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Predicting attenuation of salinized surface- and groundwater-resources from legacy energy development in the Prairie Pothole Region

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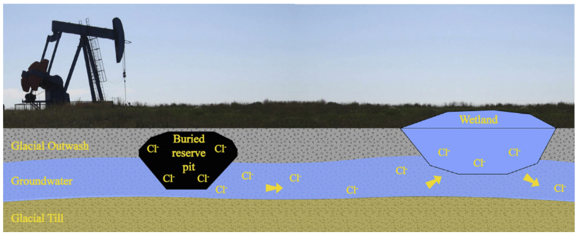
HIGHLIGHTS

- The Prairie Pothole Region overlies part of the oil and gas producing Williston Basin.
- Persistent salinization of aquatic resources occurred from legacy energy development.
- Natural attenuation of chloride to background levels will take another 150–250 years.
- Chloride emitted in 2018 will migrate nearly 1 km until dilution to background levels.
- One third of Prairie Pothole Region wetlands are within 1 km of energy development.

GRAPHICAL ABSTRACT



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ABSTRACT

Oil and gas (energy) development in the Williston Basin, which partly underlies the Prairie Pothole Region in central North America, has helped meet U.S. energy demand for decades. Historical handling and disposal practices of saline wastewater co-produced during energy development resulted in salinization of surface and groundwater at numerous legacy energy sites. Thirty years of monitoring (1988–2018) at Goose Lake, which has been producing since the 1960s, documents long-term spatial and temporal changes in water quality from legacy energy development. Surface water quality was highly variable and decoupled from changes in groundwater quality, likely due to annual and regional climatic fluctuations. Therefore, changes in surface water-quality were not considered a reliable indicator of subsurface chloride migration. However, chloride concentrations in monitoring wells near wastewater sources exhibited systematic temporal reductions allowing for estimates of the time required for natural attenuation of groundwater to U.S. Environmental Protection Agency acute and chronic chloride toxicity benchmarks and a local background level. Point attenuation rates differed based on sediment type (outwash vs till) and yielded a range of predicted years when water-quality targets will be reached: acute – 2045 to 2113; chronic – 2069 to 2160; background – 2126 to 2275. Bulk attenuation rates from four separate years of data were used to calculate the distances chloride could migrate downgradient from the largest wastewater source. Potential distances of downgradient migration before dilution to water-quality targets decreased

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from 1989 to 2018: acute – 949 to 673 m; chronic – 1220 to 922 m; background – 1878 to 1525 m. Several downgradient wetlands are within these distances and will continue to receive saline contaminated groundwater for years. While these results demonstrate chloride attenuation at a legacy energy site, they also highlight the persistence of saline wastewater contamination and the need to mitigate future spills to prevent long-term salinization from energy development.

1. Introduction

Saline water (often referred to as brine) is commonly co-produced with oil and gas (energy) resources and the release of oilfield wastewaters results in the secondary salinization of soils (Keiffer and Ungar, 2002; Pichtel, 2016), surface water (Brantley et al., 2014; Cozzarelli et al., 2017; Vengosh et al., 2014), and groundwater (Fontenot et al., 2013; Jackson et al., 2013; Preston et al., 2014). In addition to salts, oilfield wastewaters include toxic and radioactive elements from the producing formation and chemical additives used during drilling and production (Engle et al., 2014; Lauer et al., 2016). The rapid increase in domestic U.S. energy production from unconventional resources has been accompanied by a concurrent increase in the volume of produced water, with the U.S. generating over 21 billion barrels of wastewater annually (Veil, 2015). Environmental effects of produced water spills have been studied at numerous energy plays (e.g., Bakken Formation, North Dakota (Cozzarelli et al., 2017); Barnett Formation, Texas (Fontenot et al., 2013); Marcellus Formation, Pennsylvania (Jackson et al., 2013)); however, differences in wastewater chemistry, contaminant pathways, climate, and physiography exert considerable influence on the fate and transport of oilfield wastewaters and the associated risks to natural resources and local organisms (Cozzarelli et al., 2017). Therefore, understanding the long-term effects of secondary salinization from legacy energy production within an individual energy play is critical to developing effective remediation strategies for current and future discharges of oil and gas wastewaters.

The Williston Basin has been a leading source of U.S. energy production since the 1960s. Overlying the northeastern portion of the Williston Basin is the Prairie Pothole Region (PPR), a glacial till plain characterized by numerous depressional wetlands within a grassland matrix that provides critical habitat to migratory waterfowl (Batt et al., 1989), grassland birds (Swengel and Swengel, 1998), aquatic and terrestrial invertebrates (Euliss et al., 1999), amphibians (Mushet et al., 2012), and other wetland-dependent wildlife. Secondary salinization from legacy and contemporary energy production is well documented within the PPR (Cozzarelli et al., 2017; Lauer et al., 2016; Preston et al., 2014; Thamke and Smith, 2014; Reiten and Tischmak, 1993; Rouse et al., 2013) and has resulted in persistent increases in surface and groundwater salinity (Beal et al., 1987; Preston et al., 2014; Rouse et al., 2013), decreased biodiversity and ecosystem services in PPR wetlands (Hossack et al., 2018, 2017; Preston et al., 2018; Preston and Ray, 2017; Smalling et al., 2019), and reduced seed germination of pasture and agricultural crops (Keiffer and Ungar, 2002; Meehan et al., 2017; Munn and Stewart, 1989).

A major source of saline contamination in the Williston Basin is from legacy practices that used unlined earthen reserve or evaporation pits to store and dispose of co-produced water (Beal et al., 1987; Lauer et al., 2016; Murphy et al., 1988; Preston et al., 2014; Reiten and Tischmak, 1993). Reserve pits were used temporarily (months to years) at individual production well sites until more permanent infrastructure was constructed. Evaporation pits were generally located at collection facilities, serviced multiple production wells, and operated for extended periods (years to decades) to dispose of wastewater through evaporation and infiltration. Pits were reclaimed by burial after use, and it is estimated that the average reserve pit held approximately 260 tons of dissolved sodium chloride salts during operation (Reiten and Tischmak, 1993). Of particular concern are numerous legacy pits constructed in permeable glacial outwash deposits, which allowed for rapid infiltration of salts and other contaminants into the shallow groundwater system. The high hydraulic conductivity of these deposits promoted the development of laterally extensive groundwater plumes that could discharge into downgradient wetlands (Baker and Brendecke, 1983; Beal et al., 1987; Preston and Chesley-Preston, 2015; Preston et al., 2014; Reiten and Tischmak, 1993; Rouse et al., 2013). More than 20,500 oil and gas wells were drilled in the Williston Basin prior to pit regulations in the 1980s and 1990s (Beal et al., 1987; Preston et al., 2014). These abandoned pits represent potential point source locations of saline wastewater. Pipeline breaks, spills, illegal discharges, and reuse applications, such as road deicers, are also contemporary sources of saline contamination.

Much of the surface and shallow groundwater in the PPR has naturally elevated total dissolved solid (TDS) concentrations; however, the chemistry of these naturally saline waters is distinctly different from the deep formational groundwater in the production zones (oilfield wastewater). The salinity of surface waters in the PPR generally increases along the continuum of groundwater recharge, flow-through, and discharge wetlands (LaBaugh et al., 1987; Winter, 2003), yet TDS concentrations are generally <10,000 mg/L (Tangen et al., 2014). TDS concentrations in the shallow aquifers are generally below 3000 mg/L (Gorham et al., 1983). Wetlands unaffected by oilfield wastewaters in the PPR are predominantly sulfate (62% of wetlands) or magnesium (33%) type waters with relatively few wetlands containing water dominated by chloride (5%; Swanson et al., 1988). In contrast, deep formational waters in the Williston Basin are some of the most saline waters in the U.S. and have TDS concentrations ranging from 30,000 to >450,000 mg/L, and ionic compositions are typically dominated by chloride (Iampen and Rostron, 2000; Otton, 2006). The introduction of oilfield wastewaters to uncontaminated waters can increase TDS and specific conductance and shift sulfate/bicarbonate-dominated waters to chloride-dominated waters (Preston et al., 2014; Reiten and Tischmak, 1993).

Ecological manifestations of wastewater contamination in the PPR are numerous and span multiple trophic levels. Contamination from oilfield wastewaters reduced the above-ground biomass of hardstem bulrush (*Schoenoplectus acutus*) and taxonomic richness and diversity of macroinvertebrates in PPR wetlands (Preston et al., 2018; Preston and Ray, 2017). Boreal chorus frog (*Pseudacris maculata*) larvae reared from contaminated sediments had reduced survival rates relative to larvae reared from uncontaminated sediments (Hossack et al., 2017) and contaminated wetlands in the PPR had lower abundances of boreal chorus frogs, northern leopard frogs (*Rana* [= *Lithobates*] *pipiens*), and barred tiger salamanders (*Ambystoma mavortium*) compared to uncontaminated wetlands (Hossack et al., 2018). Finally, survival of larval fathead minnows (*Pimephales promelas*) was reduced in a North Dakota stream over a 6-month period after a wastewater spill (Cozzarelli et al., 2017). Combined, these studies imply that the effects of wastewater contamination may be synergistic responses to salt toxicity and other associated contaminants, such as metals and radionuclides, commonly found in oilfield wastewater (Farag and Harper, 2014).

The Williston Basin contains tens of thousands of oilfield sites with buried, unlined reserve pits and examining the processes of secondary salinization at a long-term monitoring site provides a better understanding of the likely fate and transport of contaminants at other legacy sites in the basin and provides critical knowledge to aid future wastewater management and restoration efforts. The Goose Lake study site, located in the PPR of the Williston Basin, has a well-documented history of secondary salinization from oil and gas development (Preston et al., 2014; Peterman et al., 2012; Reiten and Tischmak, 1993; Rouse et al., 2013). Historical data and recent water-quality sampling in 2015 and 2018 provide a 30-year time series to examine long-term spatial and temporal changes in surface- and groundwater-quality at a legacy energy development site. Previous studies identified contaminant sources and likely contamination pathways and began to explore changes in surface and groundwater-quality. Recently collected data allowed for enhanced understanding of the temporal (long-term and seasonal) changes in the geochemical signatures of wastewater-contaminated surface and groundwater resources. Specifically, our main objectives were to evaluate the influences of regional climatic fluctuations on the observed relations between surface and groundwater salinity and to provide the first known quantitative predictions on the timeframe required for natural attenuation of chloride in shallow groundwater systems adjacent to known wastewater sources in the Williston Basin.

2. Methods

2.1. Site description

The Goose Lake study site (Goose Lake hereafter) in Sheridan County, northeast Montana, USA, includes the Rabenberg Waterfowl Production Area and surrounding private lands (Fig. 1; Reiten and Tischmak, 1993). Potential wastewater sources to aquatic resources at Goose Lake include reserve pits at 11 oilfield sites, targeting the Mississippian-aged Madison Limestone, installed during 1965–1969 and an evaporation pit at the tank battery. Based on historical aerial photos, the individual reserve pits were generally active for a few years after well construction, whereas the evaporation pit at the tank battery was in operation from at least 1967 through 1992. Other potential wastewater sources include spills and intentional discharges onto the land surface and into nearby wetlands as reported by local

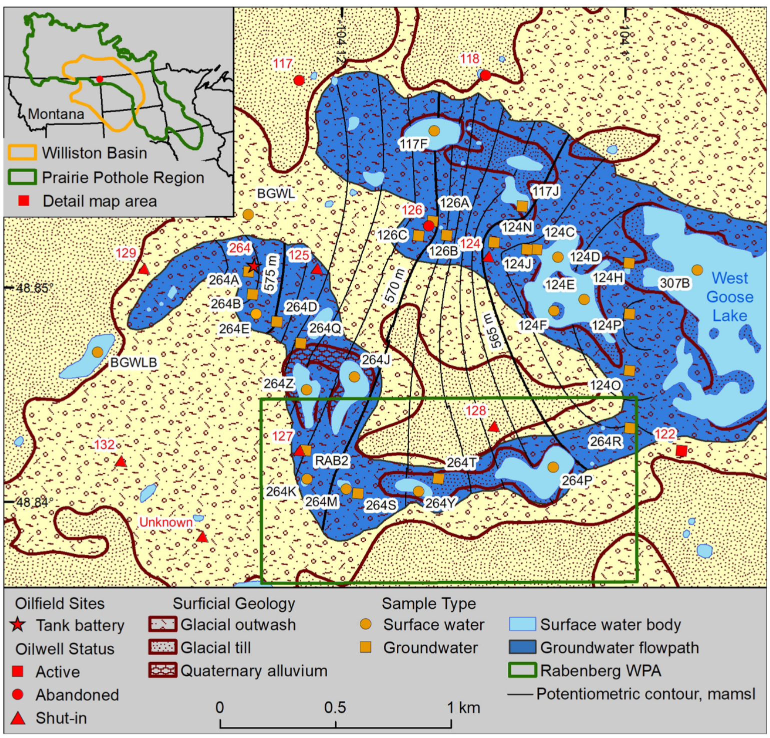


Fig. 1. Map of the Goose Lake study site (Sheridan County, Montana, USA) showing locations of the Rabenberg Waterfowl Production area (WPA), oilfield infrastructure, surface (wetland) and groundwater samples, surficial geology (Reiten and Tischmak, 1993), the northern and southern flowpaths and the associated potentiometric contours (altitude at which water level would have stood in tightly cased wells, May, 2009. Contour interval 1 m, Datum is mamsl – meters above mean sea level). Numbers next to oilfield infrastructure and water quality sampling locations refer to site names in the text. Surface water sites BGWL and BGWL-B are hydraulically upgradient of oilfield sites and are uncontaminated by oilfield wastewaters whereas the remainder of the water-quality sites are downgradient of oilfield sites and have wastewater contamination signatures. Inset map shows the location of the study site in relation to the Williston Basin and Prairie Pothole Region.

landowners and previous researchers (Reiten and Tischmak, 1993). As of 1 April 2019, four of the 11 oil wells were plugged and abandoned, six were shut-in (meaning they could produce again), and one was active, as was the tank battery. Oilfield site names follow the nomenclature of Reiten and Tischmak (1993).

Goose Lake is located within the Missouri Coteau, an area characterized by hummocky terrain with poorly developed drainage systems and thousands of depressional wetlands (Martin and Hartman, 1987). Surficial geology of the Missouri Coteau is comprised of tens of meters of glacial deposits consisting primarily of fine-grained, clay-rich glacial tills and coarse-grained glaciofluvial (e.g., outwash) deposits (Fullerton et al., 2004). Goose Lake sits at a transition from clay-rich glacial till in the northwest portion of the study area to coarse-grained outwash deposits overlying glacial till in the southeast part of the study area (Fig. 1). The outwash deposits are 1–7 m thick and saturated, creating unconfined aquifer conditions, and these deposits have allowed contamination to spread across the study site (Preston et al., 2014; Reiten and Tischmak, 1993; Rouse et al., 2013). Two groundwater flowpaths have been delineated in the outwash deposits, with both flowpaths discharging into West Goose Lake (site 307B; Reiten and Tischmak, 1993). Depth to groundwater generally becomes shallower downgradient in the northern flowpath, whereas the opposite is true for the southern flowpath. As of 2009, wastewater plumes had migrated 600–1600 m from probable sources and contaminated both flowpaths as well as all wetlands downgradient from the oilfield sites (Preston et al., 2014; Reiten and Tischmak, 1993; Rouse et al., 2013). Wetlands 124D, 124E, and 124F, which appear as a single water body in the National Wetland Inventory dataset (Fig. 1; U.S. Fish and Wildlife Service (USFWS, 2015), are three distinct wetland basins most years.

To place the timing of energy development and sample collection into a historical climate perspective, precipitation data were obtained from nearby U.S. Cooperative Observer Program (National Oceanic and Atmospheric Administration (NOAA, 2018) stations for January 1960 – September 2018 (Fig. 2). When available, data from Westby, Montana (station ID 248777; 674 months), located 5 km northeast of Goose Lake, were used. Missing data were filled in first from Fortuna, North Dakota (323196; 23) and then from Plentywood, Montana (246586; 5), which are located 25 km northeast and 34 km southwest of Goose Lake, respectively. No data were available from either station for March 2011. Goose Lake has a semi-arid climate characterized by warm summers and cold winters (average July and January temperatures in Westby, Montana of 20.4° and −14.2 °C, respectively). Average precipitation in Westby, Montana, is 35.8 cm/year with over one-half (18 cm) falling in June, July, and August, primarily during thunderstorms. These storms are often localized and can produce considerable variation in precipitation totals over tens of kilometers. Additionally, total precipitation generally increases from west to east across the Williston Basin. Energy development in the late-1960s occurred during a relatively wet period that continued until approximately 1980. Between 1980 and 1990, precipitation was generally below average, followed by above-average precipitation between 1990 and 2000. Drier conditions returned from 2000 to 2009, with variable conditions from 2010 to 2018 including the wettest (2011) year since 1960.

2.2. Temporal changes in water-quality

2.2.1. Previous research

Goose Lake has a long history of water-quality monitoring, and a combination of previously published and new data were used to extend estimates of the fate and transport of chloride. For brevity, sampling procedures and analytical methods of historical data are provided in the referenced literature. The Montana Bureau of Mines and Geology (MBMG) conducted the initial research at Goose Lake, installing the majority of shallow monitoring wells and collecting numerous groundwater and surface-water samples from 1988 to 1990. All sites shown in Fig. 1 were sampled by MBMG except for BGWL, BGWL-B, RAB2, and 264Y (Reiten and Tischmak, 1993). The U.S. Fish and Wildlife Service resampled seven locations (264A, 264B, 264D, 264J, 264K, 264M, and 264P) and installed a new groundwater well (RAB2) in 2005 and resampled eight locations (117F, 124C, 124H, 124J, 124N, 124O, 124P, and 126B) in 2006 (Rouse et al., 2013). In 2009, the U.S. Geological Survey (USGS) sampled all sites except for BGWL-B, 264B and 264Z and also collected a produced water sample from the tank battery to characterize contaminant sources at Goose Lake (Preston et al., 2014; Preston et al., 2012). Water samples were analyzed for major ions and trace-metals during all sampling efforts.

2.2.2. New sample collection and comparison metrics

The USGS conducted additional, targeted water-quality sampling in 2015 followed by sampling four wetlands and all but one groundwater well in 2018 using published methods (USGS, variously dated). Water-quality samples were collected from three wetlands (BGWL-B, 264J, and 264P), three groundwater wells (264D, 264T, 264R), and from the inflow and separator tank at the tank battery (264) in 2015. Samples were analyzed for major ion and trace-metal concentrations at the Reston Biogeochemical Processes in Groundwater Laboratory (Reston, Virginia, USA) using published methods (USEPA, 2007; Cozzarelli et al., 2017). Chloride concentration and specific conductance were measured onsite at five additional wetlands (124F, 264E, 264K, and 264M) during July 2015. Chloride concentrations were determined onsite with Hach QuanTab test strips (catalog nos. 27449-40, 27513-40) and specific conductance was measured with a YSI EXO multimeter. Onsite chloride and specific conductance measurements were collected from wetlands 264J, 264K, and 264M in May 2018 and 307B in June 2018 using Hach

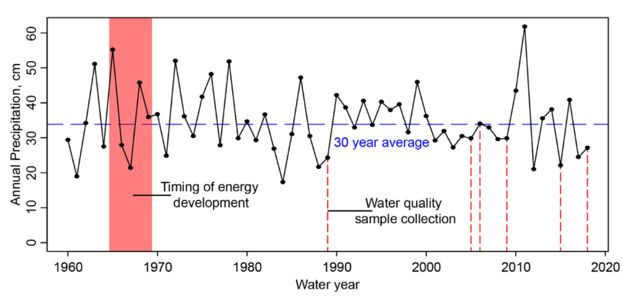


Fig. 2. Total precipitation per water year (October 1–September 30) at Westby, Montana, which is located approximately 5 km northeast of the Goose Lake study site, Sheridan County, Montana, USA. Data from NOAA (2018) with the 30-year average calculated from 1988 to 2018.

chloride test strips and a Hana HI98130 meter. Both Hossack et al. (2018) and Reiten and Tischmak (1993) reported a nearly 1:1 relations between laboratory- and QuanTab-derived chloride concentrations across a range of values and data from both methods are used interchangeably here. Two groundwater wells (124N and 264D) were sampled in May 2018 and all groundwater wells (except 124H) were sampled during September 2018. The 2018 groundwater samples were field filtered and shipped on ice for analysis at the MBMG Analytical Laboratory (Butte, Montana, USA) using published methods (U.S. Environmental Protection Agency (USEPA, 2007)); however, May samples were analyzed for chloride and specific conductance only, whereas September samples were analyzed for major ions and trace-metals. Chloride and specific conductance data collected from 1989-2018 at Goose Lake are provided in Preston (2019).

Given the high variability in natural salinity in the PPR, an empirically derived Contamination Index (CI) has been used at Goose Lake to differentiate naturally saline waters from those contaminated by oilfield wastewaters (Reiten and Tischmak, 1993). The CI is the ratio of chloride concertation (mg/L) to specific conductance (μS/cm) in a water sample. CI values >0.035 in the region have typically been considered indicative of wastewater contamination whereas CI values >0.35 and chloride concentrations >10,000 mg/L indicate extremely contaminated sites (Preston et al., 2014). In general, the changes in surface water chemistry are more dynamic than groundwater. Therefore, temporal changes in water quality between the different sampling efforts are evaluated by comparing changes in both CI values and accompanying chloride concentrations.

2.3. Natural attenuation rates

Natural attenuation rates for a given contaminant are often calculated from first-order relations developed from temporal concentration data from single monitoring wells and/or from concentration versus distance relations developed from numerous monitoring wells sampled concurrently (Beyer et al., 2007; Newell et al., 2002). Point attenuation rates from single monitoring wells allow for the estimation of the time required to reach remediation or water-quality targets at that location, whereas bulk attenuation rates provide distance-based estimates to evaluate whether a plume will expand, remain stable, or shrink. These approaches are best documented at petroleum spill sites where contaminants experience microbial degradation or sorption reactions; however, attenuation rates can also be calculated for conservative contaminants (such as chloride at Goose Lake) where the only form of attenuation is advective-dispersive transport from the source zone (Newell et al., 2002).

Chloride concentration point attenuation rates were calculated at six groundwater monitoring wells (124N, 126B, 264A, 264B, 264D, and 264Q; Fig. 1) near known wastewater source zones. Point attenuation rates for each monitoring well were calculated from the slope of a linear regression fit to log transformed chloride concentration data as a function of time (Table 1). The 2018 chloride concentrations from monitoring well 124N were highly elevated relative to 2009, possibly owing to the downgradient migration of wastewater from oilfield site 126, and point attenuation rates were calculated both with and without the 2018 data. Bulk attenuation rates were calculated from the slope of linear regressions fit to log transformed chloride concentration data from groundwater wells 264A, 264B, 264D, and 264Q, located on the inferred centerline of the contaminant plume (Fig. 1), as a function of distance from the tank battery (264). Individual bulk attenuation rates were obtained from log transformed chloride concentrations from the 1989, 2005, 2009, and 2018 data to evaluate potential temporal changes in bulk attenuation rates (Table 1). Robust standard errors were calculated for all regression models. Statistical differences in regression coefficients (rate constants) were determined with Tukey pairwise comparisons for both the point attenuation and bulk attenuation constants. From the point attenuation constants, the time required for chloride concentrations to reach three water-quality targets were predicted for each well: acute (860 mg/L) and chronic (230 mg/L) toxicity benchmarks (USEPA, 1988) and a local background level (9.5 mg/L; the average chloride concentration of uncontaminated groundwater samples from 126C and 264T in 1989) were predicted for each well. From the bulk attenuation constants, the distance that chloride emitted from the source zone at the time of sampling would travel before being diluted to these same water-quality targets was calculated.

3. Results

3.1. Temporal changes in water-quality

The 2015 produced-water and separator-tank samples (oilfield wastewaters) generally had greater major-ion concentrations than the 2009 produced-water sample; however, the relative trends in major-ion distributions were similar. For example, specific conductance and chloride concentrations in produced water samples were 229,000 μS/cm and 145,430 mg/L in 2015 compared to 223,000 μS/cm and 121,000 mg/L in 2009. Produced-water samples in both years were dominated by sodium and chloride ions and contained elevated concentrations of other major ions including calcium, magnesium, potassium, and ammonium and elements such as barium, bromine, lithium, nickel, rubidium, and strontium relative to uncontaminated surface and groundwater (Preston et al., 2014; Hossack et al., 2017; Smalling et al., 2019).

Results for water-quality samples collected during 1989, 2005–06, and 2009 are provided in Table 1 of Preston et al. (2012) and briefly summarized here. Initial sampling in 1989 revealed widespread chloride contamination at Goose Lake; however, groundwater from two monitoring wells (126C and 264T) had CI values less than contamination threshold (0.035; Figs. 3 and 4). Samples from 3 wetlands and 10

Table 1
Sample dates and chloride concentrations (mg/L) from groundwater samples used to determine natural attenuation rates at the Goose Lake study site in Sheridan County, Montana.

124N		126B		264A		264B		264D		264Q	
Date	Chloride	Date	Chloride	Date	Chloride	Date	Chloride	Date	Chloride	Date	Chloride
		12/13/1988[a]	24,630	12/12/1988[a]	72,400	12/12/1988[a]	50,200	12/13/1988[a]	21,760	10/13/1989	10,300
10/17/1989[b]	12,000	4/20/1989[b]	26,540	10/13/1989[b]	73,395	4/20/1989[b]	37,780	4/20/1989[b]	18,120	6/26/1990[a]	10,465
6/20/1990[a]	10,915	6/20/1990[a]	26,705	6/27/1990[a]	54,600	6/26/1990[a]	48,740	6/27/1990[a]	18,150		
				9/15/2005[c]	30,841	9/15/2005[c]	22,638	9/15/2005[c]	7302	9/15/2005[c]	11,330
4/4/2006[c]	4217	4/5/2006[c]	15,382								
5/14/2009[c]	3340	5/14/2009[c]	15,800	5/13/2009[c]	37,500			5/15/2009[c]	5330	5/15/2009[c]	3940
								6/23/2015[c]	4644		
5/20/2018[c]	3860							5/20/2018[c]	3743		
9/4/2018[c]	5221	9/5/2018[c]	11,458	9/5/2018[c]	21,972	9/5/2018[c]	17,298	9/5/2018[c]	4248	9/6/2018[c]	3356

[a] Onsite chloride measurement.
[b] Average of concurrent onsite and laboratory measurements.
[c] Laboratory measurement.

groundwater wells classified as extremely contaminated (CI > 0.35 and chloride concentration >10,000 mg/L) in 1989. All water-quality samples in 2005–06, which did not include reference sites BGWL and BGWL-B or groundwater wells 126C and 264 T, indicated wastewater contamination. Between 1989 and 2005–06, chloride concentrations had increased in three wetland (range: 860 to 1800%) and two groundwater samples (range: 23 to 34%) and decreased in one wetland (18%) and nine groundwater samples (range: 31 to 84%). Samples from five groundwater wells were classified as extremely contaminated in 2005–06. In 2009, all sampled sites except the background wetland (BGWL), which is upgradient from oilfield sites, were contaminated by wastewater, including the two groundwater wells that had not been contaminated in 1989. Between 2005 and 06 and 2009, chloride concentrations had decreased in all four wetlands (range: 53 to 92%) and four groundwater wells (range: 21 to 78%) and increased in six groundwater wells (range: 3 to 570%). Samples from four groundwater wells were classified as extremely contaminated in 2009.

New water-quality sampling in 2015 and 2018 indicated persistent wastewater contamination at Goose Lake (Figs. 3 and 4); however, chloride concentrations generally decreased in groundwater samples and increased in surface water samples relative to previous sampling events. A second background wetland (BGWL-B) upgradient from oilfield sites was first sampled in 2015, and along with the 2009 BGWL sample, represents surface water from wetlands unaffected by wastewater. Surface-water samples from these wetlands had the lowest chloride concentrations (10 and 9 mg/L, respectively) at Goose Lake. Between 2009 and 2015, chloride concentrations had decreased in two groundwater samples (6 and 13%) and increased in one groundwater sample (120%) and in all six contaminated wetland samples (range 0.2 to 3800%). No water samples collected in 2015 were classified as extremely contaminated. Wastewater contamination was evident in all 2018 water-quality samples. Between 2015 and 2018, chloride concentrations had decreased in two of the three groundwater (9 and 25%) and wetland samples (5 and 80%) and increased in the other groundwater (22%) and wetland sample (270%). The other 15 sites sampled in 2018 had not been sampled since 2009 and chloride concentrations decreased since that time in 10 groundwater samples (range 15 to 50%) and increased in samples from the other 4 groundwater wells (range 63 to 390%) and wetland 307B (120%). Samples from two groundwater wells were classified as extremely contaminated in 2018. In general, the groundwater samples in 2015 and 2018 that had decreased chloride concentrations relative to 2009 were from wells located near known wastewater source zones whereas samples with increased chloride concentrations were from wells located in distal portions of the flowpaths.

3.2. Natural attenuation rates

Chloride concentrations in six groundwater wells (124N, 126B, 264A, 264B, 264D, and 264Q) near wastewater source zones decreased between 1989 and 2018, with these significant decreases modeled by a first-order decay constant ($p < 0.05$ for all regressions) allowing for determination of point attenuation rates for each groundwater well (Fig. 5, Table 2). There were statistical differences ($p < 0.05$ in Tukey pairwise comparisons) in point attenuation rate constants between different wells. The rates for wells 124N (excluding the 2018 data) and 264D were greater than 126B, 264A and 264B whereas the rate constant for 264Q was not significantly different from any other rate constant. There is a considerable range in the time required to reach the three water-quality targets based on the regression equations for the different groundwater wells. Depending on the individual site, return to the EPA acute toxicity benchmark (860 mg/L) is not predicted until the years 2045 to 2113, EPA chronic toxicity (230 mg/L) is not predicted until 2069 to 2160, and local background (9.5 mg/L) is not predicted until 2126 to 2275. Although chloride concentrations in groundwater well 124N also followed a first-order decay constant ($p < 0.05$) both with and without 2018 data, the increased chloride concentration in 2018 limits the strength of the predictions to reach water-quality targets and were not calculated.

Chloride concentrations in 1989, 2005, 2009, and 2018 decreased with distance from the tank battery (264), with these significant decreases also modeled by a first-order decay constant ($p < 0.05$ for all regressions) allowing for determination of bulk attenuation rates for each year (Fig. 5, Table 3). There were minor differences in bulk attenuation rates between the different years; however, these differences were not statistically different ($p > 0.05$ in Tukey pairwise comparisons). Continued flushing of chloride from the source area has reduced the potential downgradient migration distance to achieve the three water-quality targets. In 1989, the respective distance that chloride was predicted to migrate downgradient before dilution to EPA acute, EPA chronic, and background levels were 949, 1220, and 1878 m whereas these predicted distances were reduced to 673, 922, 1525 m in 2018. Several wetlands in the southern flowpath are within these distances as measured downgradient from the tank battery; 264Z (500 m downgradient), 264 J (610 m), 264 K (1,000 m), 264 M (1030 m), 264Y (1230 m) and 264P (1650 m).

4. Discussion

Legacy wastewater handling and disposal practices have resulted in persistent secondary salinization at Goose Lake (Reiten and Tischmak, 1993; Rouse et al., 2013; Preston et al., 2014). Produced-water samples from the tank battery in 2009 and 2015 had high TDS and chloride concentrations (121,000 and 145,000 mg/L, respectively); however, these concentrations were on the lower end of chloride concentrations (88,000–199,000 mg/L; median; 168,000 mg/L) from 62 additional samples from the same producing unit (Blondes et al., 2017). These results demonstrate the considerable spatial and temporal variation in wastewater chemistry that can occur within a single oilfield and across different producing areas within the same formation. However, it is likely that wastewater disposed during initial development was similar

Table 2

Location of the screened interval, point attenuation rates, robust standard errors, coefficient of determination (R^2) values, and the year in which chloride concentrations in selected groundwater wells at the Goose Lake study site in Sheridan County, Montana, are predicted to reach Environmental Protection Agency (EPA) acute (860 mg/L) and chronic (230 mg/L) toxicity benchmarks and a local background level (9.5 mg/L) if the relations observed between 1988 and 2018 remain consistent.

Site	Screened Interval	Point attenuation rate/year	Robust standard error	R^2	EPA Acute toxicity	EPA Chronic toxicity	Background
124N*	100% Outwash	0.064^{A**} 0.036^{B}	0.0020 0.0089	0.99 0.65	NA	NA	NA
126B	60% Till 40% Lake	0.028^{B}	0.0017	0.97	2113	2160	2275
264A	60% Outwash 40% Lake	0.037^{B}	0.0043	0.90	2106	2141	2227
264B	60% Outwash 40% Lake	0.034^{B}	0.0048	0.89	2103	2142	2235
264D	60% Till 40% Outwash	0.055^{A}	0.0026	0.98	2045	2069	2126
264Q	100% Outwash	0.043^{A,B}	0.0051	0.95	2048	2078	2152

^{A,B}Denotes significant differences in point attenuation rate constants.

* Large increases in chloride concentrations in 2018 may indicate potential downgradient migration of brine from an upgradient source area. Given this uncertainty, the year in which water-quality targets would be reached were not calculated for this groundwater well.

** Point attenuation rate calculated from 1988 to 2009 data only.

Table 3
Bulk attenuation rates, robust standard errors, coefficient of determination (R^2) values, previous years' precipitation, and potential downgradient transport distances of chloride emitted from the source zone in 1989, 2005, 2009, and 2018 before dilution Environmental Protection Agency (EPA) acute (860 mg/L) and chronic (230 mg/L) toxicity benchmarks and a local background level (9.5 mg/L) at the Goose Lake study site in Sheridan County, Montana.

Year	Bulk attenuation rate/meter	Robust standard error	R^2	Precipitation in previous year (cm)	EPA Acute toxicity (m)	EPA Chronic toxicity (m)	Background (m)
1989	0.0049	0.00035	0.98	21.7	949	1220	1878
2005	0.0057	0.00078	0.93	30.5	693	923	1480
2009	0.0059	0.00097	0.90	29.6	668	890	1429
2018	0.0053	0.00072	0.89	24.5	673	922	1525

to contemporary wastewater and contained high TDS values dominated by sodium and chloride ions.

Although wastewater contamination is widespread at Goose Lake, there are clear differences in the CI and chloride profiles between the southern and northern groundwater flowpaths (Figs. 3 and 4) that are likely related to different contaminant pathways. In the southern flowpath, CI and chloride profiles show a rapid reduction from peak values near oilfield site 264, indicating the tank battery is the primary source of contamination. Other oilfield sites downgradient are likely associated with minor inputs of wastewater as evidenced by the slight uptick in CI values and chloride concentrations downgradient of oilfield sites 127 and 128. In the northern flowpath, contaminant plumes are associated with oilfield sites 117, 126, and 124 as evidenced by the high CI values and chloride concentrations in adjacent groundwater wells; however, no single source dominates the entire flowpath. Reiten and Tischmak (1993) noted local land owners reported direct discharges of wastewater from oilfield site 124 into wetlands 124D, 124E, and 124F. Given that within each year of sampling the peak chloride concentrations in the northern flowpath occurred in groundwater wells 124H or 124P, which are immediately downgradient from these wetlands, this volume of wastewater may have been greater than that disposed of in reserve pits near the individual oilfield sites. These results highlight the multiple contamination pathways that can exist at legacy energy production sites and the need for detailed site investigations to determine the source(s) and extent of contamination.

Evaluating the spatial and temporal changes in water-quality at Goose Lake requires an understanding of the general hydrogeologic framework in the PPR. Wetlands in the PPR are connected to the shallow groundwater system along a gradient of recharge, flow-through, and discharge wetlands with hydroperiod and salinity generally increasing downgradient (LaBaugh et al., 1987; Winter, 2003). However, annual- to decadal-scale climatic fluctuations constantly evolve the PPR wetland hydroperiod and salinity continuum (Euliss et al., 2004; Euliss and Mushet, 1996; Winter, 2003). Transient groundwater mounds commonly develop adjacent to recharge and flow-through wetlands during drawdown in response to edge-focused groundwater recharge (Arndt and Richardson, 1993; Winter, 2003). Evaporative concentration of dissolved solids is strongly edge focused as summer evaporation results in the upward capillary movement of groundwater and intrasedimentary deposition of salts (Last and Ginn, 2005; Steinwand and Richardson, 1989). Therefore, seasonal drawdown and/or extended dry periods likely results in intrasedimentary evaporitic deposition of wastewater-derived salts along the edges of contaminated wetlands and within previously saturated sediments above contaminant plumes. These salts can then be remobilized into wetland basins or back into the shallow groundwater system during spring runoff, in wet years, or after large recharge events (Reiten and Tischmak, 1993; Preston et al., 2014). This remobilization of salts may complicate remediation efforts and lengthen the time for attenuation to achieve benchmark or background concentrations.

As seen in the groundwater-quality results at Goose Lake, this episodic flushing likely produces temporary pulses of higher salinity superimposed on a general decrease in salinity near wastewater source zones. For example, an 11-year period of near- or above-average precipitation preceded the 1989 sampling followed by a return to drought conditions from 2000 to 2009 (Fig. 2). This likely raised the local water table initially and allowed for considerable flushing of wastewater before stranding salts in the capillary zone as the water table lowered. Chloride concentrations decreased noticeably in nine of the 11 groundwater wells sampled in 2005–06 relative to 1989, yet had increased in six of these wells by 2009 (Fig. 4). During the 2000–09 drought, 2006–07 had near average precipitation, which may have provided enough recharge to remobilize previously stranded salts. Precipitation was highly variable between 2009 and 2018 with more years having above-average than below-average precipitation. Chloride concentrations decreased in 10 of the 16 groundwater wells between 2009 and 2018 with most of these wells located near oilfield sites in the upgradient portions of the flowpaths. In contrast, the six wells with increased chloride concentrations were generally located in the distal portions of the flowpaths (for example 117J, 124P, 264S, and 264T) where chloride continues to be flushed downgradient. From 1993 through 2012, the PPR experienced an extended period of increased precipitation likely unequaled in the preceding 500 years (Mushet et al., 2015). Although this likely raised local water tables and increased the flushing of wastewater at numerous legacy oilfield sites, a return to drought conditions may strand salts in currently saturated sediments that then could become remobilized in future deluge cycles or following large recharge events.

In contrast to groundwater quality, the temporal changes in surface water quality were highly variable. All wetland sampling occurred in years with below-average precipitation (Fig. 2); however, the timing and quantity of precipitation in preceding years strongly influences water levels and water-quality in PPR wetlands (Niemuth et al., 2010; Ouyang et al., 2014; Zhang et al., 2009). Wetland samples in 1989 were collected in late-April or early-May when dissolved constituent concentrations are typically diluted by snowmelt. In contrast, the 2005 samples, which generally had markedly increased chloride concentrations relative to 1989, were collected in mid-June when evapoconcentration increases dissolved constituent concentrations. The 2005 samples were also collected following a longer period of below-average precipitation than the 1989 samples, which can increase salinity in wetlands that do not dry annually (Winter, 2003). Wetland samples in 2009, at the tail end of the 2000–09 drought, were collected in early-May and all samples showed a reduction in chloride concentrations relative to 2005. Finally, chloride concentrations increased in all wetland samples from early-May and late-June in 2015 and all but one wetland sample from early-May in 2018 relative to 2009, whereas groundwater chloride concentrations generally decreased. Therefore, while wetland water quality is useful in identifying wastewater contamination, temporal changes in chloride concentrations appear too dependent on the time of sampling, both seasonally and within a regional climate perspective, to provide a reliable indicator on the movement of contaminant plumes at Goose Lake and, therefore, likely other legacy energy development sites in the PPR.

Chloride concentrations decreased between 1988 and 2018 in the six selected groundwater wells (R^2 values range: 0.89 to 0.99; Fig. 5). These point attenuation rates varied among wells and predict that EPA acute (860 mg/L) and chronic (230 mg/L) toxicity thresholds will be reached in approximately 30–100 and 50–150 years, respectively, whereas background levels (9.5 mg/L) will be reached in 100–250 years (Table 2). The introduction of wastewater likely occurred during energy development in the late 1960s; therefore, the

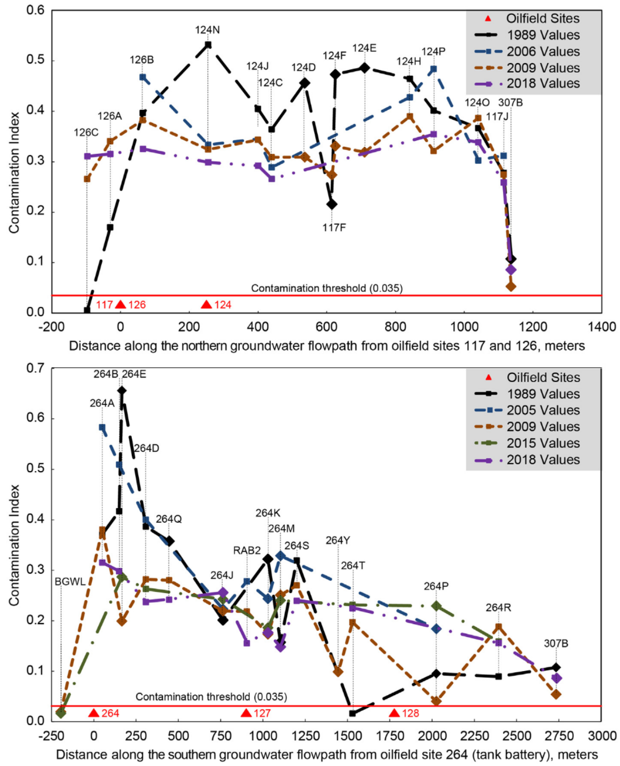
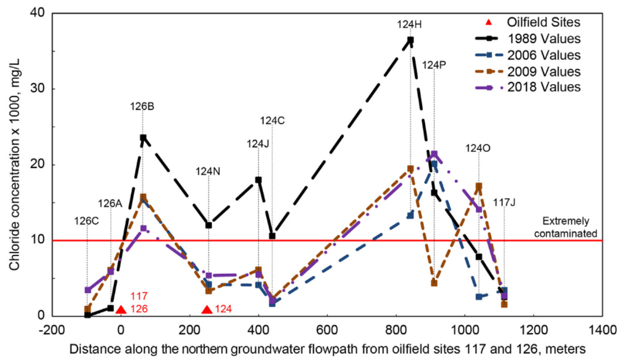


Fig. 3. Temporal Contamination Index values for surface water (wetland; large diamonds) and groundwater (small squares) samples from the northern (top) and southern (bottom) flowpaths by distance from the most upgradient oilfield site at the Goose Lake study site, Sheridan County, Montana, USA. See Fig. 1 for sample locations. In the top panel, distances to sites 117F and 117J are measured from oilfield site 117, whereas distances to all other sites are measured from oilfield site 126.

total predicted time for natural attention is approximately 50 years longer than these model estimates. Given the variation and length of time needed to reach water-quality targets, this work offers an initial estimate of the persistence of wastewater in the PPR and does not attempt to predict the exact year when remediation targets will be reached. However, these results provide the first known quantitative prediction on the natural attenuation rates of chloride contaminated groundwater near abandoned reserve pits in the PPR and support several previous qualitative predictions that leachate generation from reserve pits would continue for tens to hundreds of years (Beal et al., 1987; Murphy et al., 1988; Preston et al., 2014; Reiten and Tischmak, 1993). Furthermore, these predictions apply only to chloride whereas other cations will continue to be released after chloride is removed due to difference in cation exchange affinity (Appelo et al., 1993; Appelo et al., 1990; Valocchi et al., 1981). Therefore, calcium, sodium, potassium, strontium and other cations are likely to be released from near surface sediments for years to decades after chloride has been attenuated.

Attenuation rates derived from field data are generally considered conservative estimates (Chapelle et al., 2003) and are often overestimated (Rittmann, 2004) with additional data improving rate estimates. Although other approaches (e.g., numerical groundwater modeling) have been used to predict contaminant transport rates, we lack data such as sediment porosity/permeability and groundwater flow rates that are necessary to adequately constrain such models and are, therefore, unable to compare the field data-based approach presented here to more quantitative models. Additionally, despite having data spanning 30 years, the predicted point attenuation rate constants were calculated from relatively small sample sizes and the inclusion of 2015–18 changed the rates for some wells. Point attenuation rates derived from 1988 to 2009 data were slightly lower in two wells (126B: 1.6% and 264A: 2.8%), yet considerably greater in three others (264D: 12%; 264Q; 16% and 264B 19%) compared to rates including the 2015–18 data. Furthermore, many legacy sites consist of numerous oil wells and downgradient migration of contaminants from upgradient sources could slow attenuation as observed in groundwater well 124N. Therefore, continued decadal monitoring could validate and/or refine these attenuation rates and reduce the uncertainty in the associated predications.

The significant differences in point attenuation rates are likely related to heterogeneity within the outwash deposits and differences in hydraulic conductivities between different glacial deposits. Point attenuation rates were variable in wells fully or partially screened in the



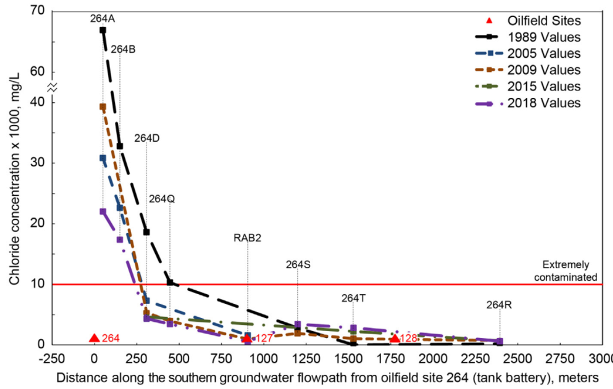


Fig. 4. Temporal chloride concentrations, in mg/L, for groundwater samples from the northern (top) and southern (bottom) flowpaths by distance from the most upgradient oilfield site at the Goose Lake study site, Sheridan County, Montana, USA. See Fig. 1 for sample locations. In the top panel, the distance to site 117J is measured from oilfield site 117, with distances to all other samples measured from oilfield site 126. All samples above the extremely contaminated line also had contamination index values >0.35 except 126B, 124O, and 264B in 2018.

outwash deposits (0.034 to 0.064 yr^{-1}) whereas the rate for 124B (0.028 yr^{-1}), which is screened entirely in glacial till and lacustrine deposits, was the lowest observed and significantly less than two of the wells in the outwash deposits. This is unsurprising as glacial tills generally have much lower hydraulic conductivities than outwash deposits. Considerable emphasis has been placed on reserve pits in permeable outwash deposits as contaminant plumes in these settings can migrate long distances (Beal et al., 1987; Preston and Chesley-Preston, 2015), yet surficial deposits in the PPR of the Williston Basin are predominately glacial tills where natural attenuation of chloride may take considerably longer. For example, the 1989 chloride concentrations in wells 264D (21,760 mg/L) and 126B (24,630 mg/L) were similar; however, background chloride levels are predicted to be reached nearly 150 years sooner in 264D than 126B.

The longevity of secondary salinization from energy development in the PPR is further illustrated by the bulk attenuation rate calculations (Fig. 5; Table 3). Chloride concentrations generally decreased between successive bulk attenuation rate calculations as advective-dispersive transport removed chloride from the source area. As a result, potential distances of downgradient chloride migration until dilution reduces concentrations to EPA acute, EPA chronic, and background levels have decreased by 276, 298, and 353 m, respectively, between 1989 and 2018. However, this equates to respective reductions of only 29, 24, and 18% of the predicted distance of downgradient migration to reach the three water-quality targets over 29 years. Furthermore, several wetlands are still within the 2018 migration distances and will likely receive contaminated groundwater for years or decades. Although no statistical differences existed between the four bulk attenuation rate constants, the variations in individual rates are possibly related to climatic fluctuations. For example, bulk attenuation rates appear positively corelated with total precipitation in the previous year and further supports the remobilization of salts stranded in the capillary zone during drought cycles.

West Goose Lake is a large discharge basin and the likely terminus for the local groundwater flowpaths at Goose Lake. West Goose Lake is further downgradient than the 2018 predicted distance for dilution of chloride to background concentration in the southern flowpath. However, the 2018 chloride concentrations in groundwater wells 124P (21,318 mg/L) and 124O (13,934 mg/L), which are located <200 m upgradient from West Goose Lake, in the northern flowpath were similar to those of 244A and 264B indicating the potential for considerable future discharges of chloride into the lake basin. Chloride will

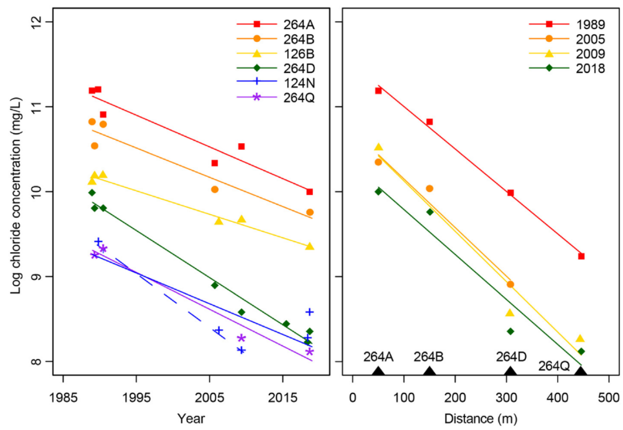


Fig. 5. Fitted linear regression models used to calculate point attenuation constants (left) and bulk attenuation rate constants (right) of chloride in the shallow groundwater system at the Goose Lake study site in Sheridan County, Montana. Dashed blue line in the left panel shows the model fit for 124N calculated from the 1989–2009 data only whereas the solid blue line is for all data (1989–2018).

likely continue to accumulate in West Goose Lake given that elevated groundwater levels in discharge basins often create hydraulic barriers against out-migration of salinity (Van der Kamp and Hayashi, 2009). In the PPR, one of the few potential removal pathways for accumulated chloride in discharge basins is through eolian deflation (removal by wind) of salts that precipitate on the surface (LaBaugh et al., 1998).

There is a growing understanding of the detrimental ecological effects of secondary salinization from energy development. Water levels in flow-through and discharge wetlands are partially sustained by shallow groundwater (Winter, 2003), and the discharge of contaminated groundwater into wetlands coupled with evapoconcentration can increase chloride concentrations seasonally with important biological implications. For example, the chloride concentrations in wetlands 264J and 264K increased roughly four- and twelve-fold (660 to 2800 mg/L and 400 to 4700 mg/L, respectively) between May and September of 2009. Amphibians have highly permeable skin making them especially susceptible to excess salts and other pollutants that can cause direct mortality or sub-lethal effects such as reduced activity, immune function, and size (Dananay et al., 2015; Elphick et al., 2011; Hopkins and Brodie, 2015; Sanzo and Hecnar, 2006). Hossack et al. (2018) documented a 50% reduction in the abundance of boreal chorus frog and tiger salamander larvae in PPR wetlands at chloride concentrations of 298 and 1260 mg/L, respectively. Chloride concentrations in 15 of the 17 groundwater samples in 2018 were >1260 mg/L and all were >298 mg/L. Therefore, evapoconcentration in wetlands receiving groundwater contaminated with wastewater may create ecological traps with these wetlands being habitable to amphibian larvae during the early spring but becoming uninhabitable as chloride concentrations increase throughout the summer. Additionally, wastewater-contaminated PPR wetlands had lower macroinvertebrate taxonomic richness and diversity than uncontaminated wetlands, with chloride concentration being the water-quality parameter most strongly correlated with macroinvertebrate community structure (Preston et al., 2018; Preston and Ray, 2017).

Oilfield wastewaters also has negative effects to terrestrial environments through plant mortality (Keiffer and Ungar, 2001) and altered soil structure (Hoffman and Shannon, 1986). Auchmoody (1989) attributed rapid revegetation of a Pennsylvanian forest following removal of a leaking reserve pit to abundant precipitation and soil flushing. However, spill sites in arid environments can remain unvegetated for years due to elevated soil salinity and deterioration of soil structure (Munn and Stewart, 1989; Murphy et al., 1988; Pichtel, 2016). Soil absorption of sodium from wastewater can decrease hydraulic conductivity, reducing infiltration below levels necessary for plant growth (Hoffman and Shannon, 1986), and create surficial salt crusts that can persist for decades (Keiffer and Ungar, 2002). Several species of salt tolerant halophytes have been shown to accumulate sodium and chloride into plant tissues and reduce soil salinity when grown on wastewater spill sites (Keiffer and Ungar, 2001; Qadir et al., 2006). At Goose Lake, *Salicornia rubra* (pickleweed; a halophytic succulent) commonly grows in several contaminated wetland basins during drought years; however, these plants need to be harvested to facilitate soil bioremediation (Keiffer and Ungar, 2002).

Secondary salinization from energy development is a global phenomenon with wastewater contamination documented on all continents (excluding Antarctica; Johnston et al., 2019) and in biomes ranging from the Siberian tundra (Moskovchenko et al., 2009) to the Amazon rainforest (Moquet et al., 2014). However, several factors (volume and chemistry of wastewater released, surficial geology, precipitation, etc.) influence natural attenuation. Williston Basin wastewater have some the highest recorded TDS values (>450,000 mg/L; Iampen and Rostron, 2000) whereas average TDS values of produced waters from other domestic energy plays are generally lower (Woodford: 30,000 mg/L; Barnett; 80,000 mg/L; Marcellus 120,000 mg/L; Acharya, 2011). Similarly, surficial geology, which control rates of groundwater movement, varies considerably between different energy plays as does precipitation, which is critical for natural attenuation. It is estimated that soils contaminated by wastewaters require at least a 100-fold dilution with fresh water to reduce salts to levels suitable for plant growth (Munn and Stewart, 1989). However, many of the leading domestic energy-producing regions have semi-arid climates that receive <25 cm of rain per year (Keiffer and Ungar, 2002). Therefore, while it is unclear if the persistence of secondary salinization observed in the Williston Basin is transferable to other areas, attenuation rates may be similar in other arid environments (i.e., the Permian Basin in Texas, the Denver Basin in Colorado, or the Green River Basin in Wyoming) and could be evaluated using similar approaches.

5. Conclusions

Energy development at Goose Lake occurred in the late 1960s and resulted in the persistent salinization of surface and groundwater resources. Long-term monitoring has documented the migration of groundwater plumes and associated changes in chloride concentrations, with a general trend of decreases in chloride concentrations in upgradient areas near likely contaminant sources and increases in downgradient areas. For example, although all sites downgradient from oilfield infrastructure were still contaminated in 2018, the number of extremely contaminated sites was reduced from 13 to 2 between 1989 and 2018. However, extremely contaminated refers to sites with chloride concentrations exceeding 10,000 mg/L, a value that is 11.6 times greater than the EPA acute toxicity threshold, and most wetlands in the study area had chloride levels detrimental to aquatic life. Annual and regional climatic fluctuations produced highly variable changes in wetland water-quality that were often decoupled from changes in groundwater-quality, indicating that surface water-quality is an unreliable method to monitor wastewater migration in the PPR. Despite the decreases in chloride concentrations observed across the study period (1988–2018), reducing chloride concentrations in monitoring wells in upgradient areas to pre-spill levels is predicted to take an additional 100–250 years, or longer. Furthermore, it is predicted that chloride flushed from the tank battery in 2018 will migrate nearly 1 km (922 m) downgradient before being diluted below the EPA chronic toxicity benchmark. Roughly one third of wetlands in the PPR are within 1 km of energy development sites (Preston et al., 2014). These results illustrate the persistence of secondary salinization from energy development in the PPR in the absence of remediation activities and highlight the need to reduce and quickly contain future spills to minimize long-term damage to surface and groundwater resources.

Compliance with ethical standards

The authors declare there are no conflicts of interest. This project was fully funded by Federal and State (Montana) government agencies. This article has been peer reviewed and approved for publication consistent with USGS Fundamental Science Practices https://pubs.usgs.gov/circ/1367/). Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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