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Effects of landuse and precipitation on pesticides and water quality in playa lakes of the southern high plains

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HIGHLIGHTS

- Playa wetlands are important habitats for wildlife.
- Pesticides used on cotton were detected in water samples collected from playas.
- Maximum measured pesticide concentrations exceeded at least one toxicity benchmark.

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ABSTRACT

The 25 000 playa wetlands within the Southern High Plains (SHP) of the United States of America (USA) are the dominant hydrogeomorphic feature in the region, providing habitat for numerous plants and wildlife. The SHP are among the most intensively cultivated regions; there are concerns over the degradation and/or loss of playa wetland habitat. We examined water quality in playa wetlands surrounded by both grassland and agriculture and measured water concentrations of pesticides used on cotton (acephate, trifluralin, malathion, pendimethalin, tribufos, bifenthrin, λ-cyhalothrin, acetamiprid, and thiamethoxam), the dominant crop in the SHP. Pesticides used on cotton were detected in water samples collected from all playas. Precipitation events and the amount of cultivation were related to pesticide concentrations in sediment and water. Our results show that pesticide concentrations were related in some circumstances to time, precipitation, and tilled-index for some but not all pesticides. We further compared measured pesticide concentrations in playas to toxicity benchmarks used by the US EPA in pesticide ecological risk assessments to obtain some insight into the potential for ecological effects. For all pesticides in water, the maximum measured concentrations exceeded at least one toxicity benchmark, while median concentrations did not exceed any benchmarks. This analysis indicates that there is a potential for adverse effects of pesticides to aquatic organisms.

1. Introduction

The Southern High Plains (SHP) of North America is an expansive semi-arid region encompassing approximately 2.6×10^7 hectares that extends into a number of states including Texas, New Mexico, and Oklahoma. Embedded within the SHP are 25 000 playa wetlands, the dominant hydrogeomorphic features in the region (Osterkamp and Wood, 1987). Playas are shallow recharge depressions each existing within their individual watersheds (Smith, 2003). In the semi-arid SHP, playas are the remaining primary sites of biodiversity (Haukos and Smith, 1994) and provide essential habitat for wildlife including migratory bird species (Anderson and Smith, 1998). There is considerable concern regarding the loss of playa wetlands as functional habitats. Factors contributing to the loss of playas include a variety of anthropogenic factors, most notably intensive agriculture of the region.

The SHP is one of the most intensively farmed regions in the Western Hemisphere (Bolen et al., 1989). The dominant crop in the SHP is cotton and indeed the region is one of the foremost cotton production areas in the US. Other crops include wheat, sorghum, corn, and sunflowers (Smith, 2003). As a result of this intensive cultivation, there has been substantial use of agrochemicals throughout the SHP (US Department of Agriculture 2005).

Although rainfall is infrequent in the region, the intensity of rain events often leads to runoff. In addition, crops are heavily irrigated which could facilitate runoff. Agrochemicals have the potential to negatively affect aquatic biota because playas often receive runoff from crop-producing areas in the SHP (Luo et al., 1997). Playas scattered throughout the landscape are also subject to drift of

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pesticides applied by aircraft or from ground-based equipment. Atmospheric deposition may also contribute to pesticide concentrations in playa wetlands (Messing et al., 2011). Moreover, since playas recharge aquifers (Smith, 2003), pesticides found in playas could potentially contaminate groundwater, including the Ogallala Aquifer, the largest aquifer in North America (Zartman et al., 1996).

Given the importance of playas to regional and hemispheric biodiversity and aquifer recharge, our objectives were to examine the water quality of playas (pH, conductivity, total organic carbon, and anion concentrations). At the same time, we determined the presence/absence of pesticides currently used on cotton (acephate, trifluralin, malathion, pendimethalin, tribufos (DEF), bifenthrin, λ-cyhalothrin, acetamiprid, thiamethoxam) and assessed the relationship between pesticide concentration in the playas and precipitation, runoff events, and cultivation intensity in the immediate watershed. We also determined the presence of organochlorine (OC) pesticides (lindane, heptachlor, aldrin, dieldrin, *pp*-DDE, endrin, *pp*-DDT, methoxychlor) in playa sediments.

Few studies however, have examined the presence and persistence of agricultural chemicals in abiotic matrices in the SHP region. General water quality data on playas are also rare. Wetlands in the Yamaska watershed (Canada) were found to contain a variety of pesticides (mostly herbicides including atrazine, metolachlor, and dicamba) at sub-ppb ($\mu g\ L^{-1}$) levels, but only the insecticide chlorpyrifos exceeded the toxicity criteria for aquatic species (Poissant et al., 2008). Venne et al. (2006) investigated metals in SHP playa sediments and amphibian tissues. They concluded metal concentrations did not appear related to current land use and that metal levels were below those known to cause effects in amphibians. Earlier, Thurman et al. (2000) screened playa water samples for pesticides used during the summer and found significant amounts of various pesticides (primarily herbicides) and their metabolites in surface waters. More recently, Belden et al. (2012) surveyed playa sediments in Nebraska, Colorado, Texas, and New Mexico (264 playas total). Results were generally consistent with results from Thurman et al. (2000) in that herbicides were more frequently detected than insecticides and fungicides. Atrazine, metolachlor, trifluralin, and pendimethalin were the most commonly detected pesticides in sediment samples from the SHP (Belden et al., 2012). Also over two decades ago, organochlorine residues were examined in birds inhabiting playas. Those studies often reported high concentrations in the studied taxa (Wallace, 1984; White and Krynitsky, 1986; Flickinger and Krynitsky, 1987).

2. Methods

2.1. Sample collection

Twelve playas that contained water at the beginning of the growing season in 2005 were selected (based on access and proximity) for study in Floyd and Briscoe counties in the Southern High Plains of Texas. The playas within each category (cropland and grassland) were similar in size, depth, and surrounding vegetation. Water was collected weekly in 2005 from all wet playas on the same day from 18 May until 27 July, then bi-weekly until 22 September. Approximately 250 mL of surface water was collected in each quartile of each playa and composited into one 1000 mL sample. Water samples were stored and frozen the day of collection at −20 °C in chemically cleaned polypropylene containers until processed in the laboratory the following day.

Playa sediments were collected at the beginning of the typical wet season in April and after playas had dried in December. Approximately 125 g of sediment were collected from the top 10 cm in each quartile of the circular shaped playa and composited into one 500 g sample for each playa in each sampling period. Sediment samples were stored the day of collection at −20 °C in chemically cleaned glass jars with Teflon-lined lids until processed in the laboratory.

2.2. Water quality

General water quality parameters were measured on playa samples. Conductivity and pH were measured (approximately 24 h after collection) on unfiltered water using conductivity (Accumet Model AB30; Fisher Scientific) and pH (Corning Model 445; Cole–Parmer) probes, respectively. Total organic carbon (TOC) was measured by oxidation using an OI Analytical 1020A TOC analyzer (College Station, TX). Common anions (fluoride, chloride, sulfate, and nitrate) were measured by ion chromatography using a Dionex DX-320 IC system (Sunnyvale, CA).

2.3. Chemicals and reagents

We measured some current-use pesticides in water and some historical-use organochlorine pesticides in sediments. For the current-use pesticides, standards were obtained for acephate, trifluralin, malathion, pendimethalin, tribufos (DEF), bifenthrin and λ-cyhalothrin from AccuStandard, Inc., and for acetamiprid and thiamethoxam from ChemService (West Chester, PA, USA). OC pesticide standards (γ-BHC, DDT, DDE, dieldrin, endrin, aldrin, heptachlor, methoxychlor, TCMX, and DCBP) were purchased from AccuStandard (New Haven, CT, USA). Acetone, hexane, and acetonitrile were purchased from Fisher Scientific (Pittsburg, PA, USA). Dehydrated Na_2SO_4 was purchased from VWR (West Chester, PA, USA). C_{18} solid-phase extraction (SPE) cartridges (500 mg) were obtained from Fisher Scientific (Pittsburg, PA, USA). Ultra-pure water (>18 MΩ) was prepared by ultrafiltration with a Milli-Q® water purification system from Millipore (Bedford, MA, USA). All solvents and standards were analytical or HPLC grade.

2.4. Sample extraction for pesticide determination

2.4.1. Water

Pesticides were extracted from water samples by solid-phase extraction (SPE). One liter of water was passed through a conditioned C_{18} cartridge (500 mg). Pesticides were eluted with 6–10 mL acetonitrile. Then, the extracts were concentrated to 2 mL using a nitrogen evaporator, and filtered through 0.2 μm PTFE membrane filter (Millipore, Bedford, MA, USA) into autosample vials.

2.4.2. Sediments

Sediment samples were dried at room temperature to a constant mass and homogenized. Extraneous organic matter (e.g. plant roots) was removed during grinding and sifting (mesh size = 2 mm). Approximately 20 g of Na_2SO_4 was added and mixed with each sediment sample. All samples were spiked with two internal standards (tetrachloro-*m*-xylene, TCMX and decachlorobiphenyl, DCBP) to determine extraction efficiency. Ottawa sand (Fisher Scientific, Pittsburg, PA) was used as an extraction blank. Sediments were extracted using a Dionex 200 accelerated solvent extractor (Sunnyvale, CA) following a procedure described previously (Zhang et al., 2005; Pan et al., 2006). Briefly, hexane:acetone (1:1) was used as the extraction solvent under the following conditions: pressure = 1400 psi, temperature = 100 °C, extraction time = 14 min. Extracts were rotary evaporated, brought to volume (2 mL), filtered (0.2 μm PTFE), and transferred into autosample vials. The extraction procedure was determined to be quantitative, therefore sediment concentrations of OCs were not adjusted for extraction efficiency.

Table 1
Water quality parameters and select ions in playas from grassland and agricultural areas. Values are presented as geometric means (minimum–maximum).

Playa	Type	tilled index	pH	Conduct[a]. (μS)	TOC (mg L^{-1})	Fl[a](mg L^{-1})	Cl^- (mg L^{-1})	SO_4^{-2} (mg L^{-1})	NO_3^-[a](mg L^{-1})
CP1	Crop	0.9275	7.31	212	78.80	0.24	3.21	1.41	0
CP2	Crop	0.7177	7.38	213	45.14	0.14	7.10	2.79	0.36
CP3	Crop	0.6243	7.15	200	39.20	0.56	3.50	0	0.13
CP4	Crop	1.0000	7.27	325	16.32	1.08	4.80	3.87	0.20
CP5	Crop	0.8907	7.02	158	34.76	0.10	2.42	1.17	0.16
CP6	Crop	0.9132	7.33	373	31.24	1.48	8.03	1.96	0.21
GS1	Grass	−0.2563	7.16	204	74.72	0.20	2.54	0.20	0.10
GS2	Grass	−0.8084	7.00	179	139.00	0.01	2.30	0.03	0
GS3	Grass	−0.6926	7.02	129	120.00	0.01	040	0	0
GS4	Grass	−0.4231	7.40	107	25.00	0.16	2.15	1.10	0
GS5	Grass	−0.4442	7.48	288	0	0.67	7.84	7.74	2.36
GS6	Grass	−0.2603	6.95	163	49.90	0.10	4.26	0	0.05
GS7	Grass	−0.2603	7.08	109	67.00	0.10	4.60	0.50	0.40

[a] Indicates significant effect of tilled index on parameter.

Table 2
Summary statistics for pesticides detected in water samples ($N = 108$) from crop and grassland playas on the Southern High Plains, 2005.

Pesticide	Mean (μg L^{-1})	SD	Median (μg L^{-1})
Trifluralin	7.3	43.8	0.5
Pendimethalin	0.5	1.5	0.1
DEF	0.6	3.4	0.1
Acephate	138	363	0.1
Cyhalothrin	3.3	12.7	0.1
Bifenthrin	3.9	9.1	0.5
Acetamiprid	2.2	7.3	0.1
Thiamethoxam	3.6	21.9	0.1
Malathion	14.3	71.7	0.1

For calculation purposes, 0.1 μg L^{-1} was used for ND.

2.5. Pesticide determination

Although sediment and water extracts were analyzed separately, the analytical method (Zhang et al., 2008) was the same. An HP 6890 Series gas chromatograph (GC) was employed. The GC was equipped with an electron capture detector (ECD) and an autosampler (Agilent, Palo Alto, California, USA) and was controlled by Chemstation® software. A 30 m × 0.32 mm i.d. (0.25 μm film thickness) DB-5 column from J&W Scientific (Folsom, CA, USA) was used to separate the analytes in the extracts. The carrier gas was He (99.99% purity), and was operated at a constant flow-rate during the run (9.2 mL min^{-1}). The makeup gas was argon:methane (95:5) at a combined flow-rate of 60.0 mL min^{-1}. The injection volume was 2 μL in the splitless mode. The injector temperature was 240 °C and the detector temperature was 270 °C. The ECD was operated in the constant current mode. Oven temperature was 110 °C for 1 min, increased to 200 °C at 15 °C min^{-1} with a hold of 2 min, then raised to 269 °C at 15 °C min^{-1} and finally held at 269 °C for 6 min. Residue concentrations were calculated based on matching retention times of sample peaks with those of the calibration standards. A portion of water and sediment samples testing positive for pesticide residues were confirmed by gas chromatography–mass spectrometry in the selected ion monitoring mode.

2.6. Statistical analysis

The values for the tilled-index for the playas used in this analysis were obtained from Tsai et al. (2007) who calculated the amount of tillage in the landscape surrounding playas from:

Tilled-index = [Tilled landscape − Untilled landscape]/[Tilled landscape + Untilled landscape].

The index values ranged from −0.81 to 1.0, and averaged 0.85 in crop playas and −0.48 in grassland playas. Pesticide concentrations were expressed as log(1 + concentration in ng g^{-1}) (Tukey, 1977).

Table 3
Concentration range and frequency of detection for pesticides in water samples from crop and grassland playas on the Southern High Plains, 2005.

Pesticide	Crop playas		Grassland playas	
	Range (μg L^{-1})	% Detection	Range (μg L^{-1})	(%) Detection
Trifluralin	ND – 437	67	ND – 21.8	56
Pendimethalin	ND – 11.3	19	ND – 6.2	18
DEF	ND – 30.9	14	ND – 9.4	10
Acephate	ND – 1545	40	ND – 2121	30
Cyhalothrin	ND – 79.7	31	ND – 27.2	23
Bifenthrin	ND – 59.5	57	ND – 26.0	73
Acetamiprid	ND – 44.1	17	ND – 26.7	4
Thiamethoxam	ND – 20.1	31	ND – 225	25
Malathion	ND – 315	29	ND – 624	15

Table 4
Results of mixed-effects models. Table values are *p*-values from analyses of variance. Only pesticides with significant effects are shown.

Factor	Malathion	Acephate	Cyhalothrin	Bifenthrin	Acetamiprid	DEF
Time	0.410	<0.001	0.006	<0.001	0.733	<0.001
Tilled index	0.422	0.942	0.323	0.458	0.576	0.404
Time × Tilled Index	0.041	0.290	0.021	0.402	0.041	0.160

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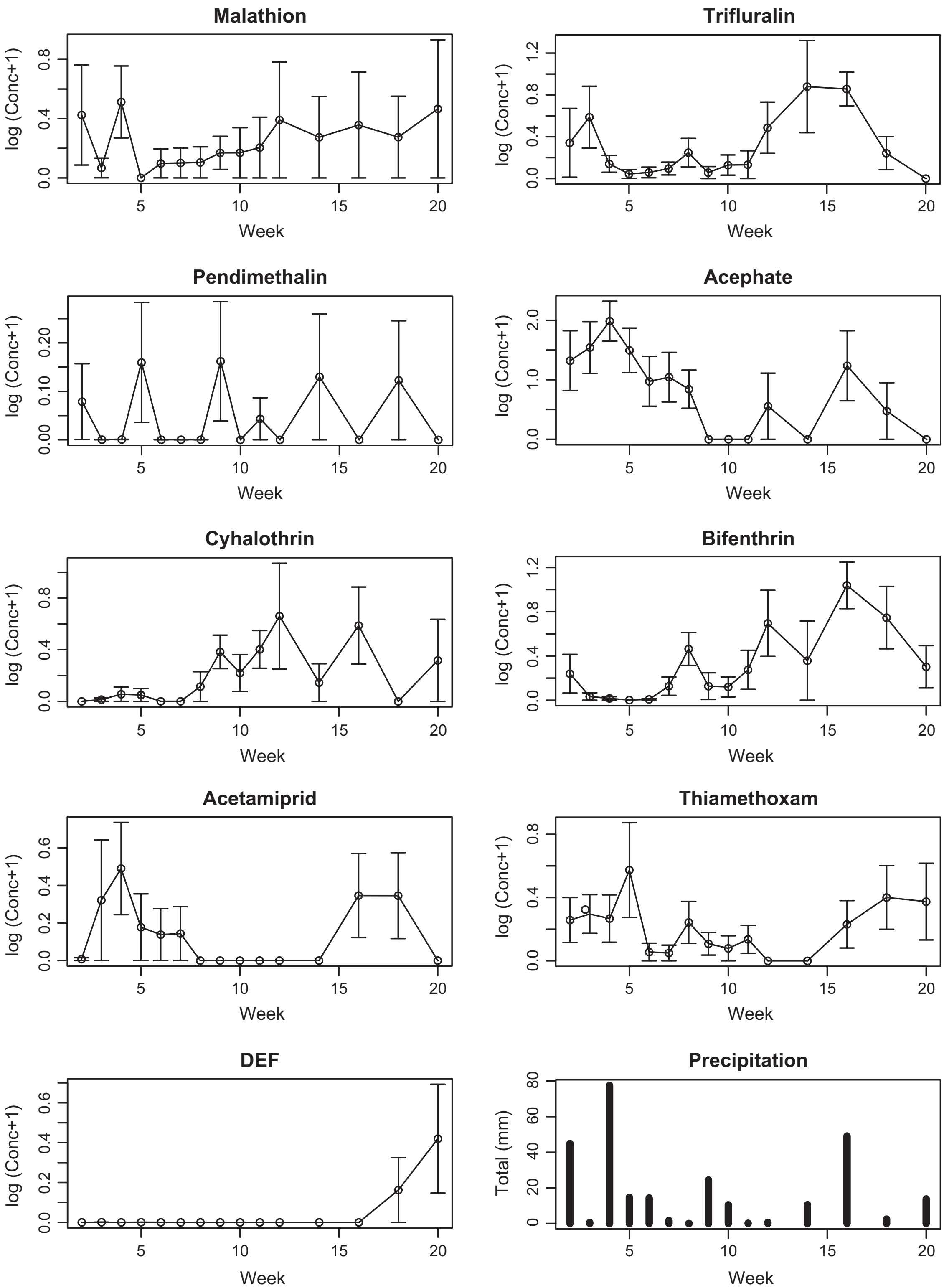

Fig. 1. Changes in pesticide levels over the sampling period. Error bars represent the standard error of the mean pesticide concentration (ng mL^{-1}) across playas. The dashed lines represent the total precipitation collected during the week prior to sample collection (scale not shown).

Linear regression was used to evaluate the relationship between pesticide levels found in sediments and tilled-index. To examine the relationship between pesticide levels found in water and tilled-index, we used a linear mixed-effects model (Pinheiro and Bates, 2000). The mixed-effects model allowed us to explicitly model potential changes in pesticide levels over time (modeled as a random effect) and account for these changes before testing for a relationship with tilled-index (modeled as a fixed effect). Similarly, a mixed-effects model was used to examine the relationship between tilled-index (fixed effect) on water quality parameters and select ions accounting for variation associated with time (random effect). To evaluate the relationship between pesticide levels

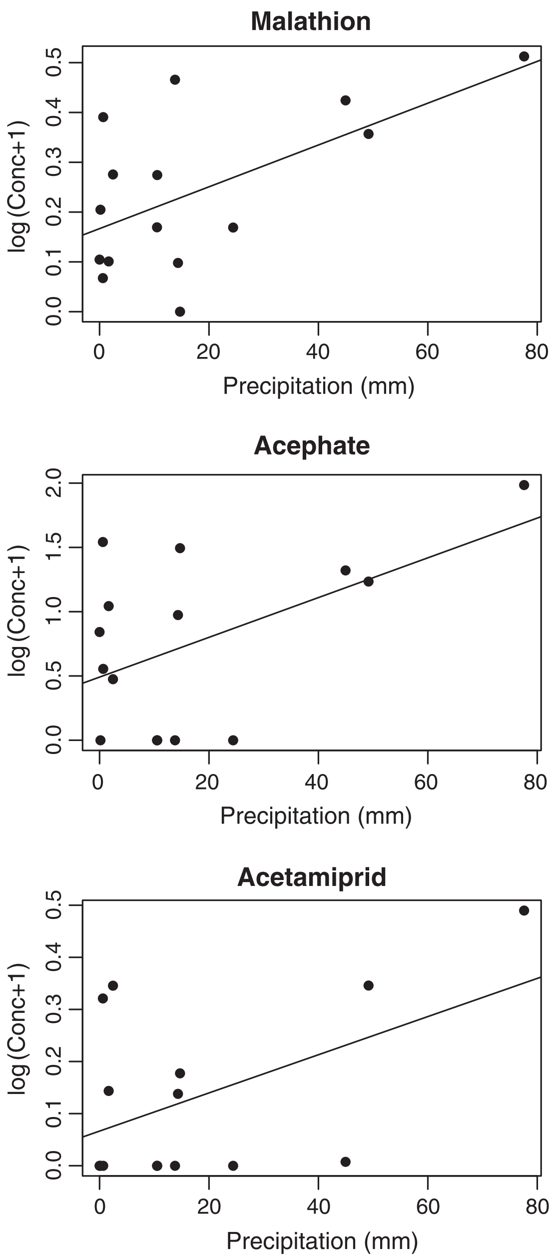

Fig. 2. Relationships between pesticide concentrations (log [concentration + 1]) and precipitation. Precipitation represents the total amount of precipitation collected for the week prior to sample collection. Only pesticides with a linear relationship with precipitation are shown. Slopes, p-values, and r^2 from regression analyses are: Malathion slope = 0.004, $p = 0.019$, $r^2 = 0.36$; Acephate slope = 0.015, $p = 0.046$, $r^2 = 0.27$; Acetamiprid slope = 0.004, $p = 0.061$, $r^2 = 0.24$.

and the amount of precipitation during the previous week, we used linear and quadratic regression. All analyses were conducted using R (R Development Core Team, 2007). Mixed-effects models were evaluated using the NLME library (Pinheiro and Bates, 2000) in R.

3. Results

Geometric mean values for water quality parameters for all playa wetlands used in this study are presented in Table 1. There was a significant effect of tilled-index on conductivity ($p = 0.002$), on fluoride concentration ($p < 0.001$), and on nitrate concentration ($p = 0.012$). Nitrate was higher in crop playas compared to grassland playas (ignoring the contribution from GS-5), while conductivity and fluoride concentration showed the reverse. There was no significant effect of time or rainfall in the preceding interval on water quality parameters. Overall, pH was near neutral (overall mean = 7.20) and conductivity was relatively low (overall mean = 229 μS).

Pesticides were detected in water samples collected from all playas (Tables 2 and 3). Concentrations ranged from below the detection limit for all pesticides in both crop and grassland playas to 2.1 ppm (acephate) in a grassland playa. Bifenthrin (an insecticide) was the most frequently detected pesticide followed by trifluralin (herbicide) and acephate (insecticide). Tribufos (DEF) was detected at quantifiable levels only towards the end of the growing season. Overall, the highest pesticide concentrations were measured in crop playas with the exception of acephate, thiamethoxam and malathion. The pattern for the frequency of detections was similar between crop and grassland playas and was chemical specific (Table 3).

Acephate, bifenthrin, and DEF exhibited a consistent trend over time (Table 4). Namely, acephate decreased over the summer, whereas bifenthrin and DEF increased (Fig. 1). Tilled-index was not significantly related to variation in the majority of pesticides (Table 4). For malathion, λ-cyhalothrin, and acetamiprid, pesticide levels tended to increase with tilled-index. Nevertheless, this pattern was not consistent across time. Malathion, acephate, and acetamiprid levels co-varied with precipitation (Fig. 2).

Residues of OC pesticides were detected in both cropland and grassland playa sediments. With the exception of DDT, concentrations were at sub-ng g^{-1} (ppb) levels. However, DDT concentrations in sediments never exceeded 5 ng g^{-1}. Heptachlor and DDT were the most frequently detected OCs. DDE, a persistent metabolite of DDT, was also present. Differences in concentrations of OCs in sediments within a playa over the 6-month sampling period were slight and within the analytical method error. Concentrations of OC pesticides in sediment samples were not related to tilled-index ($p > 0.10$ for all OCs).

4. Discussion

The primary pesticides detected in playa water were the insecticides acephate, λ-cyhalothrin, bifenthrin, malathion, and the herbicide trifluralin. Although we assumed that these compounds were present because of their use on cotton, some corn and grain sorghum is also grown in the area. In comparison, previous work on playas from the SHP (Thurman et al., 2000) indicated that cotton (or corn) herbicides were more frequently detected (97% of one-time samples from 32 playas) compared to insecticides. The pesticide concentrations we observed were consistently higher than those observed by Thurman et al. (2000) in which mean concentrations of herbicides never exceed 3 ng mL^{-1} (ppb). Thurman et al. (2000) also observed much less frequent detections of insecticides (3%, or 1 sample from 32 playas) compared to the present study, although malathion was the only insecticide monitored in both studies. In cotton, acephate is often used early in the growing season against thrips (Order Thysanoptera). The insecticides, λ-cyhalothrin and bifenthrin are used against bollworms (*Helicoverpa* sp., *Pectinophora* sp.) as cotton matures. In the SHP and elsewhere, malathion was used as part of a boll weevil (*Anthonomus grandis*) control program. These insecticides are applied aerially and by ground equipment to cotton on the SHP, and would be subject to drift by either application method. The detections of tribufos (DEF) only towards the end of the growing season was consistent with its use on cotton as a defoliant.

A recent study by Belden et al. (2012) in the same geographic area found that pesticide occurrence (primarily herbicides such as trifluralin and pendimethalin) and concentrations were higher in playa sediments surrounded by cropland compared to

Table 5
Potential risks to aquatic organisms based on acute and chronic toxicity values (μg L^{-1}) used by The US Environmental Protection Agency for pesticide risk assessments. Bolded values indicate that maximum measured playa concentrations exceeded the toxicity benchmark. No median measured pesticide concentrations exceeded the toxicity benchmarks.

Pesticide	Fish		Invertebrate		Plant		Potential risk: (playa type)[a]
	Acute	Chronic	Acute	Chronic	Nonvascular acute	Vascular acute	
Trifluralin	**20.5**	**1.14**	**280**	**2.4**	**7.52**	43.5	YES (C, G)
Pendimethalin	69	**6.3**	140	14.5	**5.2**	12.5	YES (C, G)
DEF	122.5	**3.5**	**3.4**	**1.56**	148	1100	YES (C, G)
Acephate	416000	5760	**550**	**150**			YES (C, G)
Cyhalothrin	**0.105**	**0.031**	**0.0035**	**0.002**			YES (C, G)
Bifenthrin	**0.075**	**0.04**	**0.8**	**0.0013**			YES (C, G)
Acetamiprid	100000	19200	**21**	**6**			YES (C, G)
Thiamethoxam	>50000	20000	**17.5**				YES (C, G)
Malathion	**16.4**	**8.6**	**0.3**	**0.035**			YES (C, G)

Source of toxicity benchmarks was US EPA Office of Pesticide Programs' Aquatic Life Benchmarks (https://www.epa.gov/oppefed1/ecorisk_ders/aquatic_life_benchmark.htm) and have all been used in ecological risk assessments in support of pesticide registrations under the US Federal Insecticide, Fungicide and Rodenticide Act. Most benchmark values are based on standard toxicity bioassays; specific details can be obtained from the database website and related pesticide risk assessments.
[a] Indicates at least one benchmark was exceeded in a crop playa (C) and/or a grass playa (G).

sediments from playas situated in grasslands. Similarly, we observed an increase in pesticide (malathion, λ-cyhalothrin, and acetamiprid) concentrations in playa water with an increase in tilled-index. Additional data from monitoring of wetlands in the Boreal Plains Ecozone (central Saskatchewan) indicated no strong relationship between surrounding agricultural activity (farming intensity) and herbicide (MCPA, 2,4-D, bromoxynil, dicamba, mecoprop, diclorprop) concentrations in water (Donald et al., 2001).

There are few, if any, data on water quality parameters and anion concentrations in playa wetlands. On the average, the pH of water from both playa types was near neutral; pH was also slightly higher in crop versus grassland playas. We observed slightly elevated conductivity levels in crop playas compared with grassland playas, perhaps related to nutrient runoff, however, overall conductivity values were lower than expected. Anion concentrations were consistently higher in crop playas than in grassland playas, with the exception of nitrate in GS-5. Nitrate levels in GS-5 were elevated throughout the sampling period compared with all other playas. It appears that ranching activities (cattle grazing) can contribute to nutrient input into grassland playas.

In sediment, the OC pesticides detected in this study could simply be residuals from historical use or perhaps atmospheric deposition from recent use in other countries. The presence of relatively high concentrations of DDT to DDE in this study and the lack of a relationship between these OCs and tilled-index supports a more recent input of OCs to the playas from atmospheric deposition. Sediments act as sinks for OCs in the playa environment and these residual compounds in sediments may impact playa biota (Venne et al., 2006).

Playa wetlands represent the most important resource supporting biodiversity in the Southern Great Plains, and most playas are embedded in agriculturally-dominated watersheds subject to intensive use of pesticides. Our results show that herbicides and insecticides occur in playa surface water, and concentrations are related in some circumstances to application period, precipitation, and tilled-index. Our data are supportive of the idea that pesticide input into playas is related to runoff as well as drift. Although use of playas by most wildlife species is seasonal, our monitoring data suggest considerable overlap between periods of peak pesticide runoff into playas and potential playa use by amphibian, avian, and invertebrate organisms (spring and summer). However, surprisingly few studies have examined the potential for exposure and effects to aquatic organisms from the many types of pesticides used in the SHP.

To provide an ecological context to the measured pesticide concentrations in playas, we compiled toxicity benchmarks for fish, invertebrates and plants from the US Environmental Protection Agency's Office of Pesticide Programs (Table 5, US, 2011). The comparison of measured pesticide concentrations to toxicity benchmarks is a standard practice for estimating risk and for identifying a potential for an adverse effects (US, 1998; US, 2004). The US EPA uses upper- and lower- bound estimates of pesticide exposure to better characterize potential risks. We have adopted a similar approach here and found that for maximum pesticide concentrations, there was potential for adverse effects to biota. However, none of the median pesticide concentrations exceeded any toxicity benchmark (Table 5). That said, our analysis and pesticide risk assessments, in general, evaluate pesticides as single, isolated chemicals despite the fact that many pesticides co-occur and risks for co-exposure to multiple pesticides are likely higher (Belden et al., 2007). Hence, for organisms occurring in playas with multiple pesticides, there is likely a greater probability of adverse effect if multiple pesticides are present simultaneously. Such was the case for a risk assessment conducted on Lake Taihu wetland, the third largest freshwater lake in China (Qu et al., 2011). None of the individual pesticides assessed (atrazine, DDT, dichlorvos, dimethoate, lindane, malathion, methyl-parathion, parathion) posed an unacceptable risk. However, as a mixture, the risk was unacceptable to more than 10% of the aquatic species, primarily from the toxicity contributions of dichlorvos, dimethoate, and malathion. Importantly, there are considerable uncertainties in this type of risk assessment approach and care is warranted in interpretation of the results.

Our results also indicate that potential risks are not limited to organisms inhabiting crop playas because at least one toxicity benchmark was exceeded in grass playas as well. This is an important result that indicates pesticide contamination of water bodies is not limited to those water bodies immediately surrounded by agriculture, a common assumption. The SHP are characterized by intensive agriculture and pesticides are frequently applied aerially and hence, playas in the region may represent a relative "worst case" scenario although additional research and analyses are needed before such a conclusion can be reached.

In conclusion, the results and analyses presented here indicate that pesticides are likely to occur in crop and grassland playa wetlands in the SHP. Further, pesticide concentrations may, on occasion, be high enough to elicit adverse effects in aquatic organisms. Additional research and analyses are warranted to better understand and predict the ecological consequences of agrochemical use in the SHP.

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