Plonka, J., Jaros, A., & Ostrowska, A. (2019). Potential application of Pistia stratiotes for the phytoremediation of mesotrione and its degradation products from water. *Int J Phytoremediation, 21*(11), 1090-1097. doi:10.1080/15226514.2019.1606780

Baskaran, S., Kookana, R. S., & Naidu, R. (2003). Contrasting behaviour of chlorpyrifos and its primary metabolite, TCP (3,5,6-trichloro-2-pyridinol), with depth in soil profiles. *Australian Journal of Soil Research, 41*(4), 749-760. doi:10.1071/Sr02062

Beecraft, L., & Rooney, R. (2021). Bioconcentration of glyphosate in wetland biofilms. *Science of the Total Environment, 756*, 143993. doi:10.1016/j.scitotenv.2020.143993

Berger, S. A., & Nejstgaard, J. C. (2016). A Worldwide Web-portal for Aquatic Mesocosm Facilities: WWW. MESOCOSM. EU. *American Geophysical Union, 2016*, IS34A-2323.

Beringer, C. J., Goyne, K. W., Lerch, R. N., Webb, E. B., & Mengel, D. (2021). Clothianidin decomposition in Missouri wetland soils. *J Environ Qual, 50*(1), 241-251. doi:10.1002/jeq2.20175

Besse-Hoggan, P., Alekseeva, T., Sancelme, M., Delort, A. M., & Forano, C. (2009). Atrazine biodegradation modulated by clays and clay/humic acid complexes. *Environ Pollut, 157*(10), 2837-2844. doi:10.1016/j.envpol.2009.04.005

Beulke, S., Brown, C. D., Fryer, C. J., & van Beinum, W. (2004). Influence of kinetic sorption and diffusion on pesticide movement through aggregated soils. *Chemosphere, 57*(6), 481-490. doi:10.1016/j.chemosphere.2004.06.026

Bezbaruah, A. N., & Zhang, T. C. (2005). Quantification of oxygen release by bulrush (Scirpus validus) roots in a constructed treatment wetland. *Biotechnol Bioeng, 89*(3), 308-318. doi:10.1002/bit.20332

Blachnio, M., Derylo-Marczewska, A., & Seczkowska, M. (2020). Influence of Pesticide Properties on Adsorption Capacity and Rate on Activated Carbon from Aqueous Solution. doi:10.5772/intechopen.88726

Bladel, R. V., & Laudelout, H. (1967). APPARENT IRREVERSIBILITY OF ION-EXCHANGE REACTIONS IN CLAY SUSPENSIONS *Soil Science* (Vol. 104).

Bloomfield, J. P., Williams, R. J., Gooddy, D. C., Cape, J. N., & Guha, P. (2006). Impacts of climate change on the fate and behaviour of pesticides in surface and groundwater--A UK perspective. *Science of the Total Environment, 369*(1-3), 163-177. doi:10.1016/j.scitotenv.2006.05.019

Bojcevska, H., & Tonderski, K. (2007). Impact of loads, season, and plant species on the performance of a tropical constructed wetland polishing effluent from sugar factory stabilization ponds. *Ecological Engineering, 29*(1), 66-76. doi:10.1016/j.ecoleng.2006.07.015

Bolan, N. S., & Baskaran, S. (1996). Biodegradation of 2,4-D herbicide as affected by its adsorption-desorption behaviour and microbial activity of soils. *Australian Journal of Soil Research, 34*(6), 1041-1053. doi:Doi 10.1071/Sr9961041

Bondarenko, S., Gan, J., Haver, D. L., & Kabashima, J. N. (2004). Persistence of selected organophosphate and carbamate insecticides in waters from a coastal watershed. *Environ Toxicol Chem, 23*(11), 2649-2654. doi:10.1897/03-436

Boparai, H. K., Joseph, M., & O'Carroll, D. M. (2011). Kinetics and thermodynamics of cadmium ion removal by adsorption onto nano zerovalent iron particles. *J Hazard Mater, 186*(1), 458-465. doi:10.1016/j.jhazmat.2010.11.029

Bowman, B., & Sans, W. (1985). *Partitioning Behavior of Insecticides in Soil-Water Systems: I. Adsorbent Concentration Effects* (0047-2425). Retrieved from

Braschi, I., Blasioli, S., Lavrnic, S., Buscaroli, E., Di Prodi, K., Solimando, D., & Toscano, A. (2022). Removal and fate of pesticides in a farm constructed wetland for agricultural drainage water treatment under Mediterranean conditions (Italy). *Environ Sci Pollut Res Int, 29*(5), 7283-7299. doi:10.1007/s11356-021-16033-4

Braskerud, B. C. (2001). The influence of vegetation on sedimentation and resuspension of soil particles in small constructed wetlands. *J Environ Qual, 30*(4), 1447-1457. doi:10.2134/jeq2001.3041447x

Braskerud, B. C. (2002). Design considerations for increased sedimentation in small wetlands treating agricultural runoff. *Water Science and Technology, 45*(9), 77-85. doi:10.2166/wst.2002.0209

Brix, H. (2003). *Plants used in constructed wetlands and their functions.* Paper presented at the 1st International Seminar on the use of Aquatic Macrophytes for Wastewater Treatment in Constructed Wetlands, edit. Dias V., Vymazal J. Lisboa, Portugal.

Brueske, C. C., & Barrett, G. W. (1994). Effects of Vegetation and Hydrologic Load on Sedimentation Patterns in Experimental Wetland Ecosystems. *Ecological Engineering, 3*(4), 429-447. doi:Doi 10.1016/0925-8574(94)00011-5

Brusseau, M. L., & Chorover, J. (2019). Chemical Processes Affecting Contaminant Transport and Fate. 113-130. doi:10.1016/b978-0-12-814719-1.00008-2

Budd, R., O'Geen, A., Goh, K. S., Bondarenko, S., & Gan, J. (2009). Efficacy of constructed wetlands in pesticide removal from tailwaters in the Central Valley, California. *Environ Sci Technol, 43*(8), 2925-2930. doi:10.1021/es802958q

Budd, R., O'Geen, A., Goh, K. S., Bondarenko, S., & Gan, J. (2011). Removal mechanisms and fate of insecticides in constructed wetlands. *Chemosphere, 83*(11), 1581-1587. doi:10.1016/j.chemosphere.2011.01.012

Bunyakan, C., Seaoung, S., Intamanee, J., & Chungsiriporn, J. (2006). Factors affecting the volatilization of volatile organic compounds from wastewater *Songklanakarin J Sci Technol* (Vol. 28, pp. 1083-1093).

Burrows, H. D., Canle, L. M., Santaballa, J. A., & Steenken, S. (2002). Reaction pathways and mechanisms of photodegradation of pesticides. *J Photochem Photobiol B, 67*(2), 71-108. doi:10.1016/s1011-1344(02)00277-4

Butkovskyi, A., Jing, Y., Bergheim, H., Lazar, D., Gulyaeva, K., Odenmarck, S. R., . . . Eggen, T. (2021). Retention and distribution of pesticides in planted filter microcosms designed for treatment of agricultural surface runoff. *Science of the Total Environment, 778*, 146114. doi:10.1016/j.scitotenv.2021.146114

Caceres-Jensen, L., Rodriguez-Becerra, J., Sierra-Rosales, P., Escudey, M., Valdebenito, J., Neira-Albornoz, A., . . . Villagra, C. A. (2019). Electrochemical method to study the environmental behavior of Glyphosate on volcanic soils: Proposal of adsorption-desorption and transport mechanisms. *J Hazard Mater, 379*, 120746. doi:10.1016/j.jhazmat.2019.120746

Cai, D., Yang, X., Wang, S., Chao, Y., Morel, J. L., & Qiu, R. (2017). Effects of dissolved organic matter derived from forest leaf litter on biodegradation of phenanthrene in aqueous phase. *J Hazard Mater, 324*(Pt B), 516-525. doi:10.1016/j.jhazmat.2016.11.020

Cara, I. G., Filip, M., Bulgariu, L., Raus, L., Topa, D., & Jitareanu, G. (2021). Environmental Remediation of Metribuzin Herbicide by Mesoporous Carbon-Rich from Wheat Straw. *Applied Sciences-Basel, 11*(11), 4935. doi:ARTN 4935

10.3390/app11114935

Carleton, J. N., & Montas, H. J. (2010). An analysis of performance models for free water surface wetlands. *Water Research, 44*(12), 3595-3606. doi:https://doi.org/10.1016/j.watres.2010.04.008

Cessna, A. J., Donald, D. B., Bailey, J., & Waiser, M. (2015). Persistence of the Sulfonylurea Herbicides Sulfosulfuron, Rimsulfuron, and Nicosulfuron in Farm Dugouts (Ponds). *J Environ Qual, 44*(6), 1948-1955. doi:10.2134/jeq2014.11.0503

Chaumet, B., Probst, J.-L., Payré-Suc, V., Granouillac, F., Riboul, D., & Probst, A. (2022). Pond mitigation in dissolved and particulate pesticide transfers: Influence of storm events and seasonality (Auradé agricultural catchment, SW-France). *Journal of Environmental Management, 320*, 115911. doi:10.1016/j.jenvman.2022.115911

Chaumet, B., Probst, J. L., Eon, P., Camboulive, T., Riboul, D., Payre-Suc, V., . . . Probst, A. (2021). Role of Pond Sediments for Trapping Pesticides in an Agricultural Catchment (Aurade, SW France): Distribution and Controlling Factors. *Water, 13*(13), 1734. doi:ARTN 1734

10.3390/w13131734

ChemSpider. Retrieved from https://www.chemspider.com/

Chen, Y., Wen, Y., Zhou, J., Zhou, Q., Vymazal, J., & Kuschk, P. (2015). Transformation of chloroform in model treatment wetlands: from mass balance to microbial analysis. *Environ Sci Technol, 49*(10), 6198-6205. doi:10.1021/es506357e

Cheng, S., Xiao, J., Xiao, H., Zhang, L., & Wu, Z. (2007). Technical note phytoremediation of triazophos by Canna indica Linn. in a hydroponic system. *Int J Phytoremediation, 9*(6), 453-463. doi:10.1080/15226510701709531

Chien, S. H., & Clayton, W. R. (1980). Application of Elovich Equation to the Kinetics of Phosphate Release and Sorption in Soils. *Soil Science Society of America Journal, 44*(2), 265-268. doi:DOI 10.2136/sssaj1980.03615995004400020013x

Chouinard, A., Balch, G. C., Wootton, B. C., Jørgensen, S. E., & Anderson, B. C. (2014). Chapter 19 - SubWet 2.0. Modeling the Performance of Treatment Wetlands. *Developments in Environmental Modelling, 26*, 519-537. doi:https://doi.org/10.1016/B978-0-444-63249-4.00021-X

Climent, M. J., Herrero-Hernandez, E., Sanchez-Martin, M. J., Rodriguez-Cruz, M. S., Pedreros, P., & Urrutia, R. (2019). Residues of pesticides and some metabolites in dissolved and particulate phase in surface stream water of Cachapoal River basin, central Chile. *Environ Pollut, 251*, 90-101. doi:10.1016/j.envpol.2019.04.117

Collins, R. (2004). Fecal contamination of pastoral wetlands. *J Environ Qual, 33*(5), 1912-1918. doi:10.2134/jeq2004.1912

Comoretto, L., Arfib, B., Talva, R., Chauvelon, P., Pichaud, M., Chiron, S., & Hohener, P. (2008). Runoff of pesticides from rice fields in the Ile de Camargue (Rhone river delta, France): field study and modeling. *Environ Pollut, 151*(3), 486-493. doi:10.1016/j.envpol.2007.04.021

Convertino, M., Nardi, F., Kiker, G. A., Munoz-Carpena, R., Troccolli, A., & Linkov, I. (2013). Epitomes of Bottom-Up Hydro-Geo-Climatological Analysis to Face Sea Level Rise in Complex Coastal Ecosystems. 267-282. doi:10.1016/b978-0-12-384703-4.00502-5

Cryder, Z., Wolf, D., Carlan, C., & Gan, J. (2021). Removal of urban-use insecticides in a large-scale constructed wetland. *Environ Pollut, 268*(Pt A), 115586. doi:10.1016/j.envpol.2020.115586

Damalas, C. A., & Eleftherohorinos, I. G. (2011). Pesticide exposure, safety issues, and risk assessment indicators. *International Journal of Environmental Research and Public Health, 8*(5), 1402-1419.

Dash, D. M., & Osborne, W. J. (2020). Rapid biodegradation and biofilm-mediated bioremoval of organophosphorus pesticides using an indigenous Kosakonia oryzae strain -VITPSCQ3 in a Vertical-flow Packed Bed Biofilm Bioreactor. *Ecotoxicol Environ Saf, 192*, 110290. doi:10.1016/j.ecoenv.2020.110290 de Oliveira, M., Frihling, B. E. F., Velasques, J., Filho, F., Cavalheri, P. S., & Migliolo, L. (2020). Pharmaceuticals residues and xenobiotics contaminants: Occurrence, analytical techniques and sustainable alternatives for wastewater treatment. *Science of the Total Environment, 705*, 135568. doi:10.1016/j.scitotenv.2019.135568

Deer, H. M., & Beard, R. (2001). *Effect of water pH on the chemical stability of pesticides*. Retrieved from https://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1074&context=extension_histall

Douven, S., Paez, C. A., & Gommes, C. J. (2015). The range of validity of sorption kinetic models. *J Colloid Interface Sci, 448*, 437-450. doi:10.1016/j.jcis.2015.02.053

Drouin, G., Droz, B., Leresche, F., Payraudeau, S., Masbou, J., & Imfeld, G. (2021). *Direct and indirect photodegradation of atrazine and S-metolachlor in agriculturally impacted surface water and associated C and N isotope fractionation*. Cambridge University Press (CUP). Retrieved from https://dx.doi.org/10.33774/chemrxiv-2021-sg66n

Droz, B., Drouin, G., Maurer, L., Villette, C., Payraudeau, S., & Imfeld, G. (2021). Phase Transfer and Biodegradation of Pesticides in Water-Sediment Systems Explored by Compound-Specific Isotope Analysis and Conceptual Modeling. *Environ Sci Technol, 55*(8), 4720-4728. doi:10.1021/acs.est.0c06283

Farran, A., De Pablo, J., & Barcelo, D. (1988). Identification of organophosphorus insecticides and their hydrolysis products by liquid chromatography in combination with UV and thermospray-mass spectrometric detection. *Journal of Chromatography A, 455*, 163-172. doi:10.1016/s0021-9673(01)82115-4

Fernández-Pascual, E., Bork, M., Hensen, B., & Lange, J. (2020). Hydrological tracers for assessing transport and dissipation processes of pesticides in a model constructed wetland system. *Hydrology & Earth System Sciences, 24*(1). doi:https://doi.org/10.5194/hess-24-41-2020

Fiener, P., & Auerswald, K. (2003). Effectiveness of grassed waterways in reducing runoff and sediment delivery from agricultural watersheds. *J Environ Qual, 32*(3), 927-936. doi:10.2134/jeq2003.9270

Foo, K. Y., & Hameed, B. H. (2010). Insights into the modeling of adsorption isotherm systems. *Chemical Engineering Journal, 156*(1), 2-10. doi:10.1016/j.cej.2009.09.013

Frank, M. P., Graebing, P., & Chib, J. S. (2002). Effect of soil moisture and sample depth on pesticide photolysis. *J Agric Food Chem, 50*(9), 2607-2614. doi:10.1021/jf0115746

Gan, J., Lee, S. J., Liu, W. P., Haver, D. L., & Kabashima, J. N. (2005). Distribution and persistence of pyrethroids in runoff sediments. *J Environ Qual, 34*(3), 836-841. doi:10.2134/jeq2004.0240

Garcia-Rodríguez, A., Matamoros, V., Fontàs, C., & Salvadó, V. (2014). The ability of biologically based wastewater treatment systems to remove emerging organic contaminants—a review. *Environmental Science and Pollution Research, 21*(20), 11708-11728. doi:10.1007/s11356-013-2448-5

Gassmann, M., Olsson, O., Stamm, C., Weiler, M., & Kümmerer, K. (2015). Physico-chemical characteristics affect the spatial distribution of pesticide and transformation product loss to an agricultural brook. *Science of The Total Environment, 532*, 733-743. doi:https://doi.org/10.1016/j.scitotenv.2015.06.068

Gaullier, C., Baran, N., Dousset, S., Devau, N., Billet, D., Kitzinger, G., & Coisy, E. (2019). Wetland hydrodynamics and mitigation of pesticides and their metabolites at pilot-scale. *Ecological Engineering, 136*, 185-192. doi:https://doi.org/10.1016/j.ecoleng.2019.06.019

Gaullier, C., Dousset, S., Baran, N., Kitzinger, G., & Coureau, C. (2020). Influence of hydrodynamics on the water pathway and spatial distribution of pesticide and metabolite concentrations in constructed wetlands. *J Environ Manage, 270*, 110690. doi:10.1016/j.jenvman.2020.110690

Gaullier, C., Dousset, S., Billet, D., & Baran, N. (2018). Is pesticide sorption by constructed wetland sediments governed by water level and water dynamics? *Environ Sci Pollut Res Int, 25*(15), 14324-14335. doi:10.1007/s11356-017-9123-1

Gerecke, A. C., Canonica, S., Muller, S. R., Scharer, M., & Schwarzenbach, R. P. (2001). Quantification of dissolved natural organic matter (DOM) mediated phototransformation of phenylurea herbicides in lakes. *Environ Sci Technol, 35*(19), 3915-3923. doi:10.1021/es010103x

Gobas, F., Lai, H. F., Mackay, D., Padilla, L. E., Goetz, A., & Jackson, S. H. (2018). AGRO-2014: A time dependent model for assessing the fate and food-web bioaccumulation of organic pesticides in farm ponds: Model testing and performance analysis. *Science of the Total Environment, 639*, 1324-1333. doi:10.1016/j.scitotenv.2018.05.115

Gramlich, A., Stoll, S., Stamm, C., Walter, T., & Prasuhn, V. (2018). Effects of artificial land drainage on hydrology, nutrient and pesticide fluxes from agricultural fields - A review. *Agriculture Ecosystems & Environment, 266*, 84-99. doi:10.1016/j.agee.2018.04.005

Green, R. E., & Karickhoff, S. W. (1990). Sorption Estimates for Modeling *Pesticides in the Soil Environment: Processes, Impacts and Modeling* (pp. 79-101).

Gregoire, C., Elsaesser, D., Huguenot, D., Lange, J., Lebeau, T., Merli, A., . . . Schütz, T. (2009). Mitigation of agricultural nonpoint-source pesticide pollution in artificial wetland ecosystems. *Environmental Chemistry Letters, 7*(3), 205-231. doi:https://doi.org/10.1007/s10311-008-0167-9

Grinham, A., Albert, S., Deering, N., Dunbabin, M., Bastviken, D., Sherman, B., . . . Evans, C. D. (2018). The importance of small artificial water bodies as sources of methane emissions in Queensland, Australia. *Hydrology and Earth System Sciences, 22*(10), 5281-5298. doi:10.5194/hess-22-5281-2018

Gu, L., Dai, B., Zhu, D. Z., Hua, Z. L., Liu, X. D., van Duin, B., & Mahmood, K. (2017). Sediment modelling and design optimization for stormwater ponds. *Canadian Water Resources Journal, 42*(1), 70-87. doi:10.1080/07011784.2016.1210542

Guo, L., Jury, W. A., Wagenet, R. J., & Flury, M. (2000). Dependence of pesticide degradation on sorption: nonequilibrium model and application to soil reactors. *Journal of Contaminant Hydrology, 43*(1), 45-62. doi:Doi 10.1016/S0169-7722(99)00097-2

Haith, D. A. (2010). Ecological risk assessment of pesticide runoff from grass surfaces. *Environmental Science & Technology, 44*(16), 6496-6502.

Halbach, K., Möder, M., Schrader, S., Liebmann, L., Schäfer, R. B., Schneeweiss, A., . . . Reemtsma, T. (2021). Small streams–large concentrations? Pesticide monitoring in small agricultural streams in Germany during dry weather and rainfall. *Water Research, 203*, 117535. doi:https://doi.org/10.1016/j.watres.2021.117535

Hamdi, A. (2007). Identification of point sources in two-dimensional advection-diffusion-reaction equation: application to pollution sources in a river. Stationary case. *Inverse Problems in Science and Engineering, 15*(8), 855-870.

Hand, L. H., Kuet, S. F., Lane, M. C., Maund, S. J., Warinton, J. S., & Hill, I. R. (2001). Influences of aquatic plants on the fate of the pyrethroid insecticide Lambida-cyhalothrin in aquatic environments. *Environmental Toxicology and Chemistry: An International Journal, 20*(8), 1740-1745. doi:10.1897/1551-5028(2001)020<1740:ioapot>2.0.co;2

Hantush, M. M., Kalin, L., Isik, S., & Yucekaya, A. (2013). Nutrient Dynamics in Flooded Wetlands. I: Model Development. *Journal of Hydrologic Engineering, 18*(12), 1709-1723. doi:10.1061/(Asce)He.1943-5584.0000741

Hawley, N., Redder, T., Beletsky, R., Verhamme, E., Beletsky, D., & DePinto, J. V. (2014). Sediment resuspension in Saginaw Bay. *Journal of Great Lakes Research, 40*, 18-27. doi:10.1016/j.jglr.2013.11.010

Hijosa-Valsero, M., Becares, E., Fernandez-Alaez, C., Fernandez-Alaez, M., Mayo, R., & Jimenez, J. J. (2016). Chemical pollution in inland shallow lakes in the Mediterranean region (NW Spain): PAHs, insecticides and herbicides in water and sediments. *Science of the Total Environment, 544*, 797-810. doi:10.1016/j.scitotenv.2015.11.160

Hijosa-Valsero, M., Matamoros, V., Sidrach-Cardona, R., Martin-Villacorta, J., Becares, E., & Bayona, J. M. (2010). Comprehensive assessment of the design configuration of constructed wetlands for the removal of pharmaceuticals and personal care products from urban wastewaters. *Water Res, 44*(12), 3669-3678. doi:10.1016/j.watres.2010.04.022

Ho, Y. S. (2006). Review of second-order models for adsorption systems. *J Hazard Mater, 136*(3), 681-689. doi:10.1016/j.jhazmat.2005.12.043

Ho, Y. S., & McKay, G. (1998). A comparison of chemisorption kinetic models applied to pollutant removal on various sorbents. *Process Safety and Environmental Protection, 76*(B4), 332-340. doi:Doi 10.1205/095758298529696

Ho, Y. S., & McKay, G. (1999). Batch lead (II) removal from aqueous solution by peat: Equilibrium and kinetics. *Process Safety and Environmental Protection, 77*(B3), 165-173. doi:Doi 10.1205/095758299529983

Ho, Y. S., & McKay, G. (2002). Application of kinetic models to the sorption of copper(II) on to peat. *Adsorption Science & Technology, 20*(8), 797-815. doi:Doi 10.1260/026361702321104282

Holland, J., M, B., T, B., S, S., S.R, T., Alexander, C., & Thomas, C. F. (2004). The spatial dynamics and movement of Pterostichus melanarius and P. madidus (Carabidae) between and within arable fields in the UK. *International Journal of Ecology and Environmental Sciences, 30*, 35-50.

Holland, J. F., Martin, J. F., Granata, T., Bouchard, V., Quigley, M., & Brown, L. (2004). Effects of wetland depth and flow rate on residence time distribution characteristics. *Ecological Engineering, 23*(3), 189-203. doi:https://doi.org/10.1016/j.ecoleng.2004.09.003

Höllrigl-Rosta. (2017). Establishment of a concept for comparative risk assessment of plant protection products with special focus on the risks to the environment.

Huang, W. L., Ping, P. A., Yu, Z. Q., & Fu, H. M. (2003). Effects of organic matter heterogeneity on sorption and desorption of organic contaminants by soils and sediments. *Applied Geochemistry, 18*(7), 955-972. doi:Doi 10.1016/S0883-2927(02)00205-6

Ibanez, M., Sancho, J. V., Pozo, O. J., & Hernandez, F. (2006). Use of liquid chromatography quadrupole time-of-flight mass spectrometry in the elucidation of transformation products and metabolites of pesticides. Diazinon as a case study. *Anal Bioanal Chem, 384*(2), 448-457. doi:10.1007/s00216-005-0167-6

Imfeld, G., Braeckevelt, M., Kuschk, P., & Richnow, H. H. (2009). Monitoring and assessing processes of organic chemicals removal in constructed wetlands. *Chemosphere, 74*(3), 349-362. doi:10.1016/j.chemosphere.2008.09.062

Imfeld, G., Payraudeau, S., Tournebize, J., Sauvage, S., Macary, F., Chaumont, C., . . . Probst, J. L. (2021). The Role of Ponds in Pesticide Dissipation at the Agricultural Catchment Scale: A Critical Review. *Water, 13*(9), 1202. doi:ARTN 1202

10.3390/w13091202

Inao, K., & Kitamura, Y. (1999). Pesticide paddy field model (PADDY) for predicting pesticide concentrations in water and soil in paddy fields. *Pesticide Science, 55*(1), 38-46. doi:DOI 10.1002/ps.2780550108

Jacobs, C. M. J., & Adriaanse, P. (2012). *Pesticide volatilization from small surface waters: rationale of a new parameterization for TOXSWA* (1566-7197). Retrieved from https://edepot.wur.nl/213995

Jamieson, R. C., Joy, D. M., Lee, H., Kostaschuk, R., & Gordon, R. J. (2005). Resuspension of sediment-associated Escherichia coli in a natural stream. *J Environ Qual, 34*(2), 581-589. doi:10.2134/jeq2005.0581

Johnson, A., Llewellyn, N., Smith, J., van der Gast, C., Lilley, A., Singer, A., & Thompson, I. (2004). The role of microbial community composition and groundwater chemistry in determining isoproturon degradation potential in UK aquifers. *FEMS Microbiol Ecol, 49*(1), 71-82. doi:10.1016/j.femsec.2004.03.015

Jorda, H., Huber, K., Kunkel, A., Vanderborght, J., Javaux, M., Oberdorster, C., . . . Schnepf, A. (2021). Mechanistic modeling of pesticide uptake with a 3D plant architecture model. *Environ Sci Pollut Res Int, 28*(39), 55678-55689. doi:10.1007/s11356-021-14878-3

Kadlec, R. H., & Wallace, S. (2008). *Treatment wetlands*: CRC press.

Kah, M., & Brown, C. D. (2006). Adsorption of Ionisable Pesticides in Soils. In G. W. Ware, D. M. Whitacre, L. A. Albert, P. de Voogt, C. P. Gerba, O. Hutzinger, J. B. Knaak, F. L. Mayer, D. P. Morgan, D. L. Park, R. S. Tjeerdema, R. S. H. Yang, & F. A. Gunther (Eds.), *Reviews of Environmental Contamination and Toxicology: Continuation of Residue Reviews* (pp. 149-217). New York, NY: Springer New York.

Kalin, L., & Hantush, M. M. (2017). Special Issue on Advances in Wetland Hydrology and Water Quality Processes Modeling. *Journal of Hydrologic Engineering, 22*(1), D2016001. doi:10.1061/(asce)he.1943-5584.0001483

Kang, D., Doudrick, K., Park, N., Choi, Y., Kim, K., & Jeon, J. (2020). Identification of transformation products to characterize the ability of a natural wetland to degrade synthetic organic pollutants. *Water Res, 187*, 116425. doi:10.1016/j.watres.2020.116425

Karpuzcu, M. E., Sedlak, D. L., & Stringfellow, W. T. (2013). Biotransformation of chlorpyrifos in riparian wetlands in agricultural watersheds: implications for wetland management. *J Hazard Mater, 244-245*, 111-120. doi:10.1016/j.jhazmat.2012.11.047

Kasak, K., Kill, K., Parn, J., & Mander, U. (2018). Efficiency of a newly established in-stream constructed wetland treating diffuse agricultural pollution. *Ecological Engineering, 119*, 1-7. doi:10.1016/j.ecoleng.2018.05.015

Katagi, T. (2006). Behavior of pesticides in water-sediment systems. *Reviews of Environmental Contamination and Toxicology, Vol 187, 187*, 133-251. doi:Doi 10.1007/0-387-32885-8_4

Katagi, T. (2013). Aerobic microbial transformation of pesticides in surface water. *Journal of Pesticide Science, 38*(1-2), 10-26. doi:10.1584/jpestics.D12-053

Katagi, T., & Tanaka, H. (2016). Metabolism, bioaccumulation, and toxicity of pesticides in aquatic insect larvae. *J Pestic Sci, 41*(2), 25-37. doi:10.1584/jpestics.D15-064

Kaur, P., & Kaur, P. (2018). Time and temperature dependent adsorption-desorption behaviour of pretilachlor in soil. *Ecotoxicol Environ Saf, 161*, 145-155. doi:10.1016/j.ecoenv.2018.05.081

Kim, S., Chen, J., Cheng, T., Gindulyte, A., He, J., He, S., . . . Bolton, E. E. (2021). PubChem in 2021: new data content and improved web interfaces. *Nucleic Acids Res, 49*(D1), D1388-D1395. doi:10.1093/nar/gkaa971

Kipkoech, W. (2019). *Profiling of pesticide residues in Mobego-Kabianga wetland ecosystem and its retention efficiency.* University of Kabianga.

Kjellin, J., Hallin, S., & Worman, A. (2007). Spatial variations in denitrification activity in wetland sediments explained by hydrology and denitrifying community structure. *Water Res, 41*(20), 4710-4720. doi:10.1016/j.watres.2007.06.053

Koskiaho, J. (2003). Flow velocity retardation and sediment retention in two constructed wetland-ponds. *Ecological Engineering, 19*(5), 325-337. doi:Pii S0925-8574(02)00119-2

Doi 10.1016/S0925-8574(02)00119-2

Koskinen, W., O'Connor, G., & Cheng, H. (1979). Characterization of hysteresis in the desorption of 2, 4, 5-T from soils. *Soil Science Society of America Journal, 43*(5), 871-874. doi:https://doi.org/10.2136/sssaj1979.03615995004300050010x

Krishna, K. R., & Philip, L. (2008). Adsorption and desorption characteristics of lindane, carbofuran and methyl parathion on various Indian soils. *Journal of hazardous materials, 160*(2-3), 559-567. doi:10.1016/j.jhazmat.2008.03.107

Laabs, V., Wehrhan, A., Pinto, A., Dores, E., & Amelung, W. (2007). Pesticide fate in tropical wetlands of Brazil: an aquatic microcosm study under semi-field conditions. *Chemosphere, 67*(5), 975-989. doi:10.1016/j.chemosphere.2006.10.067

Lagunas-Allué, L., Martínez-Soria, M.-T., Sanz-Asensio, J., Salvador, A., Ferronato, C., & Chovelon, J. M. (2010). Photocatalytic degradation of boscalid in aqueous titanium dioxide suspension: Identification of intermediates and degradation pathways. *Applied Catalysis B: Environmental, 98*(3-4), 122-131. doi:https://doi.org/10.1016/j.apcatb.2010.05.020

Lam, M. W., Tantuco, K., & Mabury, S. A. (2003). PhotoFate: A new approach in accounting for the contribution of indirect photolysis of pesticides and pharmaceuticals in surface waters. *Environmental Science & Technology, 37*(5), 899-907. doi:10.1021/es025902+

Lange, J., Schuetz, T., Gregoire, C., Elsasser, D., Schulz, R., Passeport, E., & Tournebize, J. (2011). Multi-tracer experiments to characterise contaminant mitigation capacities for different types of artificial wetlands. *International journal of environmental analytical chemistry, 91*(7-8), 768-785. doi:10.1080/03067319.2010.525635

Langergraber, G. (2008). Modeling of processes in subsurface flow constructed wetlands: A review. *Vadose Zone Journal, 7*(2), 830-842. doi:10.2136/vzj2007.0054

Larson, R. A., & Weber, E. J. (1994). *Reaction mechanisms in environmental organic chemistry*: CRC press.

Le Cor, F., Slaby, S., Dufour, V., Iuretig, A., Feidt, C., Dauchy, X., & Banas, D. (2021). Occurrence of pesticides and their transformation products in headwater streams: Contamination status and effect of ponds on contaminant concentrations. *Science of the Total Environment, 788*, 147715. doi:10.1016/j.scitotenv.2021.147715

Le Person, A., Siampiringue, M., Sarakha, M., Moncomble, A., & Cornard, J.-P. (2016). The photo-degradation of mesotrione, a triketone herbicide, in the presence of Cu II ions. *Journal of Photochemistry and Photobiology A: Chemistry, 315*, 76-86. doi:10.1016/j.jphotochem.2015.09.010

Lee, S., Gan, J., Kim, J. S., Kabashima, J. N., & Crowley, D. E. (2004). Microbial transformation of pyrethroid insecticides in aqueous and sediment phases. *Environ Toxicol Chem, 23*(1), 1-6. doi:10.1897/03-114

Lewis, K. A., Tzilivakis, J., Warner, D. J., & Green, A. (2016). An international database for pesticide risk assessments and management. *Human and Ecological Risk Assessment, 22*(4), 1050-1064. doi:10.1080/10807039.2015.1133242

Li, H., Sheng, G., Teppen, B. J., Johnston, C. T., & Boyd, S. A. (2003). Sorption and Desorption of Pesticides by Clay Minerals and Humic Acid-Clay Complexes. *Soil Science Society of America Journal, 67*(1), 122-131. doi:10.2136/sssaj2003.1220a

Li, J. S., Fei, L. J., Chen, Z., & Sun, X. L. (2017). Particle size distribution and settling velocity of sediments in water diverted from the Yellow River during border-strip irrigation. *Tecnología y ciencias del agua, 8*(2), 31-41. doi:DOI 10.24850/j-tyca-2017-02-03

Limousin, G., Gaudet, J. P., Charlet, L., Szenknect, S., Barthes, V., & Krimissa, M. (2007). Sorption isotherms: A review on physical bases, modeling and measurement. *Applied Geochemistry, 22*(2), 249-275. doi:10.1016/j.apgeochem.2006.09.010

Lin, J. X., & Wang, L. (2009). Comparison between linear and non-linear forms of pseudo-first-order and pseudo-second-order adsorption kinetic models for the removal of methylene blue by activated carbon. *Frontiers of Environmental Science & Engineering in China, 3*(3), 320-324. doi:10.1007/s11783-009-0030-7

Lin, K. D., & Yuan, D. X. (2005). Degradation kinetics and products of triazophos in intertidal sediment. *J Environ Sci (China), 17*(6), 933-936.

Liu, B., McConnell, L. L., & Torrents, A. (2001). Hydrolysis of chlorpyrifos in natural waters of the Chesapeake Bay. *Chemosphere, 44*(6), 1315-1323. doi:10.1016/s0045-6535(00)00506-3

Liu, T., Xu, S., Lu, S., Qin, P., Bi, B., Ding, H., . . . Liu, X. (2019). A review on removal of organophosphorus pesticides in constructed wetland: Performance, mechanism and influencing factors. *Science of the Total Environment, 651*(Pt 2), 2247-2268. doi:10.1016/j.scitotenv.2018.10.087

Llorens, E., Saaltink, M. W., Poch, M., & Garcia, J. (2011). Bacterial transformation and biodegradation processes simulation in horizontal subsurface flow constructed wetlands using CWM1-RETRASO. *Bioresour Technol, 102*(2), 928-936. doi:10.1016/j.biortech.2010.09.038

Luo, J., Huai, W., & Wang, P. (2016). Contaminant transport in a three-zone wetland: Dispersion and ecological degradation. *Journal of Hydrology, 534*, 341-351. doi:https://doi.org/10.1016/j.jhydrol.2016.01.008

Luo, J. P., Ma, M., Liu, C., Zha, J. M., & Wang, Z. J. (2009). Impacts of particulate organic carbon and dissolved organic carbon on removal of polycyclic aromatic hydrocarbons, organochlorine pesticides, and nonylphenols in a wetland. *Journal of Soils and Sediments, 9*(3), 180-187. doi:10.1007/s11368-009-0081-1

Lv, T., Carvalho, P. N., Zhang, L., Zhang, Y., Button, M., Arias, C. A., . . . Brix, H. (2017). Functionality of microbial communities in constructed wetlands used for pesticide remediation: Influence of system design and sampling strategy. *Water Res, 110*, 241-251. doi:10.1016/j.watres.2016.12.021

Lyu, T., Zhang, L., Xu, X., Arias, C. A., Brix, H., & Carvalho, P. N. (2018). Removal of the pesticide tebuconazole in constructed wetlands: Design comparison, influencing factors and modelling. *Environ Pollut, 233*, 71-80. doi:10.1016/j.envpol.2017.10.040

Ma, Y., Li, J., Wu, J., Kong, Z., Feinstein, L. M., Ding, X., . . . Wu, L. (2018). Bacterial and Fungal Community Composition and Functional Activity Associated with Lake Wetland Water Level Gradients. *Sci Rep, 8*(1), 760. doi:10.1038/s41598-018-19153-z

MacIntyre, M. E., Warner, B. G., & Slawson, R. M. (2006). Escherichia coli control in a surface flow treatment wetland. *J Water Health, 4*(2), 211-214.

Mackay, D., Shiu, W.-Y., & Lee, S. C. (2006). *Handbook of physical-chemical properties and environmental fate for organic chemicals*: CRC press.

Mahabali, S., & Spanoghe, P. (2014). Mitigation of Two Insecticides by Wetland Plants: Feasibility Study for the Treatment of Agricultural Runoff in Suriname (South America). *Water Air and Soil Pollution, 225*(1), 1771. doi:ARTN 1771

10.1007/s11270-013-1771-2

Maillard, E., & Imfeld, G. (2014). Pesticide mass budget in a stormwater wetland. *Environ Sci Technol, 48*(15), 8603-8611. doi:10.1021/es500586x

Maillard, E., Payraudeau, S., Faivre, E., Gregoire, C., Gangloff, S., & Imfeld, G. (2011). Removal of pesticide mixtures in a stormwater wetland collecting runoff from a vineyard catchment. *Science of the Total Environment, 409*(11), 2317-2324. doi:10.1016/j.scitotenv.2011.01.057

Maillard, E., Payraudeau, S., Ortiz, F., & Imfeld, G. (2012). Removal of dissolved pesticide mixtures by a stormwater wetland receiving runoff from a vineyard catchment: an inter-annual comparison. *International journal of environmental analytical chemistry, 92*(8), 979-994. doi:10.1080/03067319.2011.609935

Maltais-Landry, G., Maranger, R., Brisson, J., & Chazarenc, F. (2009). Nitrogen transformations and retention in planted and artificially aerated constructed wetlands. *Water Res, 43*(2), 535-545. doi:10.1016/j.watres.2008.10.040

Mamy, L., & Barriuso, E. (2007). Desorption and time-dependent sorption of herbicides in soils. *European Journal of Soil Science, 58*(1), 174-187. doi:10.1111/j.1365-2389.2006.00822.x

Marois, D. E., & Mitsch, W. J. (2016). Modeling phosphorus retention at low concentrations in Florida Everglades mesocosms. *Ecological Modelling, 319*, 42-62. doi:10.1016/j.ecolmodel.2015.09.024

Martínez-Lavanchy, P., Chen, Z., Lünsmann, V., Marin-Cevada, V., Vilchez-Vargas, R., Pieper, D., . . . Junca, H. (2015). Microbial toluene removal in hypoxic model constructed wetlands occurs predominantly via the ring monooxygenation pathway. *Applied and Environmental Microbiology, 81*(18), 6241-6252. doi:10.1128/AEM.01822-15

Martinez, C. J., & Wise, W. R. (2003). Analysis of constructed treatment wetland hydraulics with the transient storage model OTIS. *Ecological Engineering, 20*(3), 211-222. doi:https://doi.org/10.1016/S0925-8574(03)00029-6

Martinez, C. J., & Wise, W. R. (2003). Hydraulic analysis of Orlando Easterly Wetland. *Journal of Environmental Engineering, 129*(6), 553-560. doi:10.1061/(Asce)0733-9372(2003)129:6(553)

Masbou, J., Drouin, G., Payraudeau, S., & Imfeld, G. (2018). Carbon and nitrogen stable isotope fractionation during abiotic hydrolysis of pesticides. *Chemosphere, 213*, 368-376. doi:10.1016/j.chemosphere.2018.09.056

Matamoros, V., & Bayona, J. M. (2006). Elimination of pharmaceuticals and personal care products in subsurface flow constructed wetlands. *Environ Sci Technol, 40*(18), 5811-5816. doi:10.1021/es0607741

Mathon, B., Coquery, M., Miege, C., Vandycke, A., & Choubert, J. M. (2019). Influence of water depth and season on the photodegradation of micropollutants in a free-water surface constructed wetland receiving treated wastewater. *Chemosphere, 235*, 260-270. doi:10.1016/j.chemosphere.2019.06.140

Maynard, J. (2009). *Spatial and temporal dynamics of carbon and nutrient cycling in a eutrophic flow-through wetland.* Ph. D. dissertation. University of California, Davis.

McBride, M. (1994). *Environmental Chemistry of Soils 1994*: Oxford University Press: New York.

McBride, M. B. (1994). *Environna ental Chemistry Of So| | LS* (Vol. 141580): Obtenido de Recuperado de: https://d1wqtxts1xzle7. cloudfront. net/35536696 ….

Mccall, P. J., & Agin, G. L. (1985). Desorption-Kinetics of Picloram as Affected by Residence Time in the Soil. *Environmental Toxicology and Chemistry, 4*(1), 37-44. doi:10.1002/etc.5620040106

Mckay, G. (1995). *Use of Adsorbents for the Removal of Pollutants from Wastewater*: CRC press.

Meng, P. P., Pei, H. Y., Hu, W. R., Shao, Y. Y., & Li, Z. (2014). How to increase microbial degradation in constructed wetlands: Influencing factors and improvement measures. *Bioresource Technology, 157*, 316-326. doi:10.1016/j.biortech.2014.01.095

Miller, P. L., & Chin, Y. P. (2005). Indirect photolysis promoted by natural and engineered wetland water constituents: processes leading to alachlor degradation. *Environ Sci Technol, 39*(12), 4454-4462. doi:10.1021/es049111e

Moore, M. T., Cooper, C. M., Smith, S., Cullum, R. F., Knight, S. S., Locke, M. A., & Bennett, E. R. (2007). Diazinon mitigation in constructed Wetlands: Influence of vegetation. *Water Air and Soil Pollution, 184*(1-4), 313-321. doi:10.1007/s11270-007-9418-9

Mugni, H., Demetrio, P., Bulus, G., Ronco, A., & Bonetto, C. (2011). Effect of aquatic vegetation on the persistence of cypermethrin toxicity in water. *Bull Environ Contam Toxicol, 86*(1), 23-27. doi:10.1007/s00128-010-0143-5

Mulligan, R. A., Tomco, P. L., Howard, M. W., Schempp, T. T., Stewart, D. J., Stacey, P. M., . . . Tjeerdema, R. S. (2016). Aerobic versus Anaerobic Microbial Degradation of Clothianidin under Simulated California Rice Field Conditions. *J Agric Food Chem, 64*(38), 7059-7067. doi:10.1021/acs.jafc.6b02055

Nakano, Y., Yoshida, T., & Inoue, T. (2004). A study on pesticide runoff from paddy fields to a river in rural region--2: development and application of a mathematical model. *Water Res, 38*(13), 3023-3030. doi:10.1016/j.watres.2004.02.014

O'Geen, A. T., Budd, R., Gan, J., Maynard, J. J., Parikh, S. J., & Dahlgren, R. A. (2010). Chapter One - Mitigating Nonpoint Source Pollution in Agriculture with Constructed and Restored Wetlands. In D. L. Sparks (Ed.), *Advances in agronomy* (Vol. 108, pp. 1-76): Academic Press.

OCDE. (2000). *Test No. 106: Adsorption -- Desorption Using a Batch Equilibrium Method*.

Olvera-Velona, A., Benoit, P., Barriuso, E., & Ortiz-Hernandez, L. (2008). Sorption and desorption of organophosphate pesticides, parathion and cadusafos, on tropical agricultural soils. *Agronomy for Sustainable Development, 28*(2), 231-238. doi:10.1051/agro:2008009

Ortega-Calvo, J. J., & Saiz-Jimenez, C. (1998). Effect of humic fractions and clay on biodegradation of phenanthrene by a Pseudomonas fluorescens strain isolated from soil. *Appl Environ Microbiol, 64*(8), 3123-3126. doi:10.1128/AEM.64.8.3123-3126.1998

Papaevangelou, V. A., Gikas, G. D., Vryzas, Z., & Tsihrintzis, V. A. (2017). Treatment of agricultural equipment rinsing water containing a fungicide in pilot-scale horizontal subsurface flow constructed wetlands. *Ecological Engineering, 101*, 193-200. doi:10.1016/j.ecoleng.2017.01.045

Papp, R. (2009). Les propriétés environnementales des substances. *Collège National d'Experts en Environnement de l'Industrie Chimique, 126*.

Passeport, E., Benoit, P., Bergheaud, V., Coquet, Y., & Tournebize, J. (2011a). Epoxiconazole degradation from artificial wetland and forest buffer substrates under flooded conditions. *Chemical Engineering Journal, 173*(3), 760-765. doi:10.1016/j.cej.2011.08.044

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*(7), 1669-1676. doi:10.1002/etc.554

Passeport, E., Tournebize, J., Chaumont, C., Guenne, A., & Coquet, Y. (2013). Pesticide contamination interception strategy and removal efficiency in forest buffer and artificial wetland in a tile-drained agricultural watershed. *Chemosphere, 91*(9), 1289-1296. doi:10.1016/j.chemosphere.2013.02.053

Passeport, E., Tournebize, J., Jankowfsky, S., Promse, B., Chaumont, C., Coquet, Y., & Lange, J. (2010). Artificial Wetland and Forest Buffer Zone: Hydraulic and Tracer Characterization. *Vadose Zone Journal, 9*(1), 73-84. doi:10.2136/vzj2008.0164

Paszko, T., Muszynski, P., Materska, M., Bojanowska, M., Kostecka, M., & Jackowska, I. (2016). Adsorption and degradation of phenoxyalkanoic acid herbicides in soils: A review. *Environ Toxicol Chem, 35*(2), 271-286. doi:10.1002/etc.3212

Pattanayak, S., Chakraborty, S., Biswas, S., Chattopadhyay, D., & Chakraborty, M. (2018). Degradation of Methyl Parathion, a common pesticide and fluorescence quenching of Rhodamine B, a carcinogen using β-d glucan stabilized gold nanoparticles. *Journal of Saudi Chemical Society, 22*(8), 937-948. doi:10.1016/j.jscs.2018.02.004

Pignatello, J. (1989). Sorption dynamics of organic compounds in soils and sediments. *Reactions and movement of organic chemicals in soils, 22*, 45-80. doi:10.2136/sssaspecpub22.c3

Pignatello, J. J. (1999). The measurement and interpretation of sorption and desorption rates for organic compounds in soil media. *Advances in agronomy, 69*, 1-73. doi:https://doi.org/10.1016/S0065-2113(08)60946-3

Pignatello, J. J., & Xing, B. S. (1996). Mechanisms of slow sorption of organic chemicals to natural particles. *Environmental Science & Technology, 30*(1), 1-11. doi:DOI 10.1021/es940683g

Pohlman, A. A., & Mill, T. (1983). Peroxy Radical Interactions with Soil Constituents. *Soil Science Society of America Journal, 47*(5), 922-927. doi:DOI 10.2136/sssaj1983.03615995004700050016x

Poissant, L., Beauvais, C., Lafrance, P., & Deblois, C. (2008). Pesticides in fluvial wetlands catchments under intensive agricultural activities. *Science of the Total Environment, 404*(1), 182-195. doi:10.1016/j.scitotenv.2008.05.030

Prochaska, C. A., Zouboulis, A. I., & Eskridge, K. M. (2007). Performance of pilot-scale vertical-flow constructed wetlands, as affected by season, substrate, hydraulic load and frequency of application of simulated urban sewage. *Ecological Engineering, 31*(1), 57-66. doi:10.1016/j.ecoleng.2007.05.007

PSD. (2005). Pesticide Data Requirement

Handbook. Retrieved from http://sitem.herts.ac.uk/aeru/ppdb/

PubChem. (2021). National Institutes of Health (NIH). *National Center for Biotechnology Information.* Retrieved from https://pubchem.ncbi.nlm.nih.gov/search/search.cgi

Rai, P. K. (2009). Heavy Metal Phytoremediation from Aquatic Ecosystems with Special Reference to Macrophytes. *Critical Reviews in Environmental Science and Technology, 39*(9), 697-753. doi:Pii 914514507

10.1080/10643380801910058

Rani, S., Madan, V. K., & Kathpal, T. S. (2001). Persistence and dissipation behavior of triazophos in canal water under Indian climatic conditions. *Ecotoxicol Environ Saf, 50*(1), 82-84. doi:10.1006/eesa.2001.2103

Rayen, F., Behnam, T., & Dominique, P. (2019). Optimization of a raceway pond system for wastewater treatment: a review. *Crit Rev Biotechnol, 39*(3), 422-435. doi:10.1080/07388551.2019.1571007

Reddy, K. R., & DeLaune, R. D. (2008). *Biogeochemistry of wetlands: science and applications*: CRC press.

Rijn, L. C. v. (1984). Sediment transport, part II: suspended load transport. *Journal of Hydraulic Engineering, 110*(11), 1613-1641. doi:https://doi.org/10.1061/(ASCE)0733-9429(1984)110:11(1613)

Rogers, M. R., & Stringfellow, W. T. (2009). Partitioning of chlorpyrifos to soil and plants in vegetated agricultural drainage ditches. *Chemosphere, 75*(1), 109-114. doi:10.1016/j.chemosphere.2008.11.036

Rong, X., Zhao, G., Fein, J. B., Yu, Q., & Huang, Q. (2019). Role of interfacial reactions in biodegradation: A case study in a montmorillonite, Pseudomonas sp. Z1 and methyl parathion ternary system. *J Hazard Mater, 365*, 245-251. doi:10.1016/j.jhazmat.2018.11.019

Rose, M. T., Crossan, A. N., & Kennedy, I. R. (2008). The effect of vegetation on pesticide dissipation from ponded treatment wetlands: quantification using a simple model. *Chemosphere, 72*(7), 999-1005. doi:10.1016/j.chemosphere.2008.04.059

Rose, M. T., Sanchez-Bayo, F., Crossan, A. N., & Kennedy, I. R. (2006). Pesticide removal from cotton farm tailwater by a pilot-scale ponded wetland. *Chemosphere, 63*(11), 1849-1858. doi:10.1016/j.chemosphere.2005.10.024

Ruiz-Medina, A., Soler-Gallardo, M. I., & Llorent-Martinez, E. J. (2018). Enhanced Quenching Effect of Neonicotinoid Pesticides on Time-Resolved Terbium Luminescence in Presence of Surfactants. *Journal of Chemistry, 2018*, 5378736. doi:Artn 5378736

10.1155/2018/5378736

Saaristo, M., Brodin, T., Balshine, S., Bertram, M. G., Brooks, B. W., Ehlman, S. M., . . . Arnold, K. E. (2018). Direct and indirect effects of chemical contaminants on the behaviour, ecology and evolution of wildlife. *Proc Biol Sci, 285*(1885), 20181297. doi:10.1098/rspb.2018.1297

Sabadie, J. (2002). Nicosulfuron: alcoholysis, chemical hydrolysis, and degradation on various minerals. *J Agric Food Chem, 50*(3), 526-531. doi:10.1021/jf010873s

Saha, S., & Kulshrestha, G. (2002). Degradation of sulfosulfuron, a sulfonylurea herbicide, as influenced by abiotic factors. *Journal of agricultural and food chemistry, 50*(16), 4572-4575. doi:10.1021/jf0116653

Sahin, C., & Karpuzcu, M. E. (2020). Mitigation of organophosphate pesticide pollution in agricultural watersheds. *Science of the Total Environment, 710*, 136261. doi:10.1016/j.scitotenv.2019.136261

Sakellarides, T. M., Siskos, M. G., & Albanis, T. A. (2003). Photodegradation of selected organophosphorus insecticides under sunlight in different natural waters and soils. *International journal of environmental analytical chemistry, 83*(1), 33-50. doi:10.1080/0306731031000104227

Samso, R., & Garcia, J. (2013). Bacteria distribution and dynamics in constructed wetlands based on modelling results. *Science of the Total Environment, 461-462*, 430-440. doi:10.1016/j.scitotenv.2013.04.073

Sardar, D., & Kole, R. K. (2005). Metabolism of chlorpyrifos in relation to its effect on the availability of some plant nutrients in soil. *Chemosphere, 61*(9), 1273-1280. doi:10.1016/j.chemosphere.2005.03.078

Saripalli, P. (1997). *Fate and Transport of Organic Chemicals in the Environment: A Practical Guide*: JSTOR.

Schmid, B. H., Stephan, U., & Hengl, M. A. (2005). Sediment deposition in constructed wetland ponds with emergent vegetation: laboratory study and mathematical model. *Water Sci Technol, 51*(9), 307-314. doi:10.2166/wst.2005.0342

Schulz, R., & Peall, S. K. (2001). Effectiveness of a constructed wetland for retention of nonpoint-source pesticide pollution in the Lourens River catchment, South Africa. *Environ Sci Technol, 35*(2), 422-426. doi:10.1021/es0001198

Scow, K. M., & Johnson, C. R. (1996). Effect of sorption on biodegradation of soil pollutants. *Advances in agronomy, 58*, 1-56. doi:https://doi.org/10.1016/S0065-2113(08)60252-7

Sereshti, H., Zarei-Hosseinabadi, M., Soltani, S., Jamshidi, F., & AliAbadi, M. H. S. (2021). Hydrophobic liquid-polymer-based deep eutectic solvent for extraction and multi-residue analysis of pesticides in water samples. *Microchemical Journal, 167*, 106314. doi:ARTN 106314

10.1016/j.microc.2021.106314

Sharifi, A., Kalin, L., Hantush, M. M., Isik, S., & Jordan, T. E. (2013). Carbon dynamics and export from flooded wetlands: A modeling approach. *Ecological Modelling, 263*, 196-210. doi:10.1016/j.ecolmodel.2013.04.023

Shilton, A., & Harrison, J. (2003). Development of guidelines for improved hydraulic design of waste stabilisation ponds. *Water Sci Technol, 48*(2), 173-180. doi:10.2166/wst.2003.0114

Singh, N. C., Dasgupta, T. P., Roberts, E. V., & Mansingh, A. (1991). Dynamics of pesticides in tropical conditions. 1. Kinetic studies of volatilization, hydrolysis, and photolysis of dieldrin and. alpha.-and. beta.-endosulfan. *Journal of agricultural and food chemistry, 39*(3), 575-579. doi:10.1021/jf00003a028

Singh, T., Awasthi, G., & Tiwari, Y. (2021). Recruiting endophytic bacteria of wetland plants to phytoremediate organic pollutants. *International Journal of Environmental Science and Technology*. doi:10.1007/s13762-021-03476-y

Siobhan Fennessy, M., Brueske, C. C., & Mitsch, W. J. (1994). Sediment deposition patterns in restored freshwater wetlands using sediment traps. *Ecological Engineering, 3*(4), 409-428. doi:10.1016/0925-8574(94)00010-7

Smith, K. E., Thullner, M., Wick, L. Y., & Harms, H. (2009). Sorption to humic acids enhances polycyclic aromatic hydrocarbon biodegradation. *Environ Sci Technol, 43*(19), 7205-7211. doi:10.1021/es803661s

Song, B., Jiang, X., Liu, X., Deng, Y., Hu, D., & Lu, P. (2021). Dissipation and sorption-desorption of benzisothiazolinone in agricultural soils and identification of its metabolites. *RSC Adv, 11*(10), 5399-5410. doi:10.1039/d0ra09553b

Sørensen, S. R., Schultz, A., Jacobsen, O. S., & Aamand, J. (2006). Sorption, desorption and mineralisation of the herbicides glyphosate and MCPA in samples from two Danish soil and subsurface profiles. *Environmental Pollution, 141*(1), 184-194. doi:https://doi.org/10.1016/j.envpol.2005.07.023

Sparks, D. (2000). *Kinetics and mechanisms of soil chemical reactions*.

Sposito, G. (2008). *The chemistry of soils*: Oxford university press.

Stang, C., Bakanov, N., & Schulz, R. (2016). Experiments in water-macrophyte systems to uncover the dynamics of pesticide mitigation processes in vegetated surface waters/streams. *Environ Sci Pollut Res Int, 23*(1), 673-682. doi:10.1007/s11356-015-5274-0

Stangroom, S. J., Collins, C. D., & Lester, J. N. (2000). Abiotic behaviour of organic micropollutants in soils and the aquatic environment. A review: II. Transformation. *Environmental technology, 21*(8), 865-882. doi:Doi 10.1080/09593332108618059

Stehle, S., Elsaesser, D., Gregoire, C., Imfeld, G., Niehaus, E., Passeport, E., . . . Schulz, R. (2011). Pesticide risk mitigation by vegetated treatment systems: a meta-analysis. *J Environ Qual, 40*(4), 1068-1080. doi:10.2134/jeq2010.0510

Stottmeister, U., Wiessner, A., Kuschk, P., Kappelmeyer, U., Kastner, M., Bederski, O., . . . Moormann, H. (2003). Effects of plants and microorganisms in constructed wetlands for wastewater treatment. *Biotechnol Adv, 22*(1-2), 93-117. doi:10.1016/j.biotechadv.2003.08.010

Strawn, D., & Sparks, D. (1999). Sorption kinetics of trace elements in soils and soil materials. p. 1–28. HM Selim and KI Iskandar (ed.) Fate and transport of heavy metals in the vadose zone. Lewis Publ., Boca Raton, FL. *Sorption kinetics of trace elements in soils and soil materials. p. 1–28. In HM Selim and KI Iskandar (ed.) Fate and transport of heavy metals in the vadose zone. Lewis Publ., Boca Raton, FL.*, -.

Sudarsan, J. S., & Nithiyanantham, S. (2021). An Integrated Constructed Wetland System for Society (pp. 397-419): Springer International Publishing.

Supowit, S. D., Sadaria, A. M., Reyes, E. J., & Halden, R. U. (2016). Mass Balance of Fipronil and Total Toxicity of Fipronil-Related Compounds in Process Streams during Conventional Wastewater and Wetland Treatment. *Environ Sci Technol, 50*(3), 1519-1526. doi:10.1021/acs.est.5b04516

Sur, R., & Stork, A. (2003). Uptake, translocation and metabolism of imidacloprid in plants. *Bulletin of insectology, 56*, 35-40.

Takagi, K., Fajardo, F. F., Ishizaka, M., Phong, T. K., Watanabe, H., & Boulange, J. (2012). Fate and transport of bensulfuron-methyl and imazosulfuron in paddy fields: experiments and model simulation. *Paddy and water environment, 10*(2), 139-151. doi:10.1007/s10333-011-0276-0

Tang, X., Zhu, B., & Katou, H. (2012). A review of rapid transport of pesticides from sloping farmland to surface waters: Processes and mitigation strategies. *Journal of Environmental Sciences, 24*(3), 351-361. doi:https://doi.org/10.1016/S1001-0742(11)60753-5

Tang, X. Y., Yang, Y., Huang, W. D., McBride, M. B., Guo, J. J., Tao, R., & Dai, Y. V. (2017). Transformation of chlorpyrifos in integrated recirculating constructed wetlands (IRCWs) as revealed by compound-specific stable isotope (CSIA) and microbial community structure analysis. *Bioresource Technology, 233*, 264-270. doi:10.1016/j.biortech.2017.02.077

Tang, X. Y., Yang, Y., McBride, M. B., Tao, R., Dai, Y. N., & Zhang, X. M. (2019). Removal of chlorpyrifos in recirculating vertical flow constructed wetlands with five wetland plant species. *Chemosphere, 216*, 195-202. doi:10.1016/j.chemosphere.2018.10.150

Tejeda-Agredano, M. C., Mayer, P., & Ortega-Calvo, J. J. (2014). The effect of humic acids on biodegradation of polycyclic aromatic hydrocarbons depends on the exposure regime. *Environ Pollut, 184*, 435-442. doi:10.1016/j.envpol.2013.09.031

Thomas, K. A., & Hand, L. H. (2011). Assessing the potential for algae and macrophytes to degrade crop protection products in aquatic ecosystems. *Environ Toxicol Chem, 30*(3), 622-631. doi:10.1002/etc.412

Thorstensen, C. W., Lode, O., Eklo, O. M., & Christiansen, A. (2001). Sorption of bentazone, dichlorprop, MCPA, and propiconazole in reference soils from Norway. *J Environ Qual, 30*(6), 2046-2052. doi:10.2134/jeq2001.2046

Tomlin, C. D. (2009). *The pesticide manual: A world compendium*: British Crop Production Council.

Torabi, E., Wiegert, C., Guyot, B., Vuilleumier, S., & Imfeld, G. (2020). Dissipation of S-metolachlor and butachlor in agricultural soils and responses of bacterial communities: Insights from compound-specific isotope and biomolecular analyses. *J Environ Sci (China), 92*, 163-175. doi:10.1016/j.jes.2020.02.009

Tournebize, J., Chaumont, C., Fesneau, C., Guenne, A., Vincent, B., Garnier, J., & Mander, U. (2015). Long-term nitrate removal in a buffering pond-reservoir system receiving water from an agricultural drained catchment. *Ecological Engineering, 80*, 32-45. doi:10.1016/j.ecoleng.2014.11.051

Tournebize, J., Chaumont, C., & Mander, U. (2017). Implications for constructed wetlands to mitigate nitrate and pesticide pollution in agricultural drained watersheds. *Ecological Engineering, 103*, 415-425. doi:10.1016/j.ecoleng.2016.02.014

Tournebize, J., Passeport, E., Chaumont, C., Fesneau, C., Guenne, A., & Vincent, B. (2013). Pesticide de-contamination of surface waters as a wetland ecosystem service in agricultural landscapes. *Ecological Engineering, 56*, 52-60. doi:10.1016/j.ecoleng.2012.06.001

Tsavdaris, A., Mitchell, S., & Williams, J. (2013). Use of CFD to Model Emergent Vegetation in Detention Ponds. *ARPN Journal of Engineering and Applied Sciences (ISSN 1819-6608), 8*, 495-503.

U.S.G.S. (2020). U.S. Geological Survey. Retrieved from https://www.usgs.gov/

Uchimiya, M., Wartelle, L. H., & Boddu, V. M. (2012). Sorption of triazine and organophosphorus pesticides on soil and biochar. *J Agric Food Chem, 60*(12), 2989-2997. doi:10.1021/jf205110g

Vagi, M. C., Petsas, A. S., Kostopoulou, M. N., & Lekkas, T. D. (2010). Adsorption and desorption processes of the organophosphorus pesticides, dimethoate and fenthion, onto three Greek agricultural soils. *International journal of environmental analytical chemistry, 90*(3-6), 369-389. doi:10.1080/03067310903194980

Vallée, R. (2015). *Efficacité de zones tampons humides à réduire les teneurs en pesticides des eaux de drainage.* Université de Lorraine. Retrieved from https://hal.univ-lorraine.fr/tel-01751701/document

Vallee, R., Dousset, S., & Billet, D. (2016). Influence of substrate water saturation on pesticide dissipation in constructed wetlands. *Environ Sci Pollut Res Int, 23*(1), 109-119. doi:10.1007/s11356-015-4367-0

Vallet, B. S. (2011). *Modélisation d'un bassin d'orage en vue de l'amélioration de la qualité des rivières par la gestion en temps réel.* Université Laval Québec. Retrieved from https://corpus.ulaval.ca/jspui/handle/20.500.11794/22929;jsessionid=9A0144BC40F304793ED67882C7DA165D?mode=full

Vandermaesen, J., Horemans, B., Bers, K., Vandermeeren, P., Herrmann, S., Sekhar, A., . . . Springael, D. (2016). Application of biodegradation in mitigating and remediating pesticide contamination of freshwater resources: state of the art and challenges for optimization. *Appl Microbiol Biotechnol, 100*(17), 7361-7376. doi:10.1007/s00253-016-7709-z

Vieira, I. L., Da Silva, J. B., Ide, C. N., & Janzen, J. G. (2018). Effects of scale and Froude number on the hydraulics of waste stabilization ponds. *Water Sci Technol, 77*(1-2), 239-247. doi:10.2166/wst.2017.529

Villaverde, J., van Beinum, W., Beulke, S., & Brown, C. D. (2009). The kinetics of sorption by retarded diffusion into soil aggregate pores. *Environ Sci Technol, 43*(21), 8227-8232. doi:10.1021/es9015052

Vink, J. P. M., & VanderZee, S. E. A. T. M. (1997). Pesticide biotransformation in surface waters: Multivariate analyses of environmental factors at field sites. *Water Research, 31*(11), 2858-2868. doi:Doi 10.1016/S0043-1354(97)00139-5

Vione, D., Minella, M., Maurino, V., & Minero, C. (2014). Indirect photochemistry in sunlit surface waters: photoinduced production of reactive transient species. *Chemistry, 20*(34), 10590-10606. doi:10.1002/chem.201400413

Vymazal, J. (2013). Emergent plants used in free water surface constructed wetlands: A review. *Ecological Engineering, 61*, 582-592. doi:10.1016/j.ecoleng.2013.06.023

Vymazal, J., & Brezinova, T. (2015). The use of constructed wetlands for removal of pesticides from agricultural runoff and drainage: a review. *Environ Int, 75*, 11-20. doi:10.1016/j.envint.2014.10.026

Waiser, M. J., & Robarts, R. D. (2004). Photodegradation of DOC in a shallow prairie wetland: evidence from seasonal changes in DOC optical properties and chemical characteristics. *Biogeochemistry, 69*(2), 263-284. doi:DOI 10.1023/B:BIOG.0000031048.20050.4e

Walker Jr, W. W., & Kadlec, R. H. (2011). Modeling phosphorus dynamics in Everglades wetlands and stormwater treatment areas. *Critical Reviews in Environmental Science and Technology, 41*(S1), 430-446. doi:https://doi.org/10.1080/10643389.2010.531225

Walse, S. S., Shimizu, K. D., & Ferry, J. L. (2002). Surface-catalyzed transformations of aqueous endosulfan. *Environ Sci Technol, 36*(22), 4846-4853. doi:10.1021/es0256257

Wang, H. Z., Zuo, H. G., Ding, Y. J., Miao, S. S., Jiang, C., & Yang, H. (2014). Biotic and abiotic degradation of pesticide Dufulin in soils. *Environ Sci Pollut Res Int, 21*(6), 4331-4342. doi:10.1007/s11356-013-2380-8

Wang, M., Zhang, D. Q., Dong, J. W., & Tan, S. K. (2018). Application of constructed wetlands for treating agricultural runoff and agro-industrial wastewater: a review. *Hydrobiologia, 805*(1), 1-31. doi:10.1007/s10750-017-3315-z

Wang, Q., & Kelly, B. C. (2017). Occurrence, distribution and bioaccumulation behaviour of hydrophobic organic contaminants in a large-scale constructed wetland in Singapore. *Chemosphere, 183*, 257-265. doi:10.1016/j.chemosphere.2017.05.113

Wang, R., Yuan, Y., Yen, H., Grieneisen, M., Arnold, J., Wang, D., . . . Zhang, M. (2019). A review of pesticide fate and transport simulation at watershed level using SWAT: Current status and research concerns. *Science of the Total Environment, 669*, 512-526. doi:10.1016/j.scitotenv.2019.03.141

Warren, C. S., Mackay, D., Bahadur, N. P., & Boocock, D. G. (2002). A suite of multi-segment fugacity models describing the fate of organic contaminants in aquatic systems: application to the Rihand Reservoir, India. *Water Res, 36*(17), 4341-4355. doi:10.1016/s0043-1354(02)00161-6

Watanabe, H., Takagi, K., & Vu, S. H. (2006). Simulation of mefenacet concentrations in paddy fields by an improved PCPF-1 model. *Pest Manag Sci, 62*(1), 20-29. doi:10.1002/ps.1115

Werner, T. M., & Kadlec, R. H. (2000). Wetland residence time distribution modeling. *Ecological Engineering, 15*(1-2), 77-90. doi:Doi 10.1016/S0925-8574(99)00036-1

Wilkes, G., Edge, T., Gannon, V., Jokinen, C., Lyautey, E., Medeiros, D., . . . Lapen, D. R. (2009). Seasonal relationships among indicator bacteria, pathogenic bacteria, Cryptosporidium oocysts, Giardia cysts, and hydrological indices for surface waters within an agricultural landscape. *Water Research, 43*(8), 2209-2223. doi:10.1016/j.watres.2009.01.033

Willkommen, S., Pfannerstill, M., Guse, B., Ulrich, U., & Fohrer, N. (2018). PondR: a process-oriented model to simulate the hydrology of drainage ponds. *Journal of Hydroinformatics, 20*(1), 149-163. doi:10.2166/hydro.2017.038

Wolfe, N., Zepp, R., & Paris, D. (1978). Carbaryl, propham and chlorpropham: a comparison of the rates of hydrolysis and photolysis with the rate of biolysis. *Water Research, 12*(8), 565-571. doi:https://doi.org/10.1016/0043-1354(78)90134-3

Wu, J., Li, Z., Wu, L., Zhong, F., Cui, N., Dai, Y., & Cheng, S. (2017). Triazophos (TAP) removal in horizontal subsurface flow constructed wetlands (HSCWs) and its accumulation in plants and substrates. *Sci Rep, 7*(1), 5468. doi:10.1038/s41598-017-05874-0

Wu, J., Zhang, N., Hayes, A., Panoutsopoulou, K., & Oliver, S. G. (2004). Global analysis of nutrient control of gene expression in Saccharomyces cerevisiae during growth and starvation. *Proc Natl Acad Sci U S A, 101*(9), 3148-3153. doi:10.1073/pnas.0308321100

Wu, J. G., & Laird, D. A. (2002). Hydrolysis of chlorpyrifos in aqueous and colloidal systems. *Israel journal of chemistry, 42*(1), 99-107. doi:Doi 10.1560/A723-N65t-V12a-1axu

Wu, S. C., & Gschwend, P. M. (1988). Numerical Modeling of Sorption Kinetics of Organic-Compounds to Soil and Sediment Particles. *Water Resources Research, 24*(8), 1373-1383. doi:DOI 10.1029/WR024i008p01373

Yang, M. Y., Lu, M. Z., Bian, H. F., Sheng, L. X., & He, C. G. (2017). Effects of clogging on hydraulic behavior in a vertical-flow constructed wetland system: A modelling approach. *Ecological Engineering, 109*, 41-47. doi:10.1016/j.ecoleng.2017.09.010

Yoshida, T., & Nakano, Y. (2000). Behavior of pesticides in a paddy field with rapid water penetration. *Kagaku Kogaku Ronbunshu, 26*(6), 842-848. doi:DOI 10.1252/kakoronbunshu.26.842

Yu, X., Zhu, H., Yan, B., Xu, Y., Banuelos, G., Shutes, B., . . . Cheng, R. (2019). Removal of chlorpyrifos and its hydrolytic metabolite 3,5,6-trichloro-2-pyridinol in constructed wetland mesocosms under soda saline-alkaline conditions: Effectiveness and influencing factors. *J Hazard Mater, 373*, 67-74. doi:10.1016/j.jhazmat.2019.03.052

Yuan, C. B., Huang, T., Zhao, X. H., & Zhao, Y. Q. (2020). Numerical Models of Subsurface Flow Constructed Wetlands: Review and Future Development. *Sustainability, 12*(8), 3498. doi:ARTN 3498

10.3390/su12083498

Yuh-Shan, H. (2004). Citation review of Lagergren kinetic rate equation on adsorption reactions. *Scientometrics, 59*(1), 171-177. doi:10.1023/B:SCIE.0000013305.99473.cf

Zambonin, C. G., Losito, I., Cilenti, A., & Palmisano, F. (2002). Solid-phase microextraction coupled to gas chromatography-mass spectrometry for the study of soil adsorption coefficients of organophosphorus pesticides. *J Environ Monit, 4*(4), 477-481. doi:10.1039/b201502c

Zamora, S., Marin-Muniz, J. L., Nakase-Rodriguez, C., Fernandez-Lambert, G., & Sandoval, L. (2019). Wastewater Treatment by Constructed Wetland Eco-Technology: Influence of Mineral and Plastic Materials as Filter Media and Tropical Ornamental Plants. *Water, 11*(11), 2344. doi:ARTN 2344

10.3390/w11112344

Zeng, T., & Arnold, W. A. (2013). Pesticide photolysis in prairie potholes: probing photosensitized processes. *Environ Sci Technol, 47*(13), 6735-6745. doi:10.1021/es3030808

Zepp, R. G., Hoigne, J., & Bader, H. (1987). Nitrate-induced photooxidation of trace organic chemicals in water. *Environ Sci Technol, 21*(5), 443-450. doi:10.1021/es00159a004

Zhang, C.-B., Wang, J., Liu, W.-L., Zhu, S.-X., Ge, H.-L., Chang, S. X., . . . Ge, Y. (2010). Effects of plant diversity on microbial biomass and community metabolic profiles in a full-scale constructed wetland. *Ecological Engineering, 36*(1), 62-68. doi:10.1016/j.ecoleng.2009.09.010

Zhang, C., Li, Q., Zhang, L., & Zhang, J. (2021). Responses of submerged macrophytes Vallisneria natans and epiphytic biofilm to floating plants Eichhornia crassipes in eutrophic water. *Water Environ Res, 93*(10), 2237-2249. doi:10.1002/wer.1596

Zhang, L., Lv, T., Zhang, Y., Stein, O. R., Arias, C. A., Brix, H., & Carvalho, P. N. (2017). Effects of constructed wetland design on ibuprofen removal - A mesocosm scale study. *Science of the Total Environment, 609*, 38-45. doi:10.1016/j.scitotenv.2017.07.130

Zhang, P., Ren, C., Sun, H., & Min, L. (2018). Sorption, desorption and degradation of neonicotinoids in four agricultural soils and their effects on soil microorganisms. *Science of the Total Environment, 615*, 59-69. doi:10.1016/j.scitotenv.2017.09.097

Zhang, X., & Zhang, M. (2011). Modeling effectiveness of agricultural BMPs to reduce sediment load and organophosphate pesticides in surface runoff. *Science of the Total Environment, 409*(10), 1949-1958. doi:10.1016/j.scitotenv.2011.02.012

Zhang, Z., Fluet-Chouinard, E., Jensen, K., McDonald, K., Hugelius, G., Gumbricht, T., . . . Poulter, B. (2021). Development of the global dataset of Wetland Area and Dynamics for Methane Modeling (WAD2M). *Earth System Science Data, 13*(5), 2001-2023. doi:10.5194/essd-13-2001-2021

Zheng, H. H., & Ye, C. M. (2001). Identification of UV photoproducts and hydrolysis products of butachlor by mass spectrometry. *Environ Sci Technol, 35*(14), 2889-2895. doi:10.1021/es0016811

Zheng, S., Wang, P., Sun, H., Matsiko, J., Hao, Y., Meng, D., . . . Jiang, G. (2018). Tissue distribution and maternal transfer of persistent organic pollutants in Kentish Plovers (Charadrius alexandrines) from Cangzhou Wetland, Bohai Bay, China. *Science of the Total Environment, 612*, 1105-1113. doi:10.1016/j.scitotenv.2017.08.323

Zhuang, L. L., Yang, T., Zhang, J., & Li, X. (2019). The configuration, purification effect and mechanism of intensified constructed wetland for wastewater treatment from the aspect of nitrogen removal: A review. *Bioresour Technol, 293*, 122086. doi:10.1016/j.biortech.2019.122086

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