Removal of neonicotinoid insecticides in a large-scale constructed wetland system

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Abstract

Neonicotinoid insecticides are among the most used insecticides and their residues are frequently found in surface water due to their persistence and mobility. Neonicotinoid insecticides exhibit toxicity to a wide range of aquatic invertebrates at environmentally relevant levels, and therefore their contamination in surface water is of significant concern. In this study, we investigated the spatiotemporal distribution of six neonicotinoids in a large wetland system, the Prado Wetlands, in Southern California, and further evaluated the wetlands' efficiency at removing these insecticides. Total neonicotinoid concentrations in water ranged from 3.17 to 46.93 ng L^{-1} at different locations within the wetlands, with imidacloprid and dinotefuran among the most detected. Removal was calculated based on concentrations as well as mass fluxes. The concentration-based removal values for a shallow pond (vegetation-free), moderately vegetated cells, densely vegetated cells, and the entire wetland train were 16.9%, 34.2%, 90.2%, and 61.3%, respectively. Principal component analysis revealed that pH and temperature were the primary factors affecting the removal of neonicotinoidss removal. Results from this study demonstrated the ubiquitous presence of neonicotinoids in surface water impacted by urban runoff and wastewater effluent and highlighted the efficiency of wetlands in removing these trace contaminants due to concerted effects of uptake by wetland plants, photolysis, and microbial degradation.

Keywords

Neonicotinoid insecticides; Constructed wetland; Phyto-mitigation; Removal efficiency; Ecological risk

1. Introduction

Surface water is the primary water source for direct human consumption, agriculture, industry, and biodiversity conservation, but is often impaired by contamination of man-made chemicals (Gifford et al., 2018; Kolpin et al., 2002; Shi et al., 2019). Over the past two decades, neonicotinoids, which are broad-spectrum systemic insecticides (Simon-Delso et al., 2015), have been the most used insecticides in both agricultural and urban settings (Jeschke et al., 2011; Simon-Delso et al., 2015; Hladik and Kolpin, 2016; Gould et al., 2018; Douglas and Tooker, 2015; Jeschke et al., 2011; Simon-Delso et al., 2015). As water-soluble compounds, neonicotinoids are highly mobile and have been frequently detected in rivers and streams (Dijk et al., 2013; Hladik et al., 2014; Sánchez-Bayo and Hyne, 2014; Schaafsma et al., 2015; Starner and Goh, 2012). For example, a nationwide study of streams in the United States showed that at least one neonicotinoid compound was present in 63% of the 48 streams surveyed (Hladik and Kolpin, 2016). Neonicotinoids were ~~detected~~ found ubiquitously in all streams draining row-crop areas in the Midwest of the United States (Klarich et al., 2017), with maximal concentrations of 260, 43, and 190 ng L^{-1} for clothianidin, imidacloprid, and thiamethoxam, respectively. Another route for neonicotinoids to contaminate surface water is through wastewater treatment plant (WWTP) effluents (Sadaria et al., 2016), as they are not effectively removed via current WWTP systems (Iancu and Radu, 2018; Sadaria et al., 2016).

Recent studies highlighted the chronic toxicity of neonicotinoids, especially in aquatic invertebrates (Miles et al., 2017; Morrissey et al., 2015; Sánchez-bayo et al.,

2016). The presence of neonicotinoids in ~~S~~s**urface water** ~~exposures to neonicotinoids~~ ~~have~~ has **been** ~~correlated~~ associated **with** observable impacts ~~direct effects~~ **on invertebrates (Dijk et al., 2013; Prosser et al., 2016),** ~~and~~ as well as ~~indirect~~ consequential **effects on** ~~insectivorous~~ insect-eating **birds (Hallmann et al., 2014) and fish (Gibbons et al., 2015).** ~~Studies~~ Research indicated ~~have found~~ **the** ~~occurrence~~ presence **of neonicotinoid insecticides in** surface water within **urban** areas ~~surface water~~ **(Buzby et al., 2020) at** ~~concentrations~~ levels **that** ~~are~~ hold **toxicological** significance~~ly relevant to~~ for **aquatic invertebrates (Tennekes, 2010),** and similarly, ~~as well as~~ **in the sediment, where residues** ~~may persist~~ have the potential to endure **for** ~~extended~~ prolonged **periods** ~~after~~ following **deposition (Kuechle et al., 2019).**

Constructed wetlands (CWs) represent a potential option to remove neonicotinoid residues in surface water. Many studies have demonstrated that CWs can effectively remove nitrogen and phosphorous species (Vymazal, 2007), metals (Lima et al., 2013), antibiotic resistance genes (Du et al., 2022), and various organic compounds (Nguyen et al., 2019; Paz et al., 2019). Given that neonicotinoids are systemic insecticides (Simon-Delso et al., 2015), wetland plants such as macrophytes may be efficient at taking up neonicotinoids. The few studies to date have shown inconclusive results. In a greenhouse study, a variety of wetland plants were found to be capable of removing neonicotinoids when grown in hydroponic containers (Liu et al., 2021). However, in Sadaria et al. (2016), an engineered wetland did not show significant removal of imidacloprid or acetamiprid. In contrast, in Main et al. (2017), the

93 presence of vegetation in prairie wetlands was found to attenuate contamination of
94 clothianidin, and the reduction was attributed to accumulation by wetland
95 macrophytes.
96 As neonicotinoid use and contamination of surface waters continue to grow in
97 both frequency and spatial extent, it is important to evaluate management strategies to
98 reduce neonicotinoid contamination of surface water. Therefore, the main objective of
99 this study was to determine the ability of constructed wetlands to mitigate
100 neonicotinoid water contamination. We specifically aimed to 1) explore the spatial-
101 temporal variations of neonicotinoid insecticides in the Prado Wetlands, a large
102 wetland system receiving both urban runoff and WWTP effluent; 2) assess the removal
103 efficiencies of neonicotinoid insecticides of wetland cells with different vegetation
104 densities; and 3) evaluate the reductions in aquatic toxicity achieved by the wetlands.
105 This study provides information for ascertaining the effectiveness of constructed
106 wetlands in minimizing neonicotinoid contamination in surface flows under field and
107 environmentally relevant conditions.
108

109 **2. Materials and Methods**

110 **2.1 Study area**

111 The Prado Wetlands is a managed constructed wetland system situated near the
112 Prado Dam in Southern California. It is the largest constructed wetland (CW) on the
113 west coast of the United States, covering an area of approximately 188 ha consisting of
114 50 shallow wetland ponds (OCWD, 2019). The primary use of the Prado Wetlands

has been to remove nitrate from the wastewater-impacted Santa Ana River since 1992 (OCWD, 2019). During the dry months, approximately 50% of the Santa Ana River flow, which is dominated by discharge from twelve upstream WWTPs, is directed into the Prado Wetlands system for treatment (OCWD, 2019). During the rest of the year, stormwater runoff and snowmelt account for the majority of the river's flow.

The present study considered different wetland ponds, annotated as BB1, S7-S8, and S9-S10, as shown in Figure 1. BB1 covered 0.770 ha and was essentially absent of any vegetation; S7-S8 was 7.54 ha in size and consisted of two connected wetlands with moderate vegetation density; and S9-S10 was 9.41 ha in size and consisted of two connected wetlands with relatively high vegetation density. BB1 was located in the front section of the whole wetland system, where diverted flow entered the wetlands, while S7-S8 and S9-S10 were vegetated wetland cells located at the heart of the wetland system (Figure 1). From a rhodamine tracer experiment carried out at the Prado Wetlands (Lin et al., 2003), the hydraulic retention time of the entire Prado Wetlands was estimated to be 1.29 days. Samples and measurements were taken at the inlet weir box (inlet) and the outlet weir box (outlet) of BB1, S7-S8, and S9-S10 wetland cells, as well as at the entry (Prado inlet) and W17 exit (Prado outlet) points of the entire wetland systems (Figure 1).

2.2 Chemicals and Materials

All analytical standards used in this study were procured with reported purities ≥ 98 %. Specifically, acetamiprid, clothianidin, dinotefuran, imidacloprid,

thiamethoxam, and thiacloprid standards were purchased from Sigma-Aldrich (Saint Louis, MO). Methanol, acetone, and acetonitrile (HPLC grade) were purchased from Fisher Scientific (Fair Lawn, NJ). Ultrapure water was prepared using an in-house Milli-Q water purification system from Millipore (Carrigtwohill, Cork, Ireland).

2.3 Sample collection and water quality parameters

In order to investigate neonicotinoids removal in the Prado wetlands, a total of 54 surface water samples were collected on a monthly basis from June to November in 2022 at various locations, including Prado inlet, BB1 inlet, BB1 outlet, S7 inlet, S8 outlet, S9 inlet, S10 outlet, and Prado outlet (Figure 1). Grab samples were collected directly into 1-L amber glass bottles, kept at 4°C~~℃~~, and extracted within 24 h after collection. Additionally, plant samples including bulrush shoots (n = 5), bulrush roots (n = 5), duckweed (n = 5), hydrocotyle (n = 4) and sediment samples (n = 11) were collected in wetland cells BB1, S7-S8, and S9-S10. Sediment samples were collected by using a small hand shovel from a surface depth of 0 – 15 cm, and placed in 50 mL centrifuge tubes. Bulrush was collected along with the root, while only the shoot and leaves were collected for hydrocotyle. Duckweed was collected by using a small hand fishing net. All the plant samples were wrapped in foil and stored in a -80°C freezer until analysis. All sediment and plant samples were freeze-dried under vacuum at -60°C for three days before analysis.

The water quality parameters, including temperature (T), pH, electric conductivity (EC), TDS, and dissolved oxygen (DO), were measured *in situ* using a

YSI Pro20 meter (Yellow Spring, OH). Water samples (50 mL) were filtered through 0.45 μm-PTFE filters (ANPEL, Shanghai, China), and the filtrate was used for analysis of nutrients. The concentrations of nitrite (NO_2^--N), nitrate (NO_3^--N), and phosphorus (PO_4^{3-}-P) were measured by using a Dionex Aquion Ion Chromatography (Sunnyvale, CA), along with a Seal AQ2 Discrete Analyzer (Mequon WI) for ammonium (NH_4^+-N). Further information and details are given in Table S1.

2.4 Sample extraction and analysis

2.4.1 Extraction of water samples

A 1.0-L aliquot of water sample was filtered through glass fiber filters (GF/F, 0.7 mm, Whatman, England), followed by the addition of 500 mg $Na_4EDTA{\cdot}2H_2O$. To address the matrix effects, the filtered samples were spiked with surrogate standard. Solid-phase extraction (SPE) was carried out using an Oasis HLB cartridge (500 mg 6mL, Waters) to extract and concentrate neonicotinoid compounds. The cartridges were sequentially activated with 18 mL methanol and 6 mL Milli-Q water. Subsequently, the water samples were loaded onto the cartridges at a flow rate of 5 mL min^{-1}, and the loaded cartridges were then dried under vacuum for approximately 10 min. The sample cartridges were then eluted with 12 mL methanol and 6 mL of acetone: methanol (1:1 v/v), sequentially. The eluate was evaporated to dryness under a gentle stream of nitrogen and reconstituted with 1.0 mL methanol: H_2O (1:1 v/v). The final samples were filtered through a 0.22 μm-PTFE syringe filter into a glass HPLC vial and kept at -20 ℃ before further analysis by LC-MS/MS.

2.4.2 Extraction of sediment and plant samples

The~~Dried~~ freeze-dried **plant tissue samples** underwent a grinding process by a tissue grinder to achieve~~were ground into~~ **a fine**ly **powder**ed ~~using a tissue grinder~~**.** ~~Plant tissue samples were analyzed using a~~A modified **multi-step QuEChERS method (Sigma-Aldrich, n.d.)**~~, modified~~ was employed to extract neonicotinoids from plant tissue samples~~for macrophytes~~**.** In ~~B~~b**riefly~~,~~** a plant tissue sample weighing **1.0 g** ~~of plant tissue sample~~ **was** ~~weighed~~ measured **and** ~~placed~~ introduced **into a 50 mL centrifuge tube**~~,~~. Subsequently, ~~followed by the addition of~~ **20 mL** of **acetonitrile (ACN)** was added, **and** the mixture was **vigorous**ly ~~vortexing~~ vortexed **for** a duration of **1 min.** To this mixture, 4.0 g of anhydrous $MgSO_4$ and 1.0 g of NaCl were added, followed by vortexing for another 1 min, and sonication for 15 min. The sample tubes were centrifuged at 3500 rpm for 15 min, and a 9 mL aliquot of the supernatant was decanted into a 15 mL cleanup tube (Thermo Scientific product number 60105–205; 900 mg $MgSO_4$/400 mg PSA/400 mg GCB). The tubes were then shaken vigorously for approximately 1 min, followed by **centrifugation at 3500 rpm for 15 min. A**~~n aliquot of~~ **6 mL** portion **of the** ~~final~~ ultimate **supernatant was transferred** in**to a test tube and** subjected to ~~evaporated~~ evaporation until completely dry, using~~to dryness~~ ~~under~~ **a gentle stream of nitrogen.** The dried residue was reconstituted using 1.0 mL methanol: H_2O (1:1 v/v) and subjected to sonication for 5 mins. The mixture was then filtered through a 0.22 µm-PTFE filter and transferred to an HPLC vial. The final extracts were stored at -20 ℃ before LC-MS/MS analysis.

2.4.3 Chemical Analysis

Analysis of sample extracts was carried out on a Waters ACQUITY ultra-performance liquid chromatography (UPLC) system coupled to a Waters triple quadrupole mass spectrometer (QqQ-MS/MS) (Waters, Milford, MA). An ACQUITY BEH C18 column (100 × 2.1 mm i.d., 1.7 µm; Waters, Milford, MA) was used for chromatographic separation. The LC conditions for the neonicotinoid analysis were as follows: injection volume, 5 µl; mobile phase flow rate, 0.3 mL min^{-1}; column temperature, 40 ℃; mobile phase A, 0.1% formic acid in Milli-Q water; mobile phase B, 100% methanol. The mobile phase gradient was programmed as follows (with regard to mobile phase B): 10% (0 min), 40% (1.5 min), 50% (4 min), 100% (6 min), 10% (8 min), and 10% B (9 min). The multiple reaction monitoring (MRM) transitions of all target compounds were optimized and are provided in Table S2. Data were processed using the TargetLynx XS software (Waters, Milford, MA).

The working solutions of the six neonicotinoids were prepared by diluting standard mixtures in methanol for UPLC-MS/MS analysis. The quantification of each neonicotinoid was conducted by the external standard method. **For** each sampling batch, and instrumental lank, procedural blank, sample repetition, blank spike, and matrix spike were applied. All instrumental and procedural blanks were below the method detection limits (MDLs).~~the method detection limit (MDL) and recovery, blank samples (Milli-Q water) were spiked with the working mixture solution.~~ **The** blank **recoveries**, matrix recovery, MDLs, method quantification limits (MQLs),

instrumental detection limits (IDLs), and instrumental quantification limits (IQLs) **of the six neonicotinoids** ~~in water, plant, and sediment samples~~ **are shown in Table S3.** The limit of quantification (LOQs) was estimated as a signal-to-noise ratio (S/N) of 10, which was given by TargetLynx XS software (Table S3).

2.5 Environmental risk assessment

The risk quotient (RQ) method was used to evaluate the potential ecological risk of individual neonicotinoids for freshwater species. The RQ values in the water were calculated as follows:

$$RQ = \frac{MEC}{PNEC} \quad (1)$$

where MEC and PNEC were the measured concentrations and predicted no-effect concentrations of neonicotinoids, respectively. The PNEC values for dinotefuran, thiamethoxam, clothianidin, imidacloprid, acetamiprid, and thiacloprid were reported to be 0.953, 0.4, 0.0024, 0.18, 0.1, and 0.017 mg L^{-1}, respectively (Mahai et al., 2019; Zhang et al., 2023). The ecological risks were classified into three levels: low risk, RQ < 0.1; medium risk, 0.1 ≤ RQ < 1; and high risk, RQ > 1 (Zhang et al., 2023).

3. Results and Discussion

3.1 Occurrence of neonicotinoid insecticides at the Prado Wetlands

3.1.1 Spatiotemporal trends of neonicotinoid insecticides in water

Neonicotinoid compounds were frequently detected in water samples collected within the Prado Wetland system, with dinotefuran (87.5%) and imidacloprid (100%) detected at a higher frequency than the other compounds (Figure S1). Figure 2A and Table 1 show the concentrations of the six neonicotinoids in water samples collected from the Prado Wetlands. **The total concentrations of neonicotinoids varied from 3.~~87~~ 17 to ~~55.93~~46.9 ng L^{-1} at different sampling locations within the wetland system.** Compared to earlier studies, the concentrations of neonicotinoid insecticides in the water samples from the Prado Wetlands were relatively low. For example, previous studies reported a maximum total concentration of three neonicotinoids in the Maumee River to be 670 ng L^{-1} (Hladik et al., 2018), a maximum concentration of 0.13 µg L^{-1} of imidacloprid in the Kisco River (Phillips and Bode, 2004), a seasonal average concentration of 198.6 ng L^{-1} of four neonicotinoids (i.e., clothianidin, thiamethoxam, imidacloprid, and acetamiprid) in an intensive agricultural area in central Saskatchewan (Main et al., 2015), and a total neonicotinoid concentration up to 3290 ng L^{-1} in a river in California (Starner and Goh, 2012). The differences in the maximum concentrations between the Prado Wetlands and the surface streams in other areas could be attributed to the surrounding drainage areas, as the Prado Wetlands receive mostly treated wastewater and urban drainage water.

In this study, imidacloprid and dinotefuran were found to be the most prevalent neonicotinoid insecticides, accounting for an average of 54.82 ± 15.22% (10.8~~1~~ ± 5.81 ng L^{-1}) and 39.42 ± 15.41% (9.03 ± 5.67 ng L^{-1}) of the total neonicotinoid

concentrations in water, respectively. Imidacloprid was the most commonly detected, which was consistent with its widespread use in both agricultural (Jeschke et al., 2011) and urban areas (Sánchez-Bayo and Hyne, 2014; Simon-Delso et al., 2015). Previous research estimated that approximately 1.0 - 3.4 tons of imidacloprid was discharged into U.S. surface waters annually (Sadaria et al., 2016). Imidacloprid was the most frequently detected neonicotinoid insecticide in the Great Lakes, USA (Hladik et al., 2018), and concentrations of up to 10,400 ng L^{-1} were reported in Lake Erie and Lake Ontario (Struger et al., 2017). Globally, imidacloprid was detected at up to 4.56 µg L^{-1} in rivers near Sydney (Sánchez-Bayo and Hyne, 2014) and > 0.1 µg L^{-1} in New Brunswick, Canada (Anderson et al., 2015). Imidacloprid has relatively long persistence in aqueous environments, with half-lives of 35.9-230 d in water (Pietrzak et al., 2020). Despite the low recovery of dinotefuran, leading to its exclusion from target list **(Zhang et al., 2017)**, a small amount of~~However,~~ studies have nonetheless reported the detection of **dinotefuran** ~~is usually infrequently detected~~ **in environmental waters. It was reported that only one sample had detectable dinotefuran (1.6 ng L^{-1}) in Sope Creek, GA (Michelle L. Hladik, 2012)** ~~and~~, **the concentrations of dinotefuran ranged from 9.4 - 100 ng L^{-1} in the rivers of Osaka City, Japan (Yamamoto et al., 2012)**, and dinotefuran was the most dominant neonicotinoids (200 ± 296 ng L^{-1}) in Poyang Lake basin **(Xiong et al., 2021).** Dinotefuran has been used in residential and around commercial buildings, in professional turf management (USEPA, 2004), and also as a veterinary medicine for the prevention of fleas and ticks on dogs and cats (USEPA, 2004). The results of this

study were also supported by the annual usage of imidacloprid and dinotefuran in the region; imidacloprid and dinotefuran are the most heavily used neonicotinoids in Riverside, CA, which drains into Santa Ana River that feeds the Prado Wetlands (Table S4). The transport of neonicotinoids to surface streams has been shown to be driven by both use and precipitation, with rainfall events increasing the potential for surface water contamination (Hladik et al., 2014). For example, a previous study suggested that dry weather conditions limited the offsite transport of neonicotinoids to streams (Chiovarou and Siewicki, 2007). In this study, the relatively low concentrations of neonicotinoids observed in the Prado Wetlands as compared to their detections in other studies may be also due to the fact that sampling was carried out during the dry season with little rainfall. To capture the full extent of neonicotinoid contamination in areas with distinct temporal patterns of precipitation, wet season and stormwater runoff monitoring should also be conducted.

During the sampling period, the concentrations of neonicotinoid insecticides in water samples exhibited a clear increasing trend (Figure 3). The total concentration of neonicotinoids at each site increased steadily from June to October and then decreased from October to November. It is likely that the initial rain events in September and October mobilized some of the neonicotinoid residues, leading to their increases, while further rain events in November caused dilution, resulting in decreased concentrations (Table S5) (Hladik et al., 2014). A study of the Maumee River, a tributary of Lake Erie, showed an increase in neonicotinoid concentrations starting in May, with maximum concentrations frequently detected in July (Hladik et al., 2018).

312 Rainfall-runoff was also found to play an important role in the offsite transport of
313 neonicotinoids to streams in Struger et al. (2017), even during peak pesticide
314 applications in summer (Main et al., 2014). Findings from this and earlier studies
315 suggested that the management of neonicotinoid contamination in surface waters
316 should take into consideration the effect of precipitation on their offsite movement,
317 particularly during the rainy season.
318
319 **3.1.2 Spatiotemporal variation of neonicotinoids in sediments and wetland plants**
320 With the exception of imidacloprid, the other five neonicotinoids were below the
321 detection limits in sediment and plant samples collected from the Prado Wetlands.
322 The low occurrence or non-detection of these compounds in sediment and plant
323 samples was consistent with their high water solubility, which would limit their
324 partition into the sediment phase (Zhang et al., 2018). Figure 2B shows the
325 imidacloprid concentrations in sediment and plant samples in the Prado Wetlands.
326 The average imidacloprid concentrations in sediment, bulrush shoot, bulrush root,
327 hydrocotyle, and duckweed were 0.770, 0.760, 0.700, 0.650, and 0.900 ng g^{-1},
328 respectively. The detection of imidacloprid in sediment and plant samples from the
329 Prado Wetlands was likely due to the fact that it was present in the wetland system at
330 higher levels and that imidacloprid is more persistent than the other neonicotinoids
331 (Buzby et al., 2020; Maloney et al., 2017). The general lack of detectable
332 neonicotinoids in the wetland sediments was in line with that reported for the Walnut
333 Creek Watershed in Jasper County (Hladik et al., 2017) and Sacramento and Orange

County, CA (Ensminger et al., 2013), which also showed no or low levels of neonicotinoids in sediments. The lack of detectable systemic uptake of most neonicotinoids by plants may be attributed to the low concentrations of these compounds in the sediment, as well as to the potential effects of growth dilution and/or active metabolism of these insecticides in wetland plants (Hladik et al., 2017). Nevertheless, the finding of imidacloprid in various wetland plants underscored the potential importance of plants in contributing to the removal of neonicotinoids when the contaminated water passes through vegetated wetland systems. Despite of the infrequently detections of neonicotinoids of plants in this study, the bioaccumulation potential in plants cannot be overlooked for neonicotinoid removal. Neonicotinoids, as systemic insecticides, 2% - 20% of them can be accumulated in plant tissues due to the strong inhaling capacity of plants (**Alsafran et al., 2022**). It is usually frequently reported that neonicotinoids are readily accumulated by plant. **Pecenka and Lundgren, (2015)** found that clothianidin concentrations up to 4 μg kg^{-1} in milkweed plant, imidacloprid and thiamethoxam were the most commonly detected neonicotinoids in fruits and vegetables from USCC study and HZC study (**Lu et al., 2018**), **Ge et al, (2017)** found that imidacloprid accumulated in rice leaves and roots with 10 mg kg^{-1} and 1.37 mg kg^{-1} at a soil-treated experiment. Therefore, the bioaccumulation mechanisms of plants regarding neonicotinoids need further research.

3.2 Removal and mass fluxes of neonicotinoids

The concentration-based removal efficiencies of neonicotinoids in water as they passed through the Prado wetland system are given in Figure 4A. The removal factor (RF, in %) was calculated based on the differences in concentrations at the inlet and outlet of the system under consideration:

$$(\%)\ RF = \frac{C_{Inlet} - C_{Outlet}}{C_{Inlet}} \quad (1)$$

where C_{in} and C_{out} are the neonicotinoid concentrations at the inlet and outlet of a wetland system. To estimate the removal factor for the entire Prado Wetland system, concentrations at the Prado inlet and Prado outlet (W17) were used for the calculation. Additionally, it is important to acknowledge that the 100% removal included outlet concentrations that were below the detection limit. Throughout the duration of this study, the average removal efficiencies of the Prado inlet-Prado outlet, BB1, S7-S8, and S9-S10 were 66.59%, 27.61%, 42.65%, and 79.18%, respectively. Among the systems under evaluation, S9-S10 exhibited the highest removal efficiency, followed by Prado inlet – Prado outlet and S7-S8, whereas BB1 displayed the lowest removal values. The lowest removal observed in BB1 could be attributed to its relatively small area (0.770 ha) as well as low vegetation density. In comparison, the higher vegetation density and the relatively large area of S9-S10 likely contributed to the greater removal efficiency. However, the removal efficiency of neonicotinoids for the entire wetland system was not the highest, likely due to the fact that many wetland cells of different configurations and with varying states of vegetation and hydraulic retention times were operated in parallel before the treated water converged and discharged (Figure 1). In

addition, uncertainties caused by spot sampling and the associated flow and sediment resuspension conditions at the time of sampling could also contribute to variations in chemical concentrations and hence the derived removal efficiencies. The generally efficient removal of neonicotinoids through vegetated wetlands was in agreement with previous studies showing that the systemic neonicotinoid insecticides were effectively eliminated from hydroponic planted systems, with removal rates ranging from 9.5% to 99.9% (Liu et al., 2021).

There were no discernible monthly or seasonal patterns observed in the removal of neonicotinoids (Figure S2A). However, the peak removal efficacy was observed in August, which may be due to the relatively elevated temperature during this month, as well as active vegetation growth. The observed variations in removal efficiencies among different wetland cells could be attributed to many factors, including differences in vegetation densities (Dabrowski et al., 2006), hydraulic retention time (Gregoire et al., 2009), and environmental parameters (Main et al., 2017). The upstream Santa Ana River supplies a sufficient amount of nutrients to the wetlands (Bear et al., 2017; Vitko, 1996), which facilitates the establishment and growth of macrophytes that act to take up and metabolize neonicotinoids. Moreover, microbial communities in wetlands in warm regions such as Southern California promote active biotic degradation in the sediment, especially in root zones of wetland plants (Cryder et al., 2021).

In addition to the concentration-based removal, another essential metric for ascertaining the effectiveness of wetlands in attenuating contaminants is the mass flux

399 of chemicals (Figure 4B). In this study, the mass flux of neonicotinoids was calculated
400 using the following equation:
401
402 $$MF = C_{water} * Water\ Flow\ Rate \quad (2)$$
403
404 where MF is the mass flux, C_{water} is the chemical concentration in water, and the
405 water flow rate is estimated by the onsite weir boxes or flumes. It is important to note
406 that the mass flux values obtained were discrete estimates at the time of sampling.
407 Specifically, the mass influx, mass efflux, and changes in mass flux (Δ mass flux) were
408 calculated for the inlet and outlet of the individual wetland systems under
409 consideration. The median Δ mass flux of BB1, S7-S9, and S8-S10 were 137.89,
410 148.70, and 219.36 mg d^{-1}, respectively. Positive changes in mass flux indicate the
411 removal of neonicotinoids in a system, while a negative value would indicate a net
412 export from the system. The majority of Δ mass flux values were statistically
413 significant (*Wilcoxon* test, $P < 0.05$).
414 Positive changes in mass flux values were observed for BB1 (with a median
415 value, of 137.87 mg d^{-1}), S7-S8 (with a median value, of 148.70 mg d^{-1}), and S9-S10
416 (with a median value, of 219.36 mg d^{-1}), which provides further evidence that the
417 wetland cells were effective in removing neonicotinoid insecticides. However, there
418 were significant variations in Δ mass flux values based on specific sampling time
419 points. The 5-95% ranges were 21.700 - 819.39, 0.61000 - 748.85, and 47.780 -

1176.~~66~~ 7 mg d^{-1} for BB1, S7-S8, and S9-S10, respectively. The large variations could be attributed to changes in flow rate and flow-induced resuspension of sediment particles when the flow rate was high. Overall, these findings suggest that wetlands, including both unvegetated and vegetated wetland systems, are effective at removing neonicotinoid insecticide residues from water (Braskerud and Haarstad, 2018; Chiovarou and Siewicki, 2007; Gregoire et al., 2009). Further research is needed to better understand factors contributing to enhanced removal of neonicotinoids from water, such as plant uptake and metabolism, wetland plant species, vegetation density, photolysis, and environmental conditions.

To discern the effect of environmental parameters on the removal of neonicotinoids in the Prado Wetlands, a PCA analysis was conducted. Figure 5 shows a negative correlation between pH and temperature (T) with neonicotinoid levels, suggesting that higher pH and temperature may lead to lower neonicotinoid concentrations. Liang et al. (2019) documented an increase in photo-degradation of all neonicotinoids with increasing pH, and Guzsvány et al. (2006) observed that imidacloprid and thiamethoxam degraded rapidly under alkaline conditions. There was no significant correlation between nutrient levels (i.e., NH_4^+, NO_2^-, NO_3^-, PO_4^{3-}) and neonicotinoid concentrations in water. However, the presence of nutrients could potentially stimulate plant growth and microbial activity, which could subsequently accelerate the removal of neonicotinoids through increased plant uptake and enhanced microbial degradation. The overall findings suggested that many factors worked in concert in influencing the fate of neonicotinoids in a wetland system, such as pesticide

properties (e.g., DT_{50}, K_d), sediment resuspension, and plant uptake, as well as water characteristics (e.g., pH, temperature, conductivity). Aquatic plants may also influence the micro-environment through physical and chemical alterations, such as changing light intensity, pH, and nutrient distribution. Neonicotinoid compounds are highly water soluble and may co-exist with dissolved organic matter in water (Bonmatin et al., 2015), and could undergo indirect photolysis with dissolved organic matter as the photosensitizer (Roy et al., 1999; Zeng and Arnold, 2013). Other researchers also reported the role of photolysis in environmental degradation of neonicotinoids (Lavine et al., 2010; Wamhoff and Schneider, 1999). Photolysis may be especially pronounced in unvegetated wetlands, such as BB1 which was shallow and largely void of vegetation. Nevertheless, it is imperative not to disregard the filtration effects exerted by water and DOM on UV radiation **(Lu et al., 2015)**.

3.3 Neonicotinoid insecticide toxicity and risk assessment

Based on previous studies, contamination of neonicotinoids in rivers can pose ecological risks to aquatic organisms, particularly aquatic animals, resulting in adverse impacts on the biodiversity and overall functions of the aquatic ecosystem (Chen et al., 2019; Naumann et al., 2022). The risk quotient (RQ) was calculated based on the detected concentrations of individual neonicotinoids in the Prado Wetland system during the sampling period (Figure 6A). The monitored neonicotinoids, except for clothianidin, presented a relatively low ecological risk to aquatic ecosystems with RQ < 0.1 (Sánchez-Bayo et al., 2002). The RQs in the Prado Wetlands were comparable to

those in the Huai River (Zhang et al., 2023), the central Yangtze River (Mahai et al., 2019), and the Sousa Rivers (Sousa et al., 2019). For each sampling event, a slight reduction in RQs was observed as water passed through the wetland system, consistent with previous studies (Liu et al., 2021; Main et al., 2017).

The U.S. EPA established acute and chronic toxicity thresholds (i.e., 385 and 10 ng L^{-1}, respectively) for imidacloprid to further safeguard aquatic ecosystems (USEPA, 2017). According to the U.S. EPA aquatic life benchmark, no imidacloprid values detected in this study exceeded the current acute aquatic life benchmarks, but the chronic benchmarks were exceeded 29 times (57% of samples) (Figure 6B). In addition, previous research has demonstrated that neonicotinoid metabolites possess similar levels of toxicity as the parent compounds (Casida, 2011; Suchail et al., 2003; Jeschke et al., 2011). Therefore, it is probable that the overall ecological risks were underestimated in this study by neglecting neonicotinoid metabolites (Bonmatin et al., 2021; Chen et al., 2021; Nomura et al., 2013; Song et al., 2020).

4. Conclusions

This study provides a comprehensive characterization of the spatiotemporal variations and the removal of neonicotinoids in a large wetland system during the dry season in California. The detected neonicotinoid concentrations in the Prado Wetlands were relatively low, with imidacloprid and dinotefuran as the most frequently detected compounds. The changes in neonicotinoid concentrations and mass fluxes highlighted that constructed wetlands were effective at removing neonicotinoid insecticides, likely

due to uptake into wetland plants, photolysis, and microbial degradation. These findings suggest that constructed wetlands may be used as a low-cost efficient option for removing neonicotinoid residues from surface water. Vegetation density and hydraulic retention time were among the main variables for optimizing the removal of neonicotinoids. However, long-term monitoring considering different precipitation conditions and parent compound-metabolite mixtures is necessary to obtain a holistic understanding of wetlands as a mitigation strategy for water contaminated by neonicotinoid insecticides. In addition, the potential release of neonicotinoids sequestered by plants or sediment overtime should also be understood when evaluating the overall functions of wetlands in attenuating man-made chemicals such as neonicotinoid insecticides.

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References

Alsafran, M., Rizwan, M., Usman, K., Saleem, M.H., Jabri, H. Al, 2022. Neonicotinoid insecticides in the environment: A critical review of their distribution, transport, fate, and toxic effects. J. Environ. Chem. Eng. 10, 108485. https://doi.org/10.1016/j.jece.2022.108485

Anderson, J.C., Dubetz, C., Palace, V.P., 2015. Neonicotinoids in the Canadian aquatic environment: A literature review on current use products with a focus on fate, exposure, and biological effects. Sci. Total Environ. 505, 409–422. https://doi.org/10.1016/j.scitotenv.2014.09.090

Bear, S.E., Nguyen, M.T., Jasper, J.T., Nygren, S., Nelson, K.L., Sedlak, D.L., 2017. Removal of nutrients, trace organic contaminants, and bacterial indicator organisms in a demonstration-scale unit process open-water treatment wetland. Ecol. Eng. 109, 76–83. https://doi.org/10.1016/j.ecoleng.2017.09.017

Bonmatin, J., Mitchell, E.A.D., Glauser, G., Lumawig-heitzman, E., Claveria, F., Bijleveld, M., Lexmond, V., Taira, K., Sánchez-bayo, F., 2021. Science of the Total Environment Residues of neonicotinoids in soil , water and people ' s hair : A case study from three agricultural regions of the Philippines. Sci. Total Environ. 757, 143822. https://doi.org/10.1016/j.scitotenv.2020.143822

Bonmatin, J.M., Giorio, C., Girolami, V., Goulson, D., Kreutzweiser, D.P., Krupke, C., Liess, M., Long, E., Marzaro, M., Mitchell, E.A., Noome, D.A., Simon-Delso, N., Tapparo, A., 2015. Environmental fate and exposure; neonicotinoids and fipronil. Environ. Sci. Pollut. Res. 22, 35–67.

https://doi.org/10.1007/s11356-014-3332-7

Braskerud, B.C., Haarstad, K., 2018. Screening the retention of thirteen pesticides in a small constructed wetland 267–274.

Buzby, N., Lin, D., Sutton, R., Buzby, N., Lin, D., Sutton, R., 2020. Neonicotinoids and Their Degradates in San Francisco Bay Water Prepared by : Neonicotinoids and Their Degradates in San Francisco Bay Water.

Casida, J.E., 2011. Neonicotinoid Metabolism : Compounds , Substituents , Pathways , Enzymes , Organisms , and Relevance 2923–2931.

Chen, Qianyu, Zhang, Y., Li, J., Su, G., Chen, Qi, Ding, Z., 2021. Serum concentrations of neonicotinoids , and their associations with lipid molecules of the general residents in Wuxi City , Eastern China. J. Hazard. Mater. 413, 125235. https://doi.org/10.1016/j.jhazmat.2021.125235

Chen, Y., Zang, L., Shen, G., Liu, M., Du, W., Fei, J., Yang, L., Chen, L., Wang, X., Liu, W., Zhao, M., 2019. Resolution of the Ongoing Challenge of Estimating Nonpoint Source Neonicotinoid Pollution in the Yangtze River Basin Using a Modified Mass Balance Approach. Environ. Sci. Technol. 53, 2539–2548. https://doi.org/10.1021/acs.est.8b06096

Chiovarou, E.D., Siewicki, T.C., 2007. Comparison of storm intensity and application timing on modeled transport and fate of six contaminants 9. https://doi.org/10.1016/j.scitotenv.2007.08.029

Cryder, Z., Wolf, D., Carlan, C., Gan, J., 2021. Removal of urban-use insecticides in a large-scale constructed wetland. Environ. Pollut. 268, 115586.

https://doi.org/10.1016/j.envpol.2020.115586

Dabrowski, J.M., Bennett, E.R., Bollen, A., Schulz, R., 2006. Mitigation of azinphos-methyl in a vegetated stream: Comparison of runoff- and spray-drift. Chemosphere 62, 204–212. https://doi.org/10.1016/j.chemosphere.2005.05.021

Dijk, T.C. Van, Staalduinen, M.A. Van, Sluijs, J.P. Van Der, 2013. Macro-Invertebrate Decline in Surface Water Polluted with Imidacloprid 8. https://doi.org/10.1371/journal.pone.0062374

Douglas, M.R., Tooker, J.F., 2015. Large-Scale Deployment of Seed Treatments Has Driven Rapid Increase in Use of Neonicotinoid Insecticides and Preemptive Pest Management in U.S. Field Crops. https://doi.org/10.1021/es506141g

Du, J., Xu, T., Guo, X., Yin, D., 2022. Science of the Total Environment Characteristics and removal of antibiotics and antibiotic resistance genes in a constructed wetland from a drinking water source in the Yangtze River Delta. Sci. Total Environ. 813, 152540. https://doi.org/10.1016/j.scitotenv.2021.152540

Ensminger, M.P., Budd, R., Kelley, K.C., Goh, K.S., 2013. Pesticide occurrence and aquatic benchmark exceedances in urban surface waters and sediments in three urban areas of California, USA, 2008-2011. Environ. Monit. Assess. 185, 3697–3710. https://doi.org/10.1007/s10661-012-2821-8

Ge, J., Cui, K., Yan, H., Li, Y., Chai, Y., Liu, X., Cheng, J., Yu, X., 2017. Uptake and translocation of imidacloprid, thiamethoxam and difenoconazole in rice plants. Environ. Pollut. 226, 479–485. https://doi.org/10.1016/j.envpol.2017.04.043

Gibbons, D., Morrissey, C., Mineau, P., 2015. A review of the direct and indirect

575 effects of neonicotinoids and fipronil on vertebrate wildlife 103–118.
576 https://doi.org/10.1007/s11356-014-3180-5
577 Gifford, M., Chester, M., Hristovski, K., Westerhoff, P., 2018. Human health tradeoffs
578 in wellhead drinking water treatment: Comparing exposure reduction to
579 embedded life cycle risks. Water Res. 128, 246–254.
580 https://doi.org/10.1016/j.watres.2017.10.014
581 Gould, F., Brown, Z.S., Kuzma, J., 2018. Wicked evolution: Can we address the
582 sociobiological dilemma of pesticide resistance? Science (80-.). 360, 728–732.
583 https://doi.org/10.1126/science.aar3780
584 Gregoire, C., Elsaesser, Æ.D., Huguenot, Æ.D., Lange, Æ.J., Ralf, Æ., Gabriela,
585 S.Æ., Tournebize, J., Trevisan, Æ.M., Wanko, Æ.A., 2009. Mitigation of
586 agricultural nonpoint-source pesticide pollution in artificial wetland ecosystems
587 205–231. https://doi.org/10.1007/s10311-008-0167-9
588 Guzsvány, V., Csanádi, J., Gaál, F., 2006. NMR study of the influence of pH on the
589 persistence of some neonicotinoids in water. Acta Chim. Slov. 53, 52–57.
590 Hallmann, C.A., Foppen, R.P.B., Van Turnhout, C.A.M., De Kroon, H., Jongejans,
591 E., 2014. Declines in insectivorous birds are associated with high neonicotinoid
592 concentrations. Nature 511, 341–343. https://doi.org/10.1038/nature13531
593 Hladik, M.L., Bradbury, S., Schulte, L.A., Helmers, M., Witte, C., Kolpin, D.W.,
594 Garrett, J.D., Harris, M., 2017. Neonicotinoid insecticide removal by prairie
595 strips in row-cropped watersheds with historical seed coating use. Agric.
596 Ecosyst. Environ. 241, 160–167. https://doi.org/10.1016/j.agee.2017.03.015

597 Hladik, M.L., Corsi, S.R., Kolpin, D.W., Baldwin, A.K., Blackwell, B.R., Cavallin,
598 J.E., 2018. Year-round presence of neonicotinoid insecticides in tributaries to the
599 Great Lakes, USA. Environ. Pollut. 235, 1022–1029.
600 https://doi.org/10.1016/j.envpol.2018.01.013
601 Hladik, M.L., Kolpin, D.W., 2016. First national-scale reconnaissance of
602 neonicotinoid insecticides in streams across the USA. Environ. Chem. 13, 12–20.
603 https://doi.org/10.1071/EN15061
604 Hladik, Michelle L, Kolpin, D.W., Kuivila, K.M., 2014. Widespread occurrence of
605 neonicotinoid insecticides in streams in a high corn and soybean producing
606 region , USA. Environ. Pollut. 193, 189–196.
607 https://doi.org/10.1016/j.envpol.2014.06.033
608 Hladik, Michelle L., Kolpin, D.W., Kuivila, K.M., 2014. Widespread occurrence of
609 neonicotinoid insecticides in streams in a high corn and soybean producing
610 region, USA. Environ. Pollut. 193, 189–196.
611 https://doi.org/10.1016/j.envpol.2014.06.033
612 Iancu, V.-I., Radu, G.-L., 2018. Analytical Methods Occurrence of neonicotinoids in
613 waste water from the Bucharest treatment plant 2691–2700.
614 https://doi.org/10.1039/c8ay00510a
615 Jeschke, P., Nauen, R., Schindler, M., Elbert, A., 2011. Overview of the status and
616 global strategy for neonicotinoids. J. Agric. Food Chem. 59, 2897–2908.
617 https://doi.org/10.1021/jf101303g
618 Klarich, K.L., Pflug, N.C., DeWald, E.M., Hladik, M.L., Kolpin, D.W., Cwiertny,

D.M., LeFevre, G.H., 2017. Occurrence of neonicotinoid insecticides in finished drinking water and fate during drinking water treatment. Environ. Sci. Technol. Lett. 4, 168–173. https://doi.org/10.1021/acs.estlett.7b00081

Kolpin, D.W., Furlong, E.T., Meyer, M.T., Thurman, E.M., Zaugg, S.D., Barber, L.B., Buxton, H.T., 2002. Pharmaceuticals, Hormones, and Other Organic Wastewater Contaminants in U.S. Streams, 1999−2000: A National Reconnaissance. Environ. Sci. Technol. 36, 1202–1211. https://doi.org/10.1021/es011055j

Kuechle, K.J., Webb, E.B., Mengel, D., Main, A.R., 2019. Factors In-fl-uencing Neonicotinoid Insecticide Concentrations in Floodplain Wetland Sediments across Missouri. https://doi.org/10.1021/acs.est.9b01799

Lavine, B.K., Ding, T., Jacobs, D., 2010. LC-PDA-MS Studies of the Photochemical Degradation of Imidacloprid 2719. https://doi.org/10.1080/00032711003654013

Liang, R., Tang, F., Wang, J., Id, Y.Y., 2019. Photo-degradation dynamics of five neonicotinoids : Bamboo vinegar as a synergistic agent for improved functional duration.

Lima, L.K.S., Pelosi, B.T., Silva, M.G.C., Vieira, M.G.A., 2013. Lead and chromium biosorption by pistia stratiotes biomass. Chem. Eng. Trans. 32, 1045–1050. https://doi.org/10.3303/CET1332175

Lin, A.Y.C., Debroux, J.F., Cunningham, J.A., Reinhard, M., 2003. Comparison of rhodamine WT and bromide in the determination of hydraulic characteristics of constructed wetlands. Ecol. Eng. 20, 75–88. https://doi.org/10.1016/S0925-

8574(03)00005-3

Liu, H., Tang, X., Xu, X., Dai, Y., Zhang, X., Yang, Y., 2021. Chemosphere Potential for phytoremediation of neonicotinoids by nine wetland plants. Chemosphere 283, 131083. https://doi.org/10.1016/j.chemosphere.2021.131083

Lu, C., Chang, C.H., Palmer, C., Zhao, M., Zhang, Q., 2018. Neonicotinoid Residues in Fruits and Vegetables: An Integrated Dietary Exposure Assessment Approach. Environ. Sci. Technol. 52, 3175–3184. https://doi.org/10.1021/acs.est.7b05596

Lu, Z., Challis, J.K., Wong, C.S., 2015. Quantum Yields for Direct Photolysis of Neonicotinoid Insecticides in Water: Implications for Exposure to Nontarget Aquatic Organisms. Environ. Sci. Technol. Lett. 2, 188–192. https://doi.org/10.1021/acs.estlett.5b00136

Mahai, G., Wan, Y., Xia, W., Yang, S., He, Z., 2019. Chemosphere Neonicotinoid insecticides in surface water from the central Yangtze. Chemosphere 229, 452–460. https://doi.org/10.1016/j.chemosphere.2019.05.040

Main, A.R., Fehr, J., Liber, K., Headley, J. V., Peru, K.M., Morrissey, C.A., 2017. Reduction of neonicotinoid insecticide residues in Prairie wetlands by common wetland plants. Sci. Total Environ. 579, 1193–1202. https://doi.org/10.1016/j.scitotenv.2016.11.102

Main, A.R., Headley, J. V., Peru, K.M., Michel, N.L., Cessna, A.J., Morrissey, C.A., 2014. Widespread use and frequent detection of neonicotinoid insecticides in wetlands of Canada's prairie pothole region. PLoS One 9. https://doi.org/10.1371/journal.pone.0092821

663 Main, A.R., Michel, N.L., Headley, J. V., Peru, K.M., Morrissey, C.A., 2015.
664 Ecological and Landscape Drivers of Neonicotinoid Insecticide Detections and
665 Concentrations in Canada's Prairie Wetlands. Environ. Sci. Technol. 49, 8367–
666 8376. https://doi.org/10.1021/acs.est.5b01287
667 Maloney, E.M., Orrissey, C.H.A.M., Eadley, J.O.H.N.V.H., Eru, K.E.M.P., Ibera,
668 K.A.L., 2017. CUMULATIVE TOXICITY OF NEONICOTINOID
669 INSECTICIDE MIXTURES TO CHIRONOMUS DILUTUS UNDER ACUTE
670 EXPOSURE SCENARIOS 36, 3091–3101. https://doi.org/10.1002/etc.3878
671 Michelle L. Hladik, -and D.L.C., 2012. Analysis of the Herbicide Diuron, Three
672 Diuron Degradates, and Six Neonicotinoid Insecticides in Water—Method
673 Details and Application to Two Georgia Streams 1–20.
674 Miles, J.C., Hua, J., Sepulveda, M.S., Krupke, C.H., Hoverman, T., 2017. Effects of
675 clothianidin on aquatic communities : Evaluating the impacts of lethal and
676 sublethal exposure to neonicotinoids 1–24. https://doi.org/10.4231/R7RX992T
677 Morrissey, C.A., Mineau, P., Devries, J.H., Sanchez-bayo, F., Liess, M., Cavallaro,
678 M.C., Liber, K., 2015. Neonicotinoid contamination of global surface waters and
679 associated risk to aquatic invertebrates : A review. Environ. Int. 74, 291–303.
680 https://doi.org/10.1016/j.envint.2014.10.024
681 Naumann, T., Bento, C.P.M., Wittmann, A., Gandrass, J., Tang, J., Zhen, X., Liu, L.,
682 Ebinghaus, R., 2022. Occurrence and ecological risk assessment of
683 neonicotinoids and related insecticides in the Bohai Sea and its surrounding
684 rivers, China. Water Res. 209. https://doi.org/10.1016/j.watres.2021.117912

685 Nguyen, P.M., Afzal, M., Ullah, I., Shahid, N., Baqar, M., 2019. Removal of
686 pharmaceuticals and personal care products using constructed wetlands : effective
687 plant-bacteria synergism may enhance degradation efficiency 21109–21126.
688 Nomura, H., Ueyama, J., Kondo, T., Saito, I., Murata, K., Iwata, T., Wakusawa, S.,
689 Kamijima, M., 2013. Quantitation of neonicotinoid metabolites in human urine
690 using GC-MS thiacloprid nitenpyram. J. Chromatogr. B 941, 109–115.
691 https://doi.org/10.1016/j.jchromb.2013.10.012
692 OCWD, 2019. Facts About OCWD ' s Prado Constructed Wetlands. OCWD'S
693 Newsl. 100.
694 Paz, A. de la, Salinas, N., Matamoros, V., 2019. Unravelling the role of vegetation in
695 the attenuation of contaminants of emerging concern from wetland systems:
696 Preliminary results from column studies. Water Res. 166.
697 https://doi.org/10.1016/j.watres.2019.115031
698 Pecenka, J.R., Lundgren, J.G., 2015. Non-target effects of clothianidin on monarch
699 butterflies. Sci. Nat. 102. https://doi.org/10.1007/s00114-015-1270-y
700 Phillips, P.J., Bode, R.W., 2004. Pesticides in surface water runoff in south-eastern
701 New York State, USA: Seasonal and stormflow effects on concentrations. Pest
702 Manag. Sci. 60, 531–543. https://doi.org/10.1002/ps.879
703 Pietrzak, D., Kania, J., Kmiecik, E., Malina, G., Wątor, K., 2020. Fate of selected
704 neonicotinoid insecticides in soil–water systems: Current state of the art and
705 knowledge gaps. Chemosphere 255.
706 https://doi.org/10.1016/j.chemosphere.2020.126981

Prosser, R.S., Solla, S.R. De, Holman, E.A.M., Osborne, R., Robinson, S.A., Bartlett, A.J., Maisonneuve, F.J., Gillis, P.L., 2016. Sensitivity of the early-life stages of freshwater mollusks to neonicotinoid and butenolide insecticides *. Environ. Pollut. 218, 428–435. https://doi.org/10.1016/j.envpol.2016.07.022

Roy, S., Chandra, B., Viswavidyalaya, K., Kole, R.K., Chandra, B., Viswavidyalaya, K., Chowdhury, A., 1999. Effect of pH and Type of Formulation on the Persistence of lmidacloprid in Water Effect of pH and Type of Formulation on the Persistence of lmidacloprid in Water.

Sadaria, A.M., Supowit, S.D., Halden, R.U., 2016. Mass Balance Assessment for Six Neonicotinoid Insecticides during Conventional Wastewater and Wetland Treatment: Nationwide Reconnaissance in United States Wastewater. Environ. Sci. Technol. 50, 6199–6206. https://doi.org/10.1021/acs.est.6b01032

Sánchez-Bayo, F., Baskaran, S., Kennedy, I.R., 2002. Ecological relative risk (EcoRR): Another approach for risk assessment of pesticides in agriculture. Agric. Ecosyst. Environ. 91, 37–57. https://doi.org/10.1016/S0167-8809(01)00258-4

Sánchez-bayo, F., Goka, K., Hayasaka, D., 2016. Contamination of the Aquatic Environment with Neonicotinoids and its Implication for Ecosystems 4, 1–14. https://doi.org/10.3389/fenvs.2016.00071

Sánchez-Bayo, F., Hyne, R. V., 2014. Detection and analysis of neonicotinoids in river waters - Development of a passive sampler for three commonly used insecticides. Chemosphere 99, 143–151.

https://doi.org/10.1016/j.chemosphere.2013.10.051

Schaafsma, A., Limay-rios, V., Baute, T., Smith, J., Xue, Y., 2015. Neonicotinoid Insecticide Residues in Surface Water and Soil Associated with Commercial Maize (Corn) Fields in Southwestern Ontario 1–21. https://doi.org/10.1371/journal.pone.0118139

Shi, P., Zhang, Y., Song, J., Li, P., Wang, Y., Zhang, Xiaoming, Li, Z., Bi, Z., Zhang, Xin, Qin, Y., Zhu, T., 2019. Response of nitrogen pollution in surface water to land use and social-economic factors in the Weihe River watershed, northwest China. Sustain. Cities Soc. 50, 101658. https://doi.org/10.1016/j.scs.2019.101658

Simon-Delso, N., Amaral-Rogers, V., Belzunces, L.P., Bonmatin, J.M., Chagnon, M., Downs, C., Furlan, L., Gibbons, D.W., Giorio, C., Girolami, V., Goulson, D., Kreutzweiser, D.P., Krupke, C.H., Liess, M., Long, E., Mcfield, M., Mineau, P., Mitchell, E.A., Morrissey, C.A., Noome, D.A., Pisa, L., Settele, J., Stark, J.D., Tapparo, A., Van Dyck, H., Van Praagh, J., Van Der Sluijs, J.P., Whitehorn, P.R., Wiemers, M., 2015. Systemic insecticides (Neonicotinoids and fipronil): Trends, uses, mode of action and metabolites. Environ. Sci. Pollut. Res. 22, 5–34. https://doi.org/10.1007/s11356-014-3470-y

Song, S., Zhang, T., Huang, Y., Zhang, B., Guo, Y., He, Y., Huang, X., Bai, X., Kannan, K., 2020. Urinary Metabolites of Neonicotinoid Insecticides: Levels and Recommendations for Future Biomonitoring Studies in China. https://doi.org/10.1021/acs.est.0c01227

751 Sousa, J.C.G., Ribeiro, A.R., Barbosa, M.O., Ribeiro, C., Tiritan, M.E., Pereira,
752 M.F.R., Silva, A.M.T., 2019. Monitoring of the 17 EU Watch List contaminants
753 of emerging concern in the Ave and the Sousa Rivers. Sci. Total Environ. 649,
754 1083–1095. https://doi.org/10.1016/j.scitotenv.2018.08.309
755 Starner, K., Goh, K.S., 2012. Detections of the neonicotinoid insecticide imidacloprid
756 in surface waters of three agricultural regions of California, USA, 2010-2011.
757 Bull. Environ. Contam. Toxicol. 88, 316–321. https://doi.org/10.1007/s00128-
758 011-0515-5
759 Struger, J., Grabuski, J., Cagampan, S., Sverko, E., McGoldrick, D., Marvin, C.H.,
760 2017. Factors influencing the occurrence and distribution of neonicotinoid
761 insecticides in surface waters of southern Ontario, Canada. Chemosphere 169,
762 516–523. https://doi.org/10.1016/j.chemosphere.2016.11.036
763 Suchail, S.´everine, Debrauwer, L., Belzunces, L.P., 2003. Metabolism of
764 imidacloprid in Apis mellifera 296, 291–296. https://doi.org/10.1002/ps.772
765 Tennekes, H.A., 2010. arthropods is reinforced by exposure time 1–4. https://doi.org/
766 10.1016/j.tox.2010.07.005
767 USEPA, 2004. Pesticide fact sheet: Dinotefuran. Cond. Regist. 1–63.
768 Vitko, T.G., 1996. Seasonal fluctuations in nutrient dynamics and constructed
769 wetlands in Prado Basin, California. Calif. State Univ.
770 Vymazal, J., 2007. Removal of nutrients in various types of constructed wetlands. Sci.
771 Total Environ. 380, 48–65. https://doi.org/10.1016/j.scitotenv.2006.09.014
772 Wamhoff, H., Schneider, V., 1999. Photodegradation of Imidacloprid 3, 1730–1734.

Xiong, J., Tan, B., Ma, X., Li, H., You, J., 2021. Tracing neonicotinoid insecticides and their transformation products from paddy field to receiving waters using polar organic chemical integrative samplers. J. Hazard. Mater. 413, 125421. https://doi.org/10.1016/j.jhazmat.2021.125421

Yamamoto, A., Terao, T., Hisatomi, H., Kawasaki, H., Arakawa, R., 2012. Evaluation of river pollution of neonicotinoids in Osaka City (Japan) by LC/MS with dopant-assisted photoionisation. J. Environ. Monit. 14, 2189–2194. https://doi.org/10.1039/c2em30296a

Zeng, T., Arnold, W.A., 2013. Pesticide Photolysis in Prairie Potholes: Probing Photosensitized Processes.

Zhang, J., Wei, Y., Li, H., Zeng, E.Y., You, J., 2017. Application of Box–Behnken design to optimize multi-sorbent solid phase extraction for trace neonicotinoids in water containing high level of matrix substances. Talanta 170, 392–398. https://doi.org/10.1016/j.talanta.2017.04.031

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

Zhang, X., Cao, Y., Cao, J., Li, Q., Yan, Y., 2023. Occurrence , source , and risk assessment of neonicotinoid insecticides in the Huai River, China. Environ. Pollut. 333, 122068. https://doi.org/10.1016/j.envpol.2023.122068

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Tables:

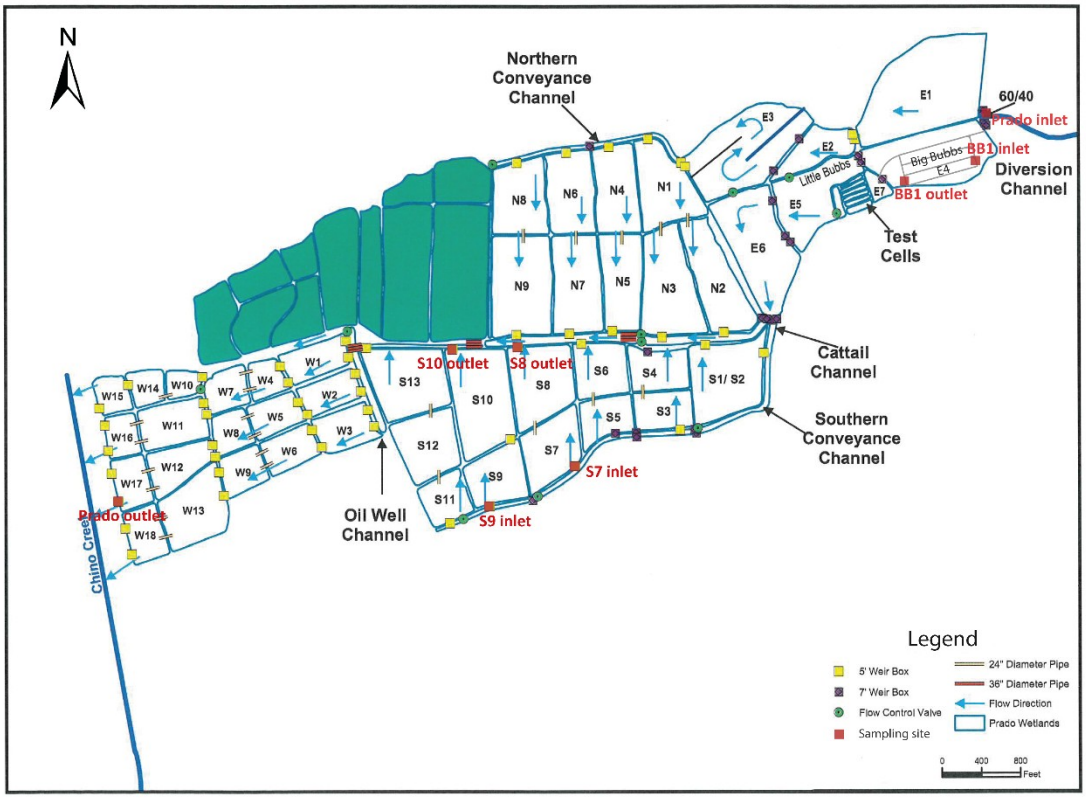
Table 1. Concentrations of six neonicotinoid insecticides of different sampling sites at the Prado Wetlands.

	Dinotefuran	Acetamiprid	Clothianidin	Thiacloprid	Imidaclopri d	Thiamethoxam
Prado in	10.02 ± 6.29	3.49 ± 5.30	1.30 ± 1.39	ND	15.51 ± 4.56	2.87 ± 4.27
BB1 in	9.43 ± 6.15	0.250 ± 0.140	1.01 ± 0.770	ND	13.75 8 ± 4.61	0.300 ± 0.190
BB1 out	8.93 ± 5.94	0.790 ± 0.300	0.760 ± 0.620	ND	12.81 ± 4.37	0.280 ± 0.07000
S7 in	10.89 ± 5.34	0.310 ± 0.290	1.13 ± 0.580	ND	12.48 5 ± 4.55	0.300 ± 0.07000
S8 out	9.25 ± 7.00	0.290 ± 0.240	0.930 ± 0.240	ND	7.08 ± 4.27	0.250 ± 0.07000
S9 in	8.84 ± 6.76	0.250 ± 0.350	0.940 ± 0.740	ND	10.971.0 ± 6.79	0.370 ± 0.260
S10 out	3.96 ± 3.41	0.250 ± 0.430	ND	ND	2.76 ± 3.07	ND
Prado out	9.11 ± 5.03	0.210 ± 0.130	0.770 ± 0.05000	ND	8.04 ± 3.70	0.200 ± 0

ND: Not detected (below detection limit)

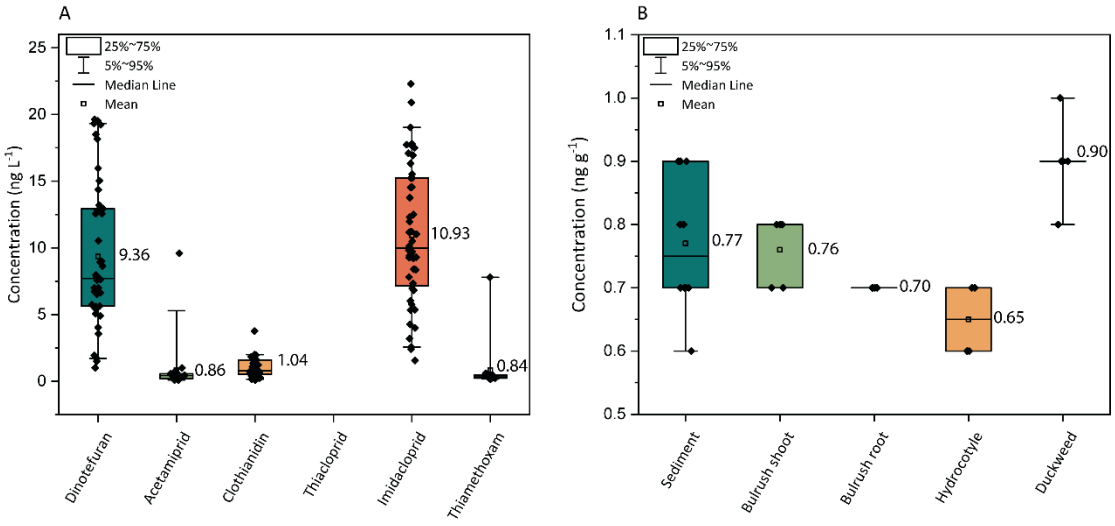
Figures:

Figure 1. Schematic map of the Prado Wetlands in Corona, California. Red squares are sampling points for BB1, S7-S8, and S9-S10 wetland cells, and Prado inlet and Prado outlet of the whole wetland system (Figure credit: Orange County Water District).



811 **Figure 2.** Total concentrations of six neonicotinoids in water samples (**A**);

812 Concentrations of imidacloprid in the sediment and plant tissue samples (**B**).



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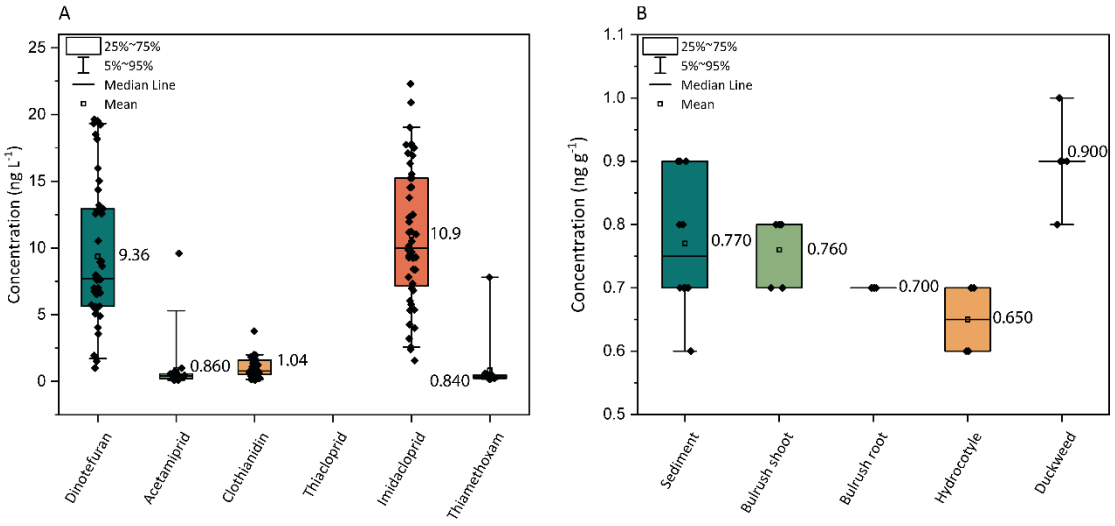

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Figure 3. Temporal distribution and compositions of neonicotinoid insecticides in water samples from S7 inlet and Prado outlet sampling points in the Prado Wetlands.

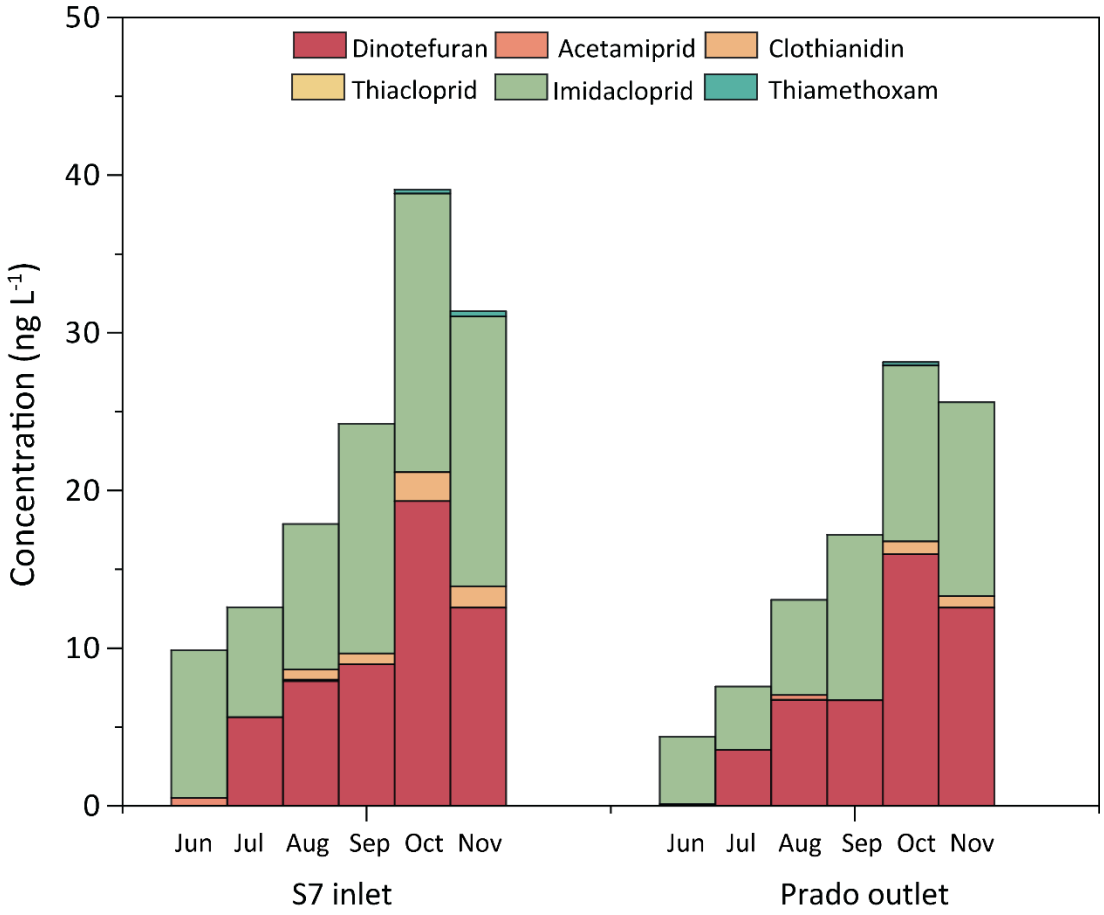
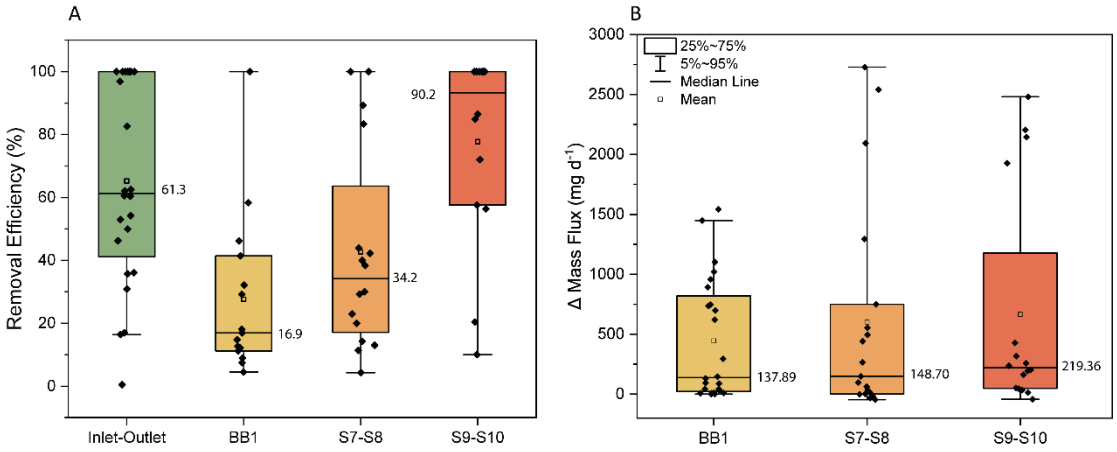


Figure 4. Removal efficiencies (A) and Δ mass flux (B) of six neonicotinoid insecticides in different cells at the Prado Wetlands. ***, $P < 0.001$; *, $P < 0.05$; NS, no significant difference.



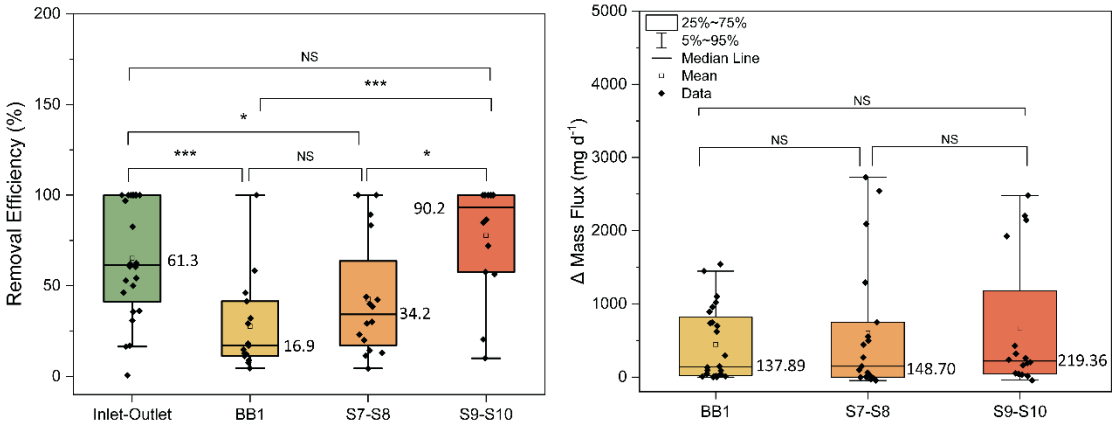
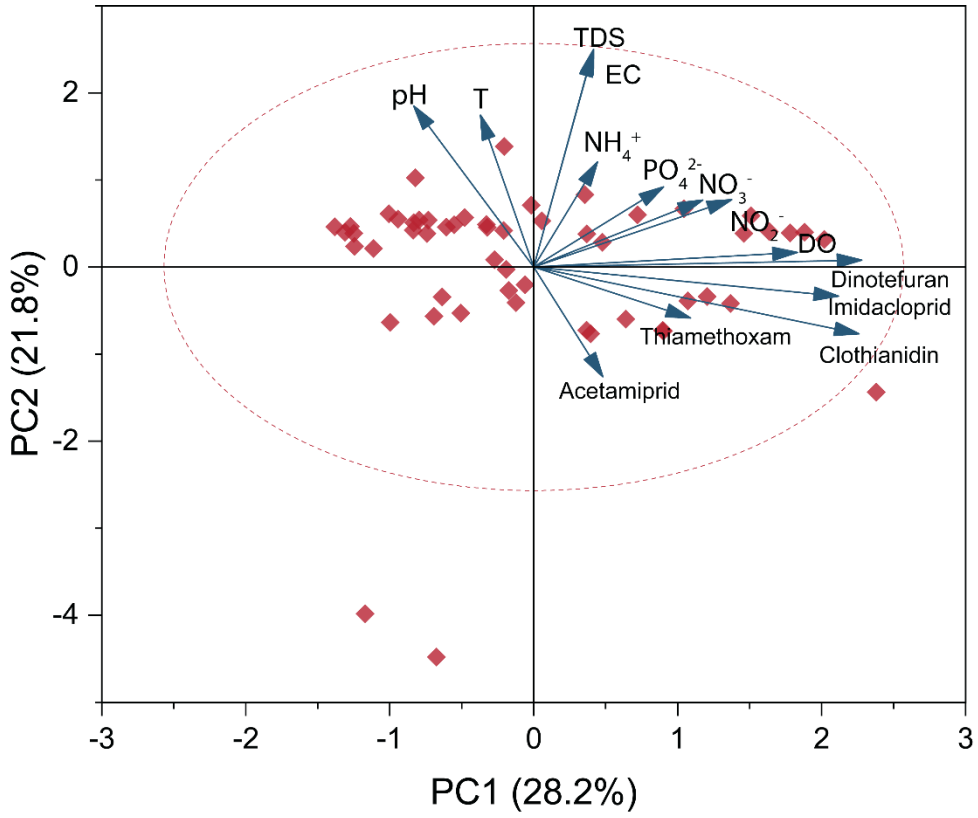
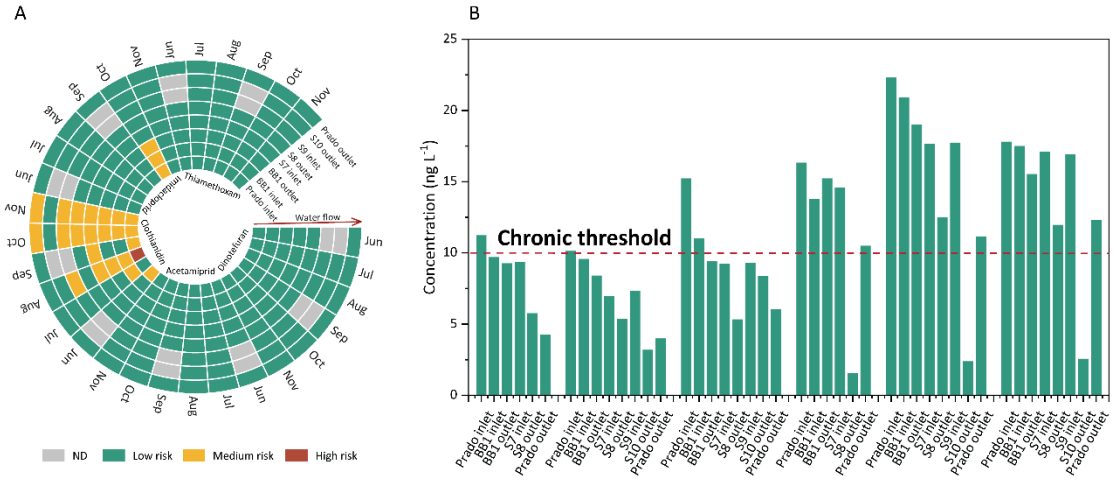


Figure 5. PCA biplots of 14 hydrogeochemical variables for the surface water of the Prado Wetlands. Arrows represent the PC1 and PC2 loading of each variable. The dots signify the PC1 and PC2 scores for each sampling site. The circles characterize the 95% confidence interval.



832 **Figure 6.** The ecological risk quotient of individual neonicotinoid (A); the ecological

833 risk of imidacloprid in the water samples at Prado Wetlands.



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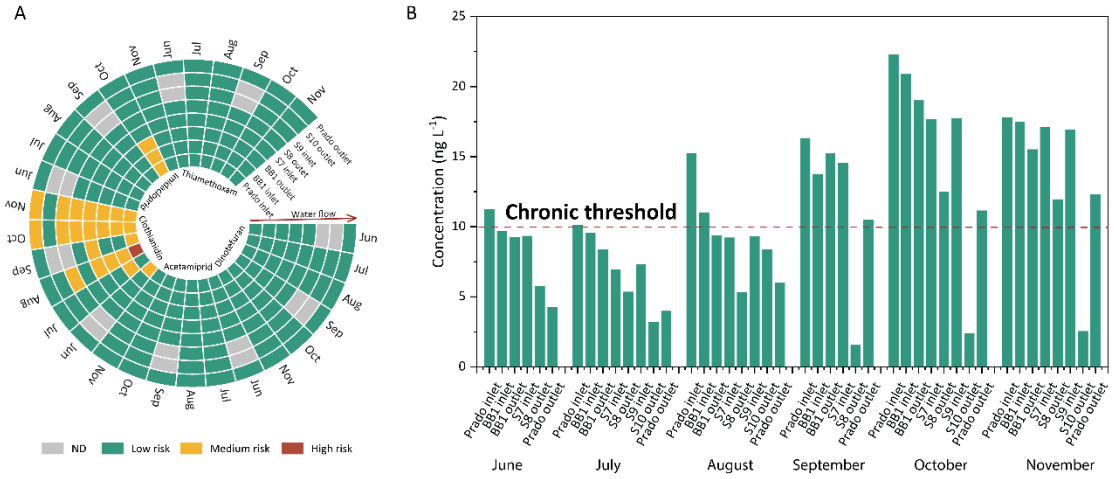

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