REEXAMINING SALINE CONTAMINATION ASSOCIATED WITH OIL AND GAS DEVELOPMENT IN THE PRAIRIE POTHOLE REGION, SHERIDAN COUNTY, MT

by

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APPROVAL

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This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citation, bibliographic style, and consistency and is ready for submission to The Graduate School.

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STATEMENT OF PERMISSION TO USE

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ABSTRACT

Oil and gas development in the Williston Basin often involves the extraction of saline brines and presents a major source of saline contamination to surface and groundwater resources. The Prairie Pothole Region (PPR) is superimposed over much of the Williston Basin and provides critical habitats for migratory birds and waterfowl. Surface and shallow groundwaters in the PPR often contain high levels of total dissolved solids (TDS); however, salts produced from energy development are chemically different than the near-surface salts. To differentiate between saline contaminated and naturally high TDS water samples, Reiten and Tischmak (1993) developed a Contamination Index (CI = chloride concentration / specific conductance), with contamination indicated by values above 0.035. The Goose Lake study area in Sheridan County, Montana lies within the PPR and has a documented history of saline contamination. Local stratigraphy consists of coarse grained glacial outwash deposits overlying clay-rich glacial till, with extensive contamination in the saturated outwash deposits and wetlands evidenced from 1989 water samples. In 2009, water samples were analyzed from thirty locations including twenty six sites sampled in 1989 to determine changes in chemistry. Additionally, geophysical surveys were conducted to determine stratigraphic controls on contaminant transport using a Geonics EM-31 and EM-34.

Widespread contamination was documented in 2009, with one water sample considered uncontaminated based upon the CI data. Two uncontaminated groundwater wells in 1989 had 2009 CI values indicating contamination, documenting the increased aerial extent of contamination due to down-gradient transport. Significant reductions in the median CI values from sites contaminated in 1989 were observed. Assuming reductions in CI values continue at the same rate, contamination would remain for roughly 140 years at the most contaminated site. Geophysical studies showed elevated apparent conductivities associated with saline contamination throughout the study. Contaminant transport is mainly occurring laterally within the outwash deposits, and is evident a minimum of 1,600 m from the source. In contrast, saline penetration into the underlying till is likely less than 30 m. These results demonstrate the potential for co-produced water contamination to aquatic resources far down-gradient of oil and gas developments in glacial outwash deposits within the PPR.

INTRODUCTION

Problem Statement

Development of oil and gas resources in the geologic Williston Basin of the northern Great Plains (Fig. 1) often involves the removal of highly saline brines from the oil bearing formation. Extracted brines, or co-produced waters, are potentially a major source of saline contamination to surface and groundwater resources. Sheridan County, Montana, is within the Williston Basin and has a documented history of saline contamination from oil and gas operations (Reiten and Tischmak, 1993). Production of oil and gas resources began in the late 1950's and continues to the present, with major new developments expected within the Devonian-Mississippian Bakken Formation. The regional landscape is characterized by hummocky topography, with poorly developed drainage and thousands of wetlands. Contamination from co-produced waters results in changes in surface and groundwater saline chemistries (Reiten and Tischmak, 1993), which can affect primary productivity in wetlands, degrade and destroy domestic and stock water resources, and cause declines in production or loss of farmable acreage.

Sheridan County, Montana, lies within the Missouri Coteau physiographic province, an elevated highland above the Central Lowlands, southwest of the Missouri escarpment. It also lies within the Prairie Pothole Region (PPR), a broader region of till plain characterized by ice-decay features and numerous wetlands, often referred to as prairie potholes (Fig. 1). Prairie wetlands are an ecologically recognized resource, providing critical habitat and breeding grounds for numerous wetland and grassland bird species (Batt et al., 1989). Habitat conservation and waterfowl production efforts in the

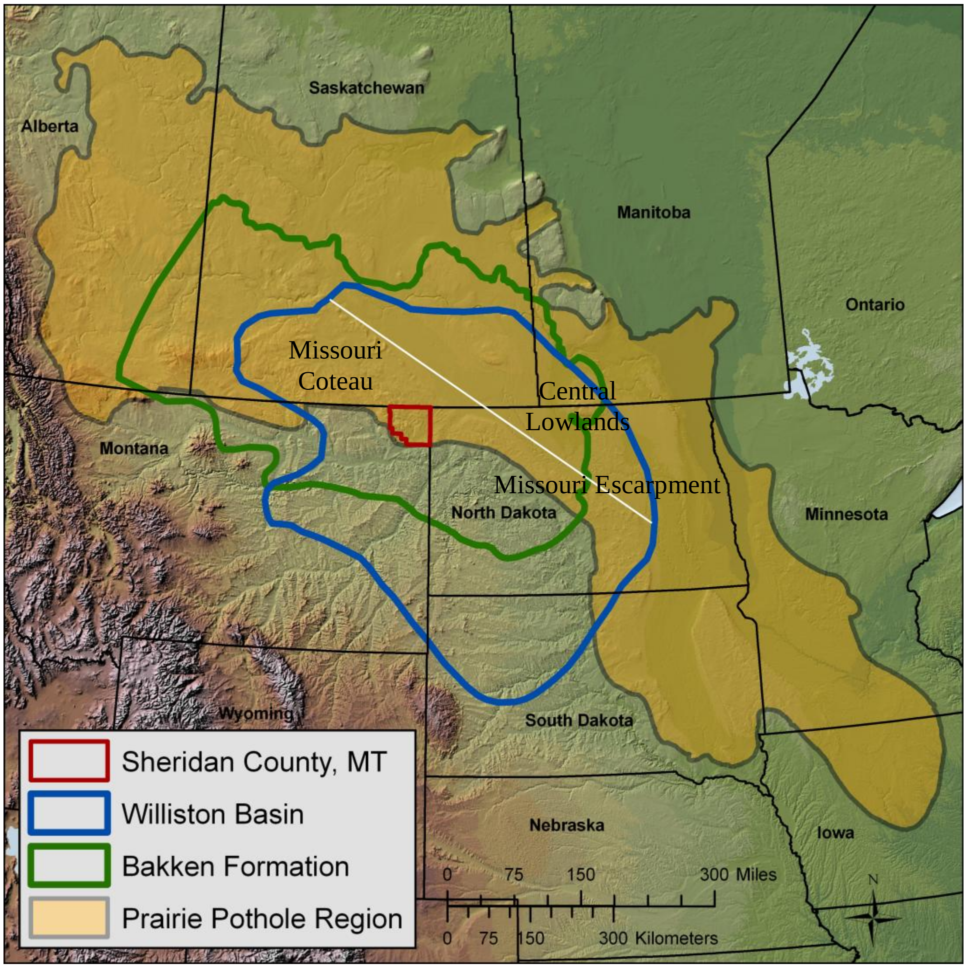

Figure 1. Map showing the prairie pothole region (brown area), Williston Basin boundaries (blue line), Bakken Formation boundaries (green line), Sheridan County, Montana (red box), and the approximate location of the Missouri Escarpment in the Williston Basin (white line).

PPR have focused on preservation of remaining prairie wetlands and retirement programs for agricultural lands adjacent to wetlands in environmentally sensitive areas.

Research presented here is focused on the Goose Lake detailed study area (Fig. 2) as defined by Reiten and Tischmak (1993), and includes areas in and adjacent to T36N, R58E, S27, within the Goose Lake oil field, Sheridan County, Montana. The southern-half section is the Rabenberg Waterfowl Production Area (WPA), managed by the United States Fish and Wildlife Service (USFWS). Potential sources of saline contamination to surface and groundwater resources at the Goose Lake detailed study area consist of eight

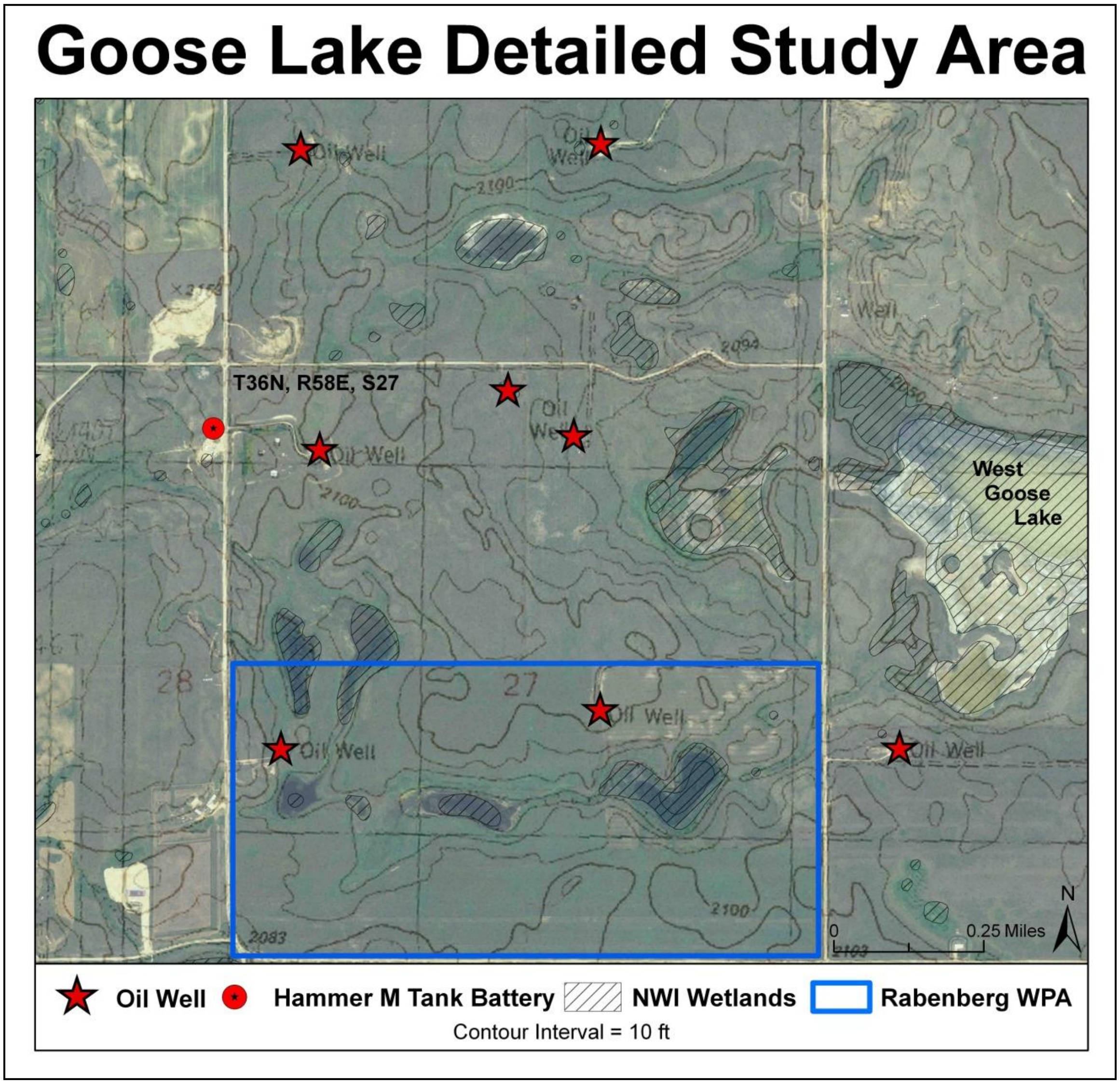

Figure 2. Map showing the Goose Lake detailed study area, active and abandoned oil well sites, the Hammer M oil tank battery, National Wetlands Inventory (NWI) wetlands, and the Rabenberg WPA.

active and abandoned oil wells and their associated reserve pits, transport lines, and the Hammer M tank battery. This area was the location of detailed hydrogeologic investigations, including water quality analysis and geophysical surveys, by the Montana Bureau of Mines and Geology (MBMG) in 1989, and similar but smaller investigations by the MBMG and the USFWS in 2004 - 2006, that documented chloride contaminated surface water and groundwater from oil field activities. In 2009, surface wetlands and groundwater monitoring wells sampled in 1989 and 2005/2006 were resampled. Geophysical surveys were also repeated at locations surveyed in 1989 and 2004 within areas of known chloride contamination in order to characterize the three dimensional geometry of the saline groundwater plumes identified at the Goose Lake site.

This area typifies much of Sheridan County and the PPR where the legacy of saline contamination of surface and groundwater resources from historical oil and gas development intersects with wildlife conservation efforts and surrounding agricultural operations. Understanding the nature and movement of saline contaminated groundwater plumes and the effects to surface waters is paramount to maintaining the continued biological productivity of prairie potholes and the viability of surrounding agricultural operations. In particular, land managers and owners need information regarding the migration of pre-existing saline groundwater plumes and the time required for the natural attenuation of chloride contaminated surface and groundwater.

To address these issues, I will examine two hypotheses. 1) Saline groundwater plumes have continued to migrate down-gradient, towards West Goose Lake, increasing the extent of contamination within the study area. Despite the expected increase in the extent of contamination, the natural attenuation of contaminated surface and groundwater resources is expected to have produced a decrease in the magnitude of contamination in areas identified in 1989. 2) Contaminated groundwater plumes have migrated laterally; mainly confined within the permeable outwash sediments mantling the underlying less permeable glacial till. It is expected that the till is acting as an aquitard, allowing little penetration and downward movement of saline groundwaters relative to lateral flow directions. To test the first hypothesis, field and laboratory water quality data from surface and groundwater samples collected in 1989 and 2009 will be compared to assess temporal changes in water chemistry associated with oil and gas development. The second hypothesis will be tested by analysis of results from geophysical surveys conducted in 2009 that utilized multiple exploration depths, creating a depth profile of saline contaminated groundwater plumes.

Regional Geologic Setting

Subsurface Geology

Located in the Northern Great Plains, the Williston Basin (Fig. 1) is a saucer-shaped intracratonic basin containing 3700 m of sedimentary strata, representing all periods of the Phanerozoic (Klein, 1995). Intermittent subsidence has been occurring in the basin for approximately 500 ma, with conditions favorable for oil formation during roughly half that time (Hansen, 1966). The predominant rock types are carbonates and evaporites, with lesser amounts of sandstone and shale, with strata subdivided according to Sloss's intracratonic sequences. Sloss (1963) recognized six North American intracratonic sequences, separated by regional unconformities, each recording a complete transgressive-regressive cycle. Within the Williston Basin, the majority of the oil producing units are Paleozoic in age (Fig. 3).

Paleozoic sedimentary rocks in the Williston Basin include a sequence of dominantly shallow-water marine carbonates, clastic, and evaporite deposits of Middle Cambrian through Early Permian age (Peterson and MacCary, 1985). The middle

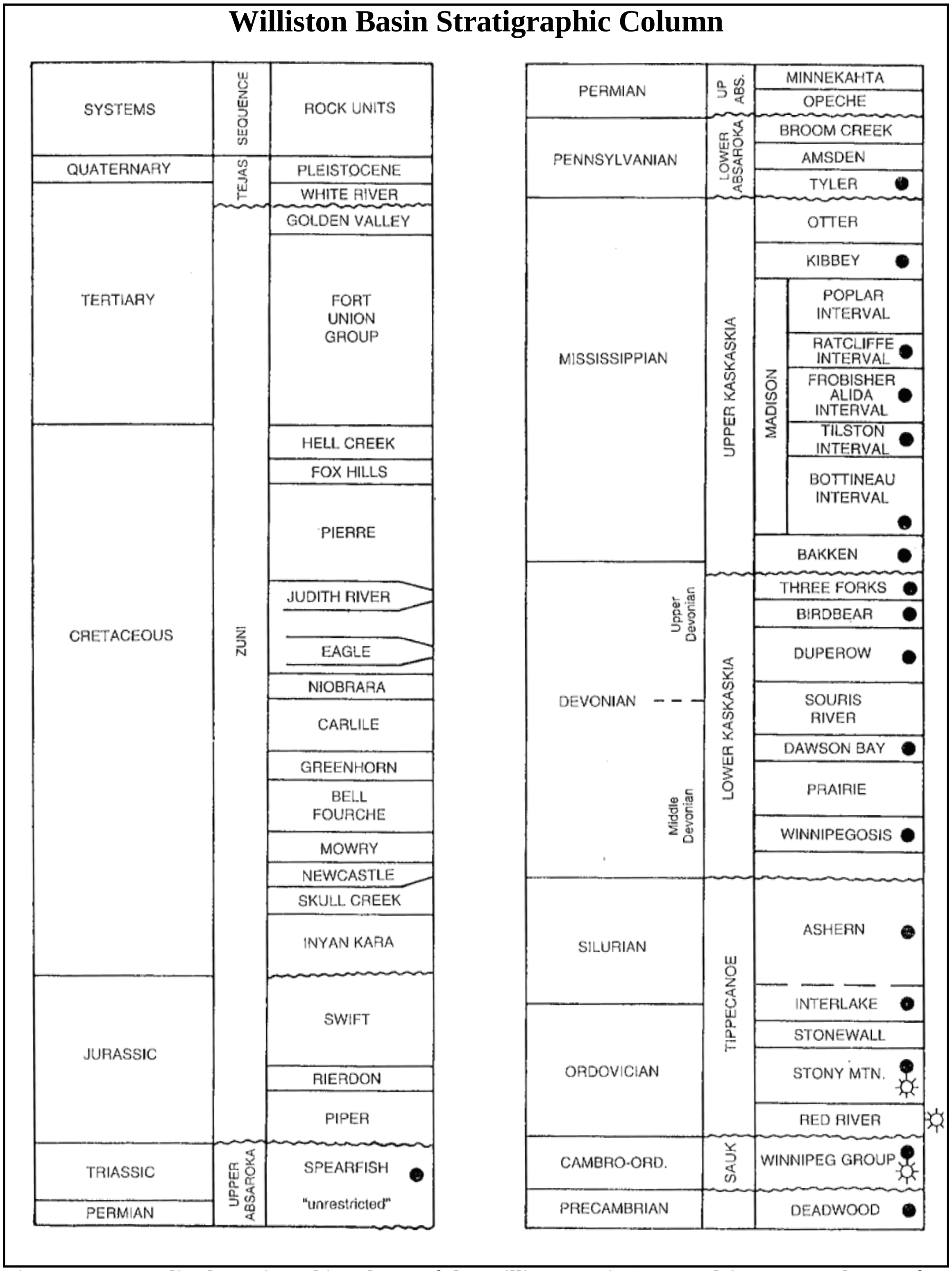

Figure 3. Generalized stratigraphic column of the Williston Basin (From Klein, 1995; redrawn after Gerhard et al., 1990). Black filled circles indicate oil producing units. Open circles with radial lines indicates natural gas producing units.

section of the Paleozoic strata, ranging from the Upper Ordovician through the Middle Mississippian, is primarily carbonate beds, with cyclic intervals of evaporites and fine grained clastics, and contains numerous oil producing units (Peterson and MacCary, 1985). Near the top of the middle Paleozoic strata is the Mississippian Madison Group, deposited during the upper Kaskaskia transgression (Klein, 1995).

The Madison Group limestone and its equivalents form thick sequences of mostly carbonate rocks covering parts of Arizona, Nevada, California, Utah, Colorado, Wyoming, Idaho, Montana, the Dakotas, and the southern Canadian plains (Sando and Dutro, 1974). Deposition occurred in shallow seas transgressing from the west, partially in response to the Antler Orogeny which occurred further west (Rose, 1976). Transgressing seas encountered stable shelf provinces to the south and north in present day Wyoming and the Saskatchewan-Alberta region (Fig. 4). Across Montana, unstable shelf conditions persisted due to the reactivation of ancient faults (Smith, 1972). In the subsurface of the Williston Basin, the Madison Group is divided into three formations (Fig. 5). The thinly bedded Lodgepole Formation is conformably overlain by the massive Mission Canyon Formation. This unit intertongues with the base of the Charles Formation, consisting of interbedded carbonates, terrigenous clastics, and evaporitic deposits, equivalent to the upper Mission Canyon in western Montana (Sando and Dutro, 1974).

Petroleum from the Goose Lake oil field is produced from the Ratcliffe interval within the Mississippian Madison Group (Figs. 3 and 5). The ~250 ma Ratcliffe interval in the Williston Basin is an informal stratigraphic unit, and includes parts of the upper

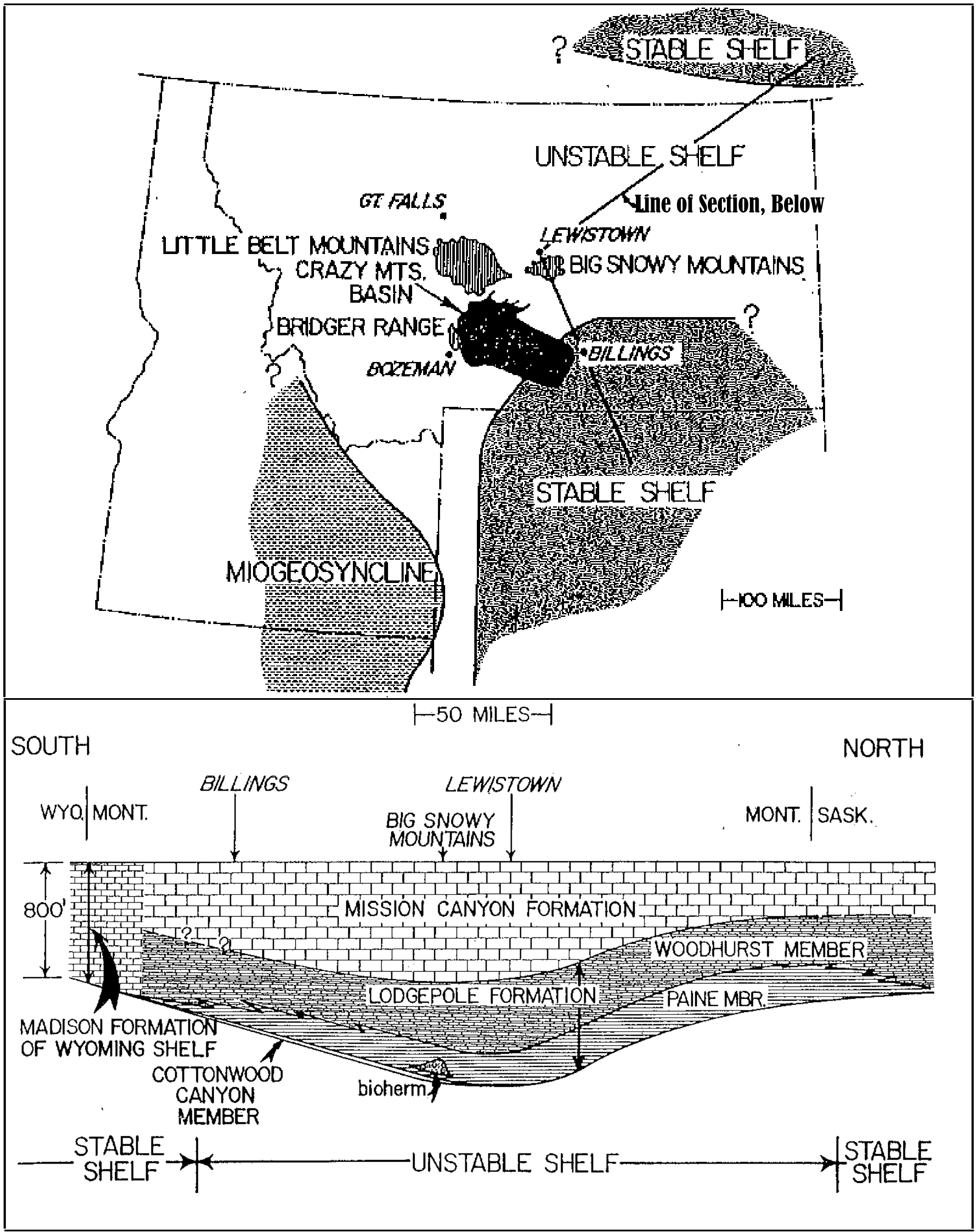

Figure 4. Upper panel shows the location map and Mississippian provinces in the Northern Rocky Mountains; unstable shelf province of central Montana is unshaded (From Smith, 1972; after Sando, 1967). Lower panel is a diagrammatic cross section of the Madison Group across the Montana unstable shelf, showing postulated and documented relationships between stable and unstable shelves (From Smith, 1972). Location of the Goose Lake oil field is roughly along the unstable/stable shelf boundary near the border separating Montana and Saskatchewan.

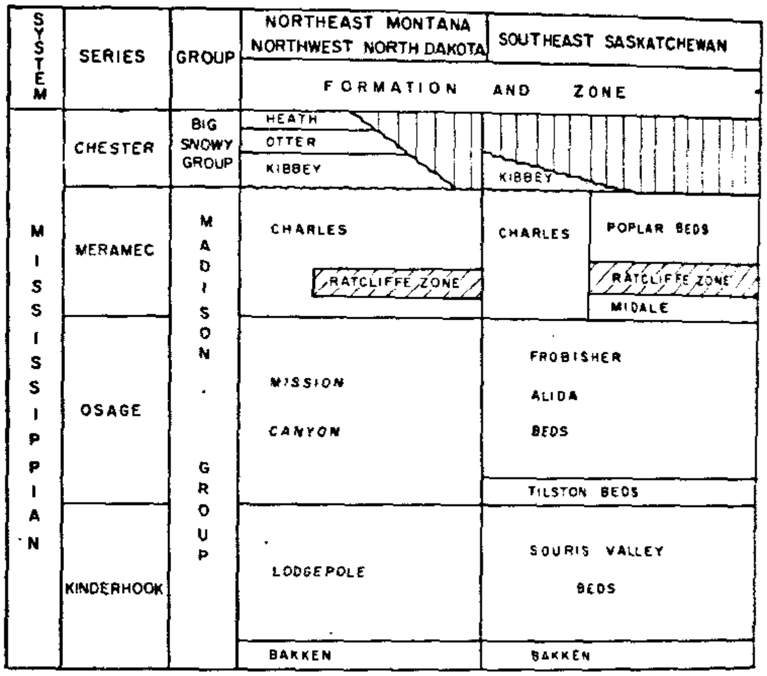

Figure 5. Stratigraphic column showing Mississippian age rocks in the subsurface of the Williston Basin (From Hansen, 1966).

Mission Canyon and lower Charles Formations (Century, 1982). Deposition occurred in the shallow and organic rich Ratcliffe Sea, an open to progressively restricted marine setting along the eastern margin of the basin (Century, 1982). Reactivation of Paleozoic-aged faults created sufficient changes in the bathymetry of the shallow sea to raise areas of the floor into the high energy zone favorable for reef development. Algal pelletoid reef growth occurred along these faults, creating the host rocks for indigenous and migrated hydrocarbons.

The first Ratcliffe oil field discovery was the Dywer reef complex in 1960, (Hansen, 1966) located fifteen miles south of the Goose Lake oil field (Fig. 6). Similar reef complexes were soon located, with the majority of oil wells producing from the Ratcliffe interval drilled in the late 1960’s. Examining the reef complex, Hansen (1966) found three main rock types associated with Ratcliffe oil production. The main

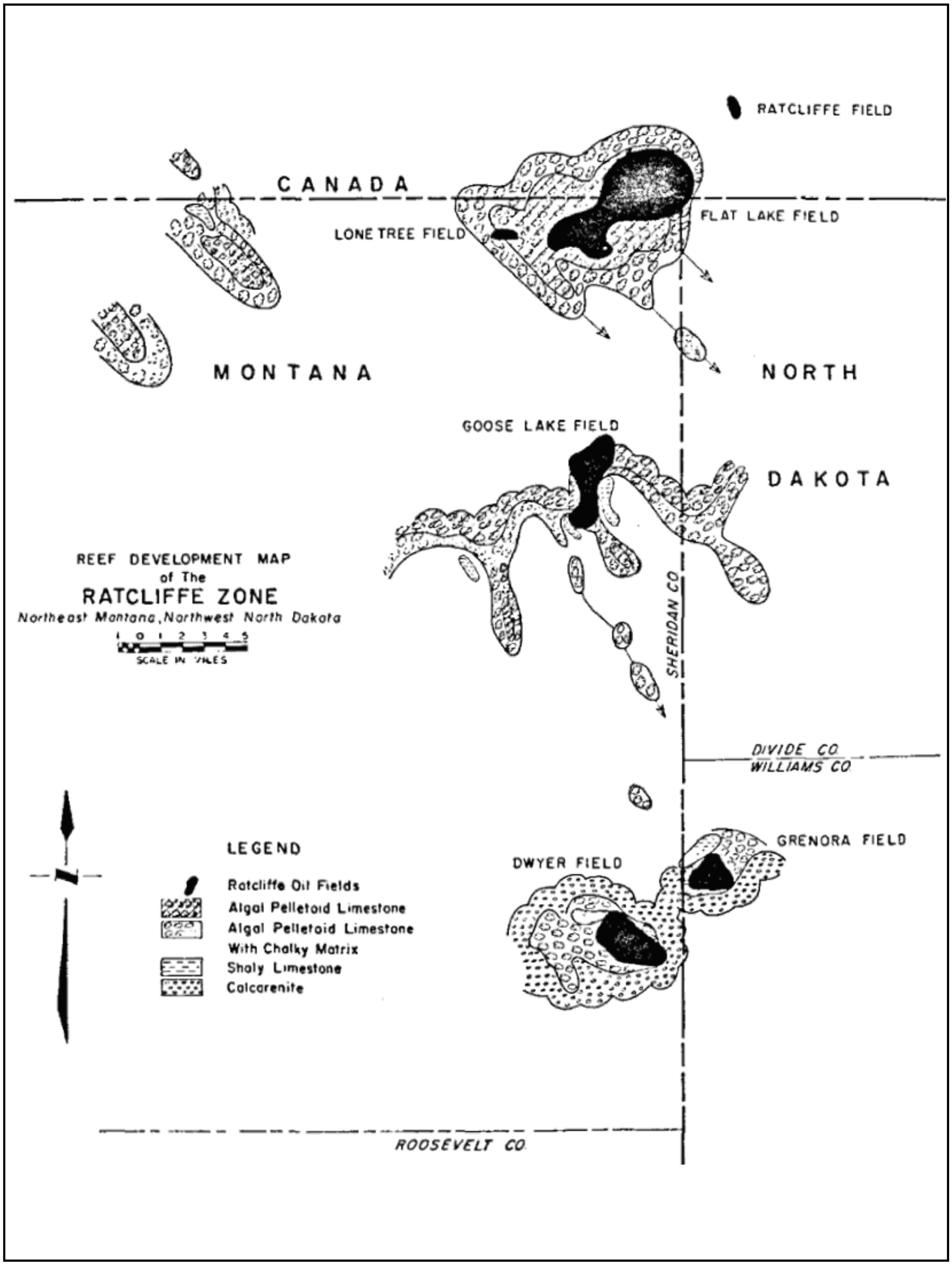

Figure 6. Reef development map of Ratcliffe zone showing various lithologic types associated with reef accumulations. Arrows are undefined in original publication (from Hansen, 1966).

component of these reefs, and the host rock for hydrocarbons, is an algal pelletoid limestone. Distinct lenses of broken and fragmented pellets indicate violent storms periodically interrupted normal pelletoid growth. Surrounding the main structure is a reef-edge calcarenite, composed of a heterogeneous mixture of algal pellets, argillaceous and shaly lagoonal deposits, and minor amounts of open marine fossils and clastic fragments. Grain sizes are largest on the southeastern edge facing into the predominant wind direction. Lastly, lagoonal shale deposits, comprised of shaly and argillaceous limestone, are found in the most protected areas of the reef complex. While not described in detail, this structure is likely similar for other reef complexes. However, Hansen (1966) shows little or no reef-edge calcarenite north of the Dywer and Grenora oil fields.

Development of petroleum resources in the Williston Basin has intensified with new assessments of recoverable oil and gas reserves in the Bakken Formation (Figs. 1, 3, and 5), and drilling successes in Montana within this Formation (Wall Street Journal, April 5, 2006). In 2008, the USGS estimated undiscovered oil and gas resources in the Bakken Formation of 3.65 billion barrels and 1.85 trillion cubic feet, respectively (USGS Pollastro et al., 2008). Despite the large oil and gas resources, all but 4 million barrels of oil and 2 billion cubic feet of natural gas are contained within continuous reservoirs, or reservoirs that lack major structural features to trap hydrocarbons. The Bakken is a thin (less than 40 m) formation consisting of two shale members separated by a middle calcareous siltstone member, with the two shale layers being the hydrocarbon source rocks and part of the continuous reservoir system (Webster, 1984). Variations in the thickness, lithology, and petrophysical properties of the middle siltstone are common and can locally enhance oil production. Due to the continuous nature of the Bakken Formation, oil and gas development requires extended reach horizontal drilling and hydraulic fracturing techniques, and will require drilling thousands of new oil wells.

Surficial Geology

The northern part of the Williston Basin is mantled by a blanket of glacial drift and hosts abundant ecological resources. Thickness of the glacial drift varies by locality, but can be hundreds of feet in Sheridan County, Montana. Lands covered by glacial sediments in this region are considered part of the Prairie Pothole Region (PPR), which spans approximately 770,000 km^2 in the north central United States and south central Canada (Fig. 1). Formed from successive Pleistocene glaciations (Fullerton et al., 2004), topography is generally hummocky with poorly developed drainage systems that result in closed or poorly drained basins (Martin and Hartman, 1987). Subsequently, the region contains thousands of shallow, depressional wetlands that provide critical wetland and grassland habitats for waterfowl, wetland and grassland birds; producing from 50 – 80% of the continent’s ducks (Batt et al., 1989).

The southwestern portion of the PPR, which includes eastern Sheridan County, Montana, is superimposed over the Missouri Coteau. The Missouri Coteau is a physiographic region southwest of the Missouri escarpment, characterized as an elevated highland above the Central Lowlands (Fig. 1). As the advancing ice sheets overrode the NW – SE trending escarpment, large quantities of fine grained sediment in the basal ice were carried onto the highlands (Eyles and Menzies, 1983). The elevated nature of the

Missouri Coteau at the end of the Pleistocene resulted in the stagnation and ablation of the ice, leaving behind widespread ground moraine. Stagnation processes, including melt-out, fluvial outwash, and sediment slumping are responsible for the hummocky landscape, characterized by unintegrated drainage, present in the region. Numerous pothole wetlands are the byproduct of depressions created by the persistence of large ice remnants within the stagnating ground moraine (NDGS 2009). The regional near-surface stratigraphy is therefore a complex, three dimensional mixture of poorly sorted clay-loam till, fine grained glaciolacustrine deposits, and coarse grained glacial outwash channel deposits (Grisak and Cherry, 1975; Miall, 1983; Hendry, 1988; Fullerton et al., 2004). A thorough description of the near surface stratigraphy at the Goose Lake detailed study area is provided in the Study Area and Hydrogeologic Investigations section.

From X-ray diffraction and petrological investigations, Mills and Zwarich (1970) concluded that the mineralogy of the lacustrine silts and clays and the clay-loam tills are essentially identical. This stands to reason as the predominant sediment input to these lacustrine environments was from the fluvial reworking of clay-rich tills. The majority of the clay minerals belong to the montmorillonite group with minor amounts of chlorite, vermiculite and kaolinite. The lithic fragments consist mainly of Precambrian gneiss and schist, Paleozoic carbonates and dolomites, and Mesozoic shale and siltstones (Grisak and Cherry, 1975). Despite local variations in textural and mineralogical properties, glacial tills of the Great Plains are laterally homogenous over large geographic areas (Fullerton et al., 2004).

Last glacial tills of the Interior Great Plains are composed of two distinct zones: an upper, weathered and vertically fractured segment and a lower unweathered and unfractured zone (Hendry, 1982; Keller et al., 1986). The origin of vertical fractures in the tills of the Great Plains is enigmatic, but likely relates to crustal extension from isostatic rebound during post glacial time, resulting in the upward propagation of vertical fractures in the underlying bedrock (Grisak et al., 1980). The fractures in the weathered zone, which have an average spacing of 150 mm, are commonly lined with carbonate and oxide precipitates and are continuous across the boundaries between glaciolacustrine and till deposits (Grisak and Cherry, 1975). Depth of the weathered till is variable, commonly 5-10 m, and often extends below the present water table (Keller et al., 1986). The occurrence of carbonate and oxide precipitates in fractures, and the presence of tritium, produced during nuclear weapons testing after 1954, in shallow groundwater samples indicate the weathered till is hydrogeologically active (Hendry, 1988). Fractures in the weathered zone result in bulk hydraulic conductivities up to three or four orders of magnitude greater than the same, unfractured till (Grisak and Cherry, 1975; Hendry 1988). The interconnected nature of fractures in the weathered till results in lateral groundwater velocities that are up to seven times that of vertical velocities, and therefore the lateral movement of contaminants in areas covered by glacial till is of most concern (Murphy et al, 1988; Hendry, 1988). Keller et al. (1986) also provided evidence of vertical fracture networks existing in the unweathered zone at a till site in Saskatchewan. Despite the presence of fractures in the unweathered till, the swelling of smectitic clays in the constantly saturated sediments decreases fracture permeability (Murphy et al., 1988).

Continental ice sheet lobes that terminate on land release vast quantities of meltwater that can erode, rework, and redeposit previously emplaced glacial sediments (Miall, 1983). Glaciofluvial reworking and deposition often begins underneath the ice sheet as high energy meltwater streams, flowing parallel to ice movement, develop beneath the ice margin. In areas where the substrate is fine grained sedimentary strata, these streams commonly deposit sands and gravels into channels cut into the underlying lodgement till (Clark and Walder, 1994). Changes in ice-flow patterns and meltwater channel configuration results in channels being overlain and interbedded with younger tills, creating a locally complex stratigraphy. In addition, rapid rates of aggradation have been reported for glacial meltwater streams (Maill, 1983), with the burial of the thinning ice margin by outwash deposits observed for many ice sheets and glaciers. As the buried ice melts, the mantling outwash sediments collapse unto the underlying till, creating collapsed outwash deposits. The presence of glaciofluvial deposits effect the bulk hydrogeologic properties of glacial sediments, as the hydraulic conductivity of coarse gravels ranges from $9 \times 10^{-7} – 6 \times 10^{-3}$ m/s compared to $1 \times 10^{-12} – 2 \times 10^{-6}$ m/s for fractured and unfractured glacial till (Schwartz and Domenico, 1998). Lloyd (1983) reported hydraulic conductivity values of $1 \times 10^{-5} – 1 \times 10^{-9}$ m/s and $1 \times 10^{-2} – 1 \times 10^{-5}$ m/s for variable lithology till (containing sand and gravels) and sands and gravels, respectively. Thus, coarse grained deposits within and overlying till sheets provide preferential flow paths for groundwater and contaminants. Near-surface sediments at the Goose Lake detailed study area are composed of a blanket of collapsed outwash, overlying a hummocky, pebbly clay till.

Formational Brines and Co-Produced Waters

Saline formational waters have been co-produced with oil and gas developments in much of the United States, with chemistries and salinities varying drastically by location (Fig. 7). The formation of marine derived brines in intracratonic basins are the result of specific climatic and eustatic or tectonics changes that allowed for the evaporative concentration of seawater during sediment deposition (Jensen et al., 2006; Warren, 2010). The chemistry of brines is controlled by the initial brine source, subsequent diagenetic processes, and hydrogeology (Iampen and Rostron, 2000; Warren, 2010). Local and basin-wide variations in these parameters result in the subsurface spatial distribution observed within a given basin. Many of the previous studies of

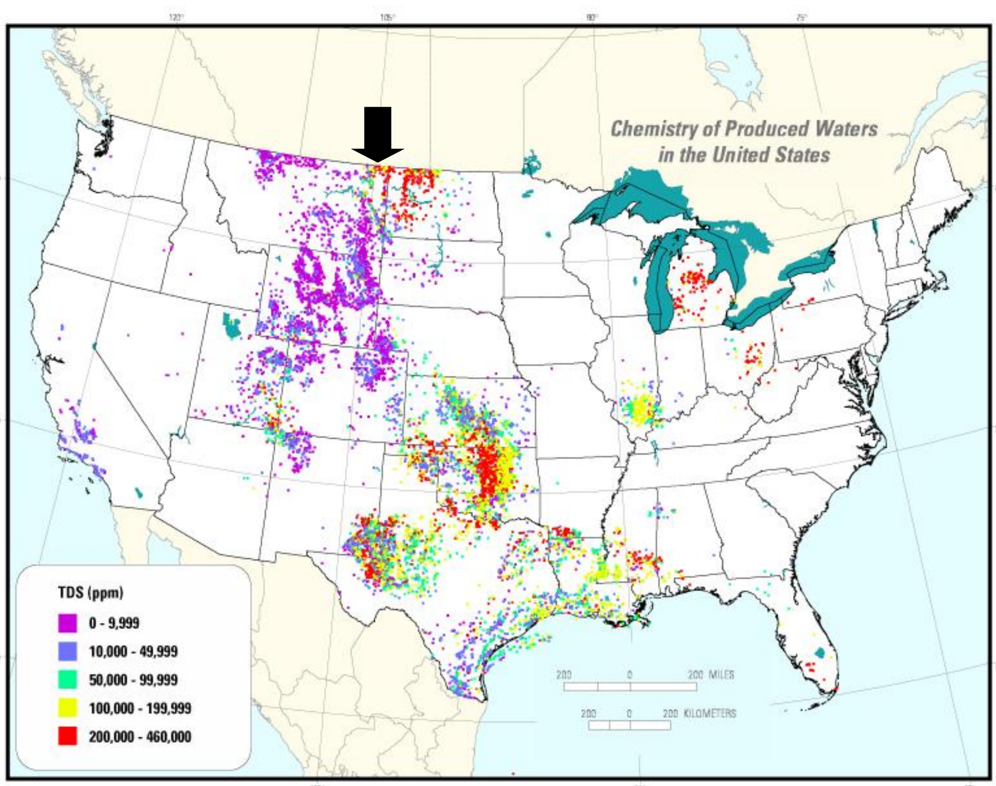

Figure 7. Map showing the Total Dissolved Solids in produced waters from oil and gas development in the United States. Black arrow shows approximate location of Sheridan County, MT (By Tracey Mercier, From http://energy.cr.usgs.gov/prov/prodwat/tds.htm accessed 11/11/10).

formational brines within the Williston Basin have focused on pre-Mississippian strata (Iampen and Rostron, 2000; Rostron and Holmden, 2003; Jensen et al., 2006) as the majority of the mineral resources (oil, gas, and potash) are in strata this age or older.

Saline brines within a given formation can be derived from groundwaters that have been altered through post depositional water-mineral interactions and/or residual seawater trapped as pore waters within sediments during deposition. Intracratonic basins generally contain numerous evaporite beds, commonly composed of halite and other minor salts. An example is the Devonian halite-and-potash-bearing Prairie Evaporite in the Williston basin, which is locally 200 m thick (Rostron and Holmden, 2003). Dissolution of evaporites by pore waters after deposition is common, and dissolution of the Prairie Evaporite has historically been accepted as the origin of pre-Mississippian brines in the Williston basin (Chipley and Kyser, 1991) as several of these brines have ionic compositions indicative of halite dissolution.

Intracratonic basins are often restricted, resulting in evaporatively concentrated seawater which can be incorporated into the formation during deposition. Using Cl/Br vs. Na/Br molar ratios, Iampen and Rostron (2000) showed that residual seawater remains in the Williston Basin, and that brines derived from subaerial evaporation of seawater are not confined to single aquifers. They cited the presence of chemically equivalent waters above and below a major aquitard as indicative of vertical mixing of brines from evaporated sea water and those from halite dissolution within the Williston basin. Therefore, the exact formational history of individual brines is difficult to ascertain, however, both of these systems can produce extremely concentrated brines.

Major ion chemistry and oxygen isotope studies (Iampen and Rostron, 2000; Rostron and Holmden, 2003) indicate three chemically different pre-Mississippian brine zones within the Williston basin developed in response to basin-wide, topographically controlled, groundwater flow initiated by Laramide uplifts to the west (Bachu and Hitchon, 1996). In the recharge zone, along the western edges of the Williston basin, there is a zone of brackish Ca – SO_4 dominated water with TDS values averaging < 30,000 mg/L. In the central part of the basin, a slow moving to stagnant, Na – Ca – Cl brine with anomalously low sulphate and TDS values > 300,000 mg/L is present. Along the northern end of the basin, TDS values range from 100,000 – 200,000 mg/L and Na – Cl is the major cation-anion pair. Furthermore, unlike many intracratonic basins, isotopic compositions do not exhibit a systematic increase with depth in pre-Mississippian Williston Basin brines (Rostron and Holmden, 2000). The authors note this trend is seen in "shallower formations in the Williston basin" but do not delineate the boundary.

Much of the oil and gas production in the Williston Basin occurs in the basin center, within the stagnant groundwater flow system, and co-produced brines are extremely saline (Fig. 7). From 50 actively producing oil wells, Iampen and Rostron (2000) documented an average total dissolved solid (TDS) concentration of 300,000 mg/L, with some samples exceeding 380,000 mg/L. Analysis of ten co-produced water samples from the Ratcliffe interval in 1989 (Reiten and Tischmak, 1993) and one from 2009 show a range of TDS and chloride contents (Table 1). All samples in Table 1 are from the Ratcliffe production interval; however, only four are from the Goose Lake oil field. Mean TDS and chloride concentrations for all eleven samples are 208,091 (range

98,282 – 310,000) mg/L, and 119,444 (range 33,200 – 190,000) mg/L, respectively. Equivalent values for the four Goose Lake samples are 233,270 (range 177,098 – 310,000) mg/L and 137,900 (range 107,000 – 190,000) mg/L. From the limited data available, it appears that formational waters within the Goose Lake oil field are less saline than the average formational waters within the Williston Basin which may be due to mixing of the stagnant and brackish groundwater flow zones described above.

Table 1. Chloride and Total Dissolved Solid concentrations for eleven produced water samples from the Ratcliffe interval. All samples were collected in 1989 with the exception of the sample in italics, which was collected in 2009 (Data from Reiten and Tischmak, 1993). Modern seawater is shown for reference.

Oil Field	Formation	Chloride (mg/L)	TDS (mg/L)
Flat Lake	Ratcliffe	116,660	195,538
Flat Lake	Ratcliffe	57,755	98,282
Flat Lake	Ratcliffe	33,200	140,216
Flat Lake	Ratcliffe	143,400	239,390
Flat Lake	Ratcliffe	152,600	254,651
N. Goose Lake	Ratcliffe	139,543	230,912
N. Goose Lake	Ratcliffe	119,122	196,933
Goose Lake	Ratcliffe	133,600	222,980
Goose Lake	Ratcliffe	107,000	177,098
Goose Lake	Ratcliffe	190,000	310,000
Goose Lake	*Ratcliffe*	*121,000*	*223,000*
Average: All		119,444	208,091
Average: Goose Lake		137,900	233,270
Modern Seawater		19,000	35,000

Formational brines are brought to the surface during oil well drilling and then continually during production. When a well is being drilled, formational waters within overlying strata and drilling mud and fluids are stored in temporary reserve pits. Due to the presence of salt beds overlying many of the target zones in the Williston Basin, salt-based drilling fluids are often employed to prevent washout, increasing the salinity of waters stored in the reserve pit. During drilling, between 54,000 and 90,000 cubic feet of salt-saturated drilling fluid and co-produced water is contained within the reserve pit. Once the well is completed, saline waters are continually extracted and must be separated from the oil before disposal. The ratio of brine to oil produced depends on the well, but usually increases over time with ratios over 2:1 not uncommon (Jon Reiten, pers. comm., 2009). Due to the sheer volume of brine produced during oil and gas production, elaborate networks of pipelines, tanker truck routes, and tank batteries (a group of large storage tanks that receive co-produced water and oil from several near-by wells) must be constructed. After separation from the oil, co-produced waters are generally trucked or pumped to injection well sites for disposal. The primary pathways for saline brines to contaminate surface and groundwater resources are infiltration from reserve pits, injection well failures, pipeline line breaks, and improper disposal methods.

Contaminant Sources and Environmental Recognition

The earliest evidence of environmental degradation caused by co-produced brines was reported in Kansas in 1932; however, clear documentation of the problems related to brine contamination were not available until the 1960's (Murphy et al., 1988). Since then, numerous effects associated with the decline in surface and groundwater quality surrounding oil field activities on ecosystem dynamics and agriculture production have been reported (Mosley, 1983; Murphy, 1983; Murphy and Kehew, 1984; Beal et al., 1987, Murphy et al., 1988). Once co-produced waters have been introduced to the environment, climatic and geologic controls determine the length and extent of contamination, unless remediation action is taken. For example, at an oil field within highly permeable sediments in Alabama that also receives heavy rains, saline groundwater contamination from oil and gas development was observed to rapidly decrease over a 10 year period (Powell et al., 1973). However, in the arid PPR, which is underlain by relatively low permeability, clay-rich tills, contamination from the 1960's is still evident and is expected to persist for tens to hundreds of years (Murphy et al., 1988).

The majority of saline contamination throughout the Williston Basin and within Sheridan County, Montana, is a legacy inherited from improper storage and disposal of co-produced waters and drilling mud (Beal et al, 1987; Murphy 1983; Murphy and Kehew, 1984, Reiten and Tishcmak 1993). Much of the saline contamination is the result of leachates generated from reserve pits, which are excavated at each oil well site to separate drilling mud and cuttings and/or hold and evaporate co-produced water. Soluble salts and exchangeable sodium ions are the most motile constituents that leach from reserve pits, and are also the most detrimental to plants and soils (Mosley, 1983). The average reserve pit measures 150 feet long by 60 ft wide by 10 ft deep and contains between 54,000 and 90,000 cubic feet of salt-saturated drilling fluid (Murphy and Kehew, 1984). Prior to 1974, reserve pits were often unlined or lined with bentonite to prevent seepage, which failed to adequately contain saline fluids. In addition to reserve pits, other pathways exist for co-produced waters to enter the environment including; uncontained discharges (dumping), injection well failures, corrosion of abandoned well casings, and breaks in pipelines that transport oil and co-produced waters to treatment and injection facilities (Reiten and Tischmak, 1993, Thamke and Craigg 1997). Therefore, while the majority of contamination is from pre-existing reserve pits, there still exists the potential for new sources of saline contamination in the PPR.

Strategies for storage and disposal of co-produced waters and drilling fluids have changed through time with the recognition of the environmental damage of reserve pits (Murphy and Kehew, 1984). In all disposal scenarios, the low viscosity liquid is pumped out and transported to an injection well, leaving behind a highly saline slurry. In the 1950's and early 1960's, excavated materials were gradually added from the sides, covering the remaining fluids. Due to the small area of the reserve pit and inability of the fluids to desiccate rapidly, this process could take anywhere from a month to a year. Reclamation strategies were later modified to include trenches in a practice known as spider legging. In this now abandoned process, a backhoe was used to create trenches away from the retired pit and the depression was backfilled, forcing the contents to drain outward into the trenches. This greatly decreased the time required for reclamation (Beal et al., 1987). In both situations, no barrier exists between the buried saline slurry and the underlying sediments. The average reserve pit contains roughly 260 tons of salt upon burial (Reiten and Tischmak, 1993). Current reserve pits in the United States are lined with plastic and modern reclamation practices still rely on burial, except now the plastic liner is folded over the remaining slurry and often a clay cap is used instead of the original earth material. However, the liners may be ripped or punctured as they are manipulated with heavy machinery. Alternative strategies for the storage of co-produced waters and drilling fluids do exist and are employed in the Canadian portion of the Williston Basin, where all new oil wells are drilled without a reserve pit. During pit-less drilling, tanker trucks replace the reserve pit, performing the separation of drilling fluids and cuttings. Therefore, the use of pit-less drilling reduces the potential for surface and groundwater contamination from co-produced brines by eliminating a prospective contaminant source.

The rate of leachate generation from abandoned, unlined reserve pits depends on the initial quantity of salt produced at a given site, local precipitation, topography, and hydraulic conductivity (Murphy et al., 1988). Investigations of leachates generated from an oilfield site within a geologically similar setting in North Dakota (Murphy et al., 1988) revealed that pore waters within the saturated and unsaturated zones below the abandoned reserve pits have the same ionic composition as the waters in the reserve pits 10-25 years ago. Examining solute transport through fractured till, Grisak et al. (1980), demonstrated that Cl is significantly retarded compared to the average fracture flow water velocity, with the retardation likely due to diffusion of the solutes into the porous media of the till. Due to the limited number of major recharge events in the region, minimal flushing of the brine has occurred as evidenced by the ionic chemistry of pore waters. As a result, it is estimated that brine contamination from old reserve pits in this region will persist for tens to hundreds of years (Murphy et al., 1988).

The presence of fractures and sand lenses within the glacial drift covering the Great Plains results in the preferred lateral transport of contamination in the upper, or weathered, section of the glacial tills (McKay et al., 1998). Vertical fractures are common in the tills of the Great Plains, yet swelling of the smectite clays in the unweathered (saturated) till reduces fracture permeability. This swelling in the permanently saturated zone retards downward leachate movement compared to lateral transport within the unsaturated (weathered) till, where wetting and drying cycles increases fracture permeability and subsequent contaminate transport (Grisak and Cherry, 1975; Murphy et al., 1998). The saline contaminated groundwater beneath the reserve pits examined by Murphy et al., (1998) was mounded 7 ft higher than the regional water table for a radius of 500 ft around the reserve pit boundaries, creating a hydraulic gradient capable of dispersing leachates. Depth of contamination was roughly 70 feet, however, a sharp decline in Na and Cl ions (~85%) was observed below 18 ft, corresponding to the color contact between the weathered and unweathered till. Additionally, the maximum Na and Cl concentrations occurred directly above this contact. The lateral rate of contaminant transport at this site was over seven times greater than the vertical contaminant transport rate (=500/70), and the lateral transport within the weathered till was almost ten times the vertical transport within the unweathered till (=500/(70-18)).

In 1989, Jon Reiten of the MBMG and Terry Tischmak of the Sheridan County Conservation District (SCCD) analyzed numerous water samples and conducted geophysical surveys to assess co-produced water contamination in Eastern Sheridan County, Montana. The introduction of co-produced brines drastically altered the saline chemistry of surface and groundwater resources as evidenced by trilinear diagrams showing analysis of 43 contaminated and uncontaminated groundwater samples from glacial till, glacial outwash deposits, and one surface water sample from Goose Lake (Fig. 8). Panels 8-A and 8-B clearly show the shift to the right on a trilinear diagram seen in water samples contaminated by co-produced waters, with the magnitude of the shift corresponding to the degree of contamination. Sodium and chloride are the dominant

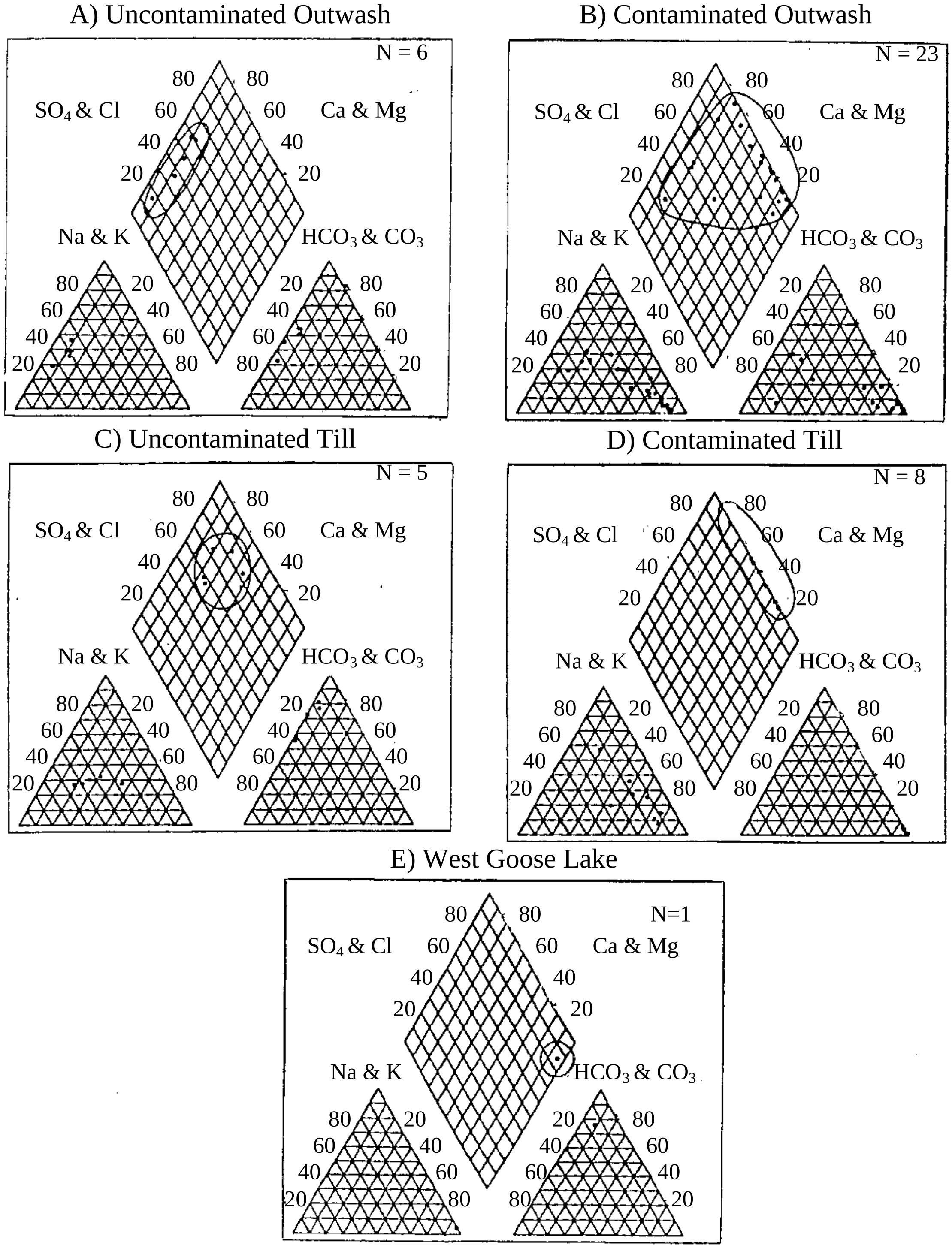

Figure 8. Trilinear diagrams developed from 43 water quality analyses from eastern Sheridan County, Montana, indicating percent reacting values of major ions in groundwaters from A) Uncontaminated outwash, B) Contaminated outwash, C) Uncontaminated till, D) Contaminated till, and E) Surface water from West Goose Lake (from Reiten and Tischmak, 1993).

ions in contaminated outwash samples; however, significant amounts of calcium, magnesium and bicarbonate are present in the minimally contaminated samples (Reiten and Tischmak, 1993).

This region is known to have surface and shallow groundwaters with elevated TDS concentrations, however, the chemistry of naturally saline waters are distinctly different than the deep groundwater produced during oil and gas development. Within the shallow groundwater system, hydrologic position in the landscape often controls natural water chemistries, with TDS increasing from areas of recharge to areas of discharge (Swanson et al., 2003). Consequently, surface water and shallow groundwater in discharge areas are often enriched in sulfate and bicarbonate and depleted in chloride, resulting in sulfate and carbonate dominated waters (Custer, 1976). In contrast, the deeper formational groundwaters (co-produced waters) are enriched in chloride, resulting in surface and groundwaters contaminated by co-produced waters being dominated by chlorides. Although both of these systems can produce surface and groundwaters that have extremely high TDS values, the differences in water chemistry allow for easy identification of contaminated water resources.

In order to document saline contamination from oil field activities, Reiten and Tischmak (1993) developed a rapid assessment tool to distinguish natural waters with extremely high TDS values and those influenced by co-produced waters. This rapid assessment tool is referred to as the contamination index (CI) and is defined as the ratio of chloride concentration in mg/L, to the specific conductance in μS/cm, in a water sample. The CI allows for field and laboratory determination of chloride contamination across a wide range of water chemistries, from pristine to extremely saline waters. Additionally, the CI remains relatively stable in groundwater wells that develop vertical density gradients. For eastern Sheridan County, Montana, water resources with a CI value greater than 0.035 are considered contaminated by oil field brines (Reiten and Tischmak, 1993).

In addition to a chemical signature, saline contaminated groundwater plumes also produce a geophysical signature (Reiten and Tishcmak, 1993; Thamke and Craigg, 1997). Due to the large quantities of salts present in co-produced waters, contaminated groundwater plumes exhibit elevated apparent conductivities in electromagnetic (EM) surveys. Equipment used for these types of surveys relies on the principle of electromagnetic induction, a process described in detail in Geonics Technical Notes (McNeill, 1980a, 1980b). However, the basic operation of EM instrumentation involves two coils, a transmitter and receiver coil, separated by a known distance (Fig. 9). The transmitter coil is energized with an alternating current, inducing small currents in the earth, which in turn create a secondary magnetic field. The receiver coil senses both the primary and secondary magnetic fields, converting the time- varying signals into electrical conductivity. EM readings are expressed as apparent conductivity which is an integration of all the subsurface material sensed by the equipment. The dipole is orientated 90^{o} to the coil and can be positioned vertically, as in Figure 9, or horizontally for most geophysical instruments. Enhanced subsurface characterization is obtained by changing the dipole orientation and/or the coil spacing, resulting in different depth integration curves and exploration depths (McNeill,1980b). However, the effective

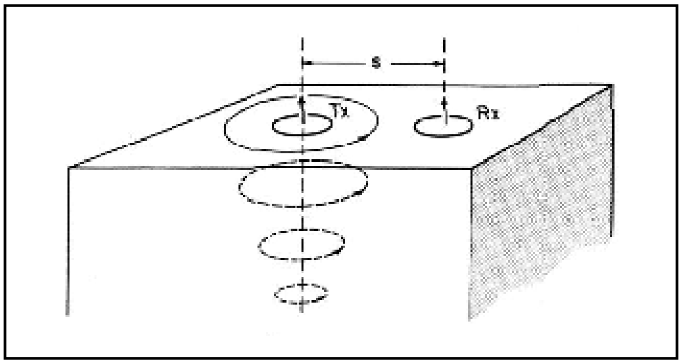

Figure 9. Diagram of the principal behind the Geonics EM-31 and EM-34 geophysical equipment. Tx is the transmitter coil, Rx is the receiver coil, and s is the intercoil spacing (from McNeill, 1980a).

exploration depth decreases for all EM equipment as the conductivity of the medium decreases (Greenhouse and Slaine, 1986).

Electromagnetic surveys measure the electrical properties of pore waters and the soil matrix, and are capable of detecting conductive saline plumes in both the saturated and vadose zones. The textural composition and degree of soil saturation can effect apparent conductivity measurements, with greater clay and water contents increasing conductivity. However, in the arid climate of northeastern Montana, soil salinity should be the most dominant variable contributing to apparent conductivity (Corwin and Lesch, 2005). In areas contaminated by co-produced waters, the highly elevated values recorded in EM surveys are primarily the result of dissolved solids, namely sodium and chloride, in saline contaminated groundwater and soil (Reiten and Tischmak, 1993). Conversely, natural groundwaters with high TDS from sulfate salts can also give elevated readings.

Study Area and Hydrogeologic Investigations

During initial hydrologic investigations in 1975, at what would become the Goose Lake detailed study area, surface water was sampled from dugouts down-gradient of the

Hammer M tank battery (Fig. 2). All samples revealed elevated chloride levels, with the furthest sampled point, roughly 0.67 of a mile below the Hammer M tank battery, containing 940 mg/L chloride (Reiten and Tischmak, 1993). Three groundwater monitoring wells were installed in 1975 and also yielded elevated chloride levels. In 1989, Reiten and Tischmak (1993) selected the area in and adjacent to T36N, R58E, S27 as the location for an in-depth hydrogeologic investigation as part of the eastern Sheridan County saline contamination assessment, denoting the site the Goose Lake detailed study area. Contamination at the Goose Lake detailed study area is due to oil field practices employed in the past and surface discharges of co-produced waters summarized in the excerpt below (Reiten and Tischmak, 1993 p. 91).

> **The obvious sources of contamination in the Goose Lake oil field (detailed study area) are the individual oil-field sites. Environmentally unsound methods of brine disposal that were commonly used at these sites prior to 1975 are the main causes of the existing contamination. The use of evaporation pits, trenching of reserve pits, and pipeline leaks all have contributed to the brine contamination. It is unlikely that the large extent and high level of surface-water and ground-water contamination existing in the Goose Lake Field could be derived from these relatively small volume sources of brine. Much of the existing contamination appears to confirm landowner reports that large volumes of brine were disposed of by simply allowing the brine to flow onto the ground surface.**

The USFWS acquired the southern half section of T36N, R58E, S27 in August of 1999 and designated it a Waterfowl Production Area (WPA). The Northeast Montana Wetland Management District (NMWMD) is responsible for managing forty four WPAs and their associated wetlands in the region. In order to determine the potential for brine contamination to these ecologically sensitive areas, Nelson (pers. comm., 2009) used archived aerial photos from the USDA Farm Services Agency in Plentywood, Montana to identify the locations of numerous active and abandoned reserve pits, oil wells, and tank batteries. Active or abandoned oil wells and associated reserve pits were documented inside the boundaries of fourteen WPAs and within a half mile of another sixteen, accounting for 68% of the WPAs managed by the NMWMD. Of these thirty sites, twenty nine had chloride indices from groundwater monitoring wells indicating contamination from co-produced waters. Contamination indices were also performed on all wetlands that contained water, with fifty percent exhibiting contamination (Nelson pers. comm., 2009). Therefore, the Goose Lake detailed study area is likely representative of other oil field sites located in glacial outwash deposits in the PPR where wildlife conservation efforts contend with historical and potential new sources of chloride contamination to surface and groundwater resources from oil and gas development.

1989 Hydrogeologic Investigations

As part of the detailed hydrogeologic investigation of eastern Sheridan County, Montana, Reiten and Tischmak (1993) produced a geologic map showing surficial glacial sediments and oil field sites (Fig. 10). Additionally, they created a hydrogeologic map (Fig. 11) and a series of cross sections (Fig. 12) illustrating the hydrogeology and subsurface sediment relationships at the Goose Lake detailed study area. The Goose Lake detailed study area (red box in Figure 10) contains a contact between glacial till to the northwest and outwash deposits in T36N, R58E, S27 (Fig. 11). Surficial sediments within the section are comprised of collapsed outwash, deposited directly on to underlying ice, with locally interbedded and mantling deposits of silt and clay from glacial and modern lacustrine environments (Reiten and Tischmak, 1993). As the ice

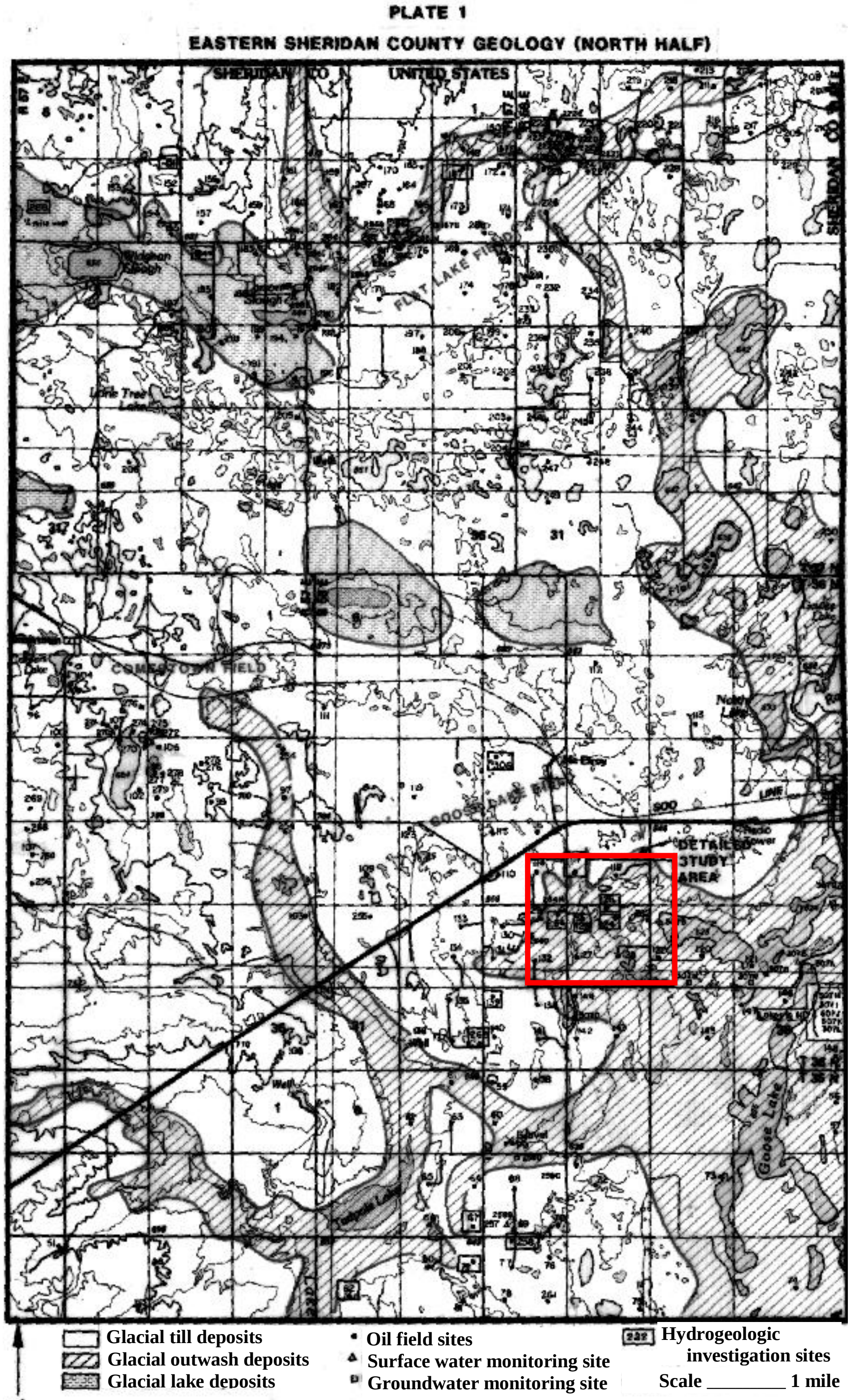

Figure 10. Plate 1 of the surficial glacial sediments and oil field sites map of eastern Sheridan County, Montana (from Reiten and Tischmak, 1993). Red box outlines the detailed study area.

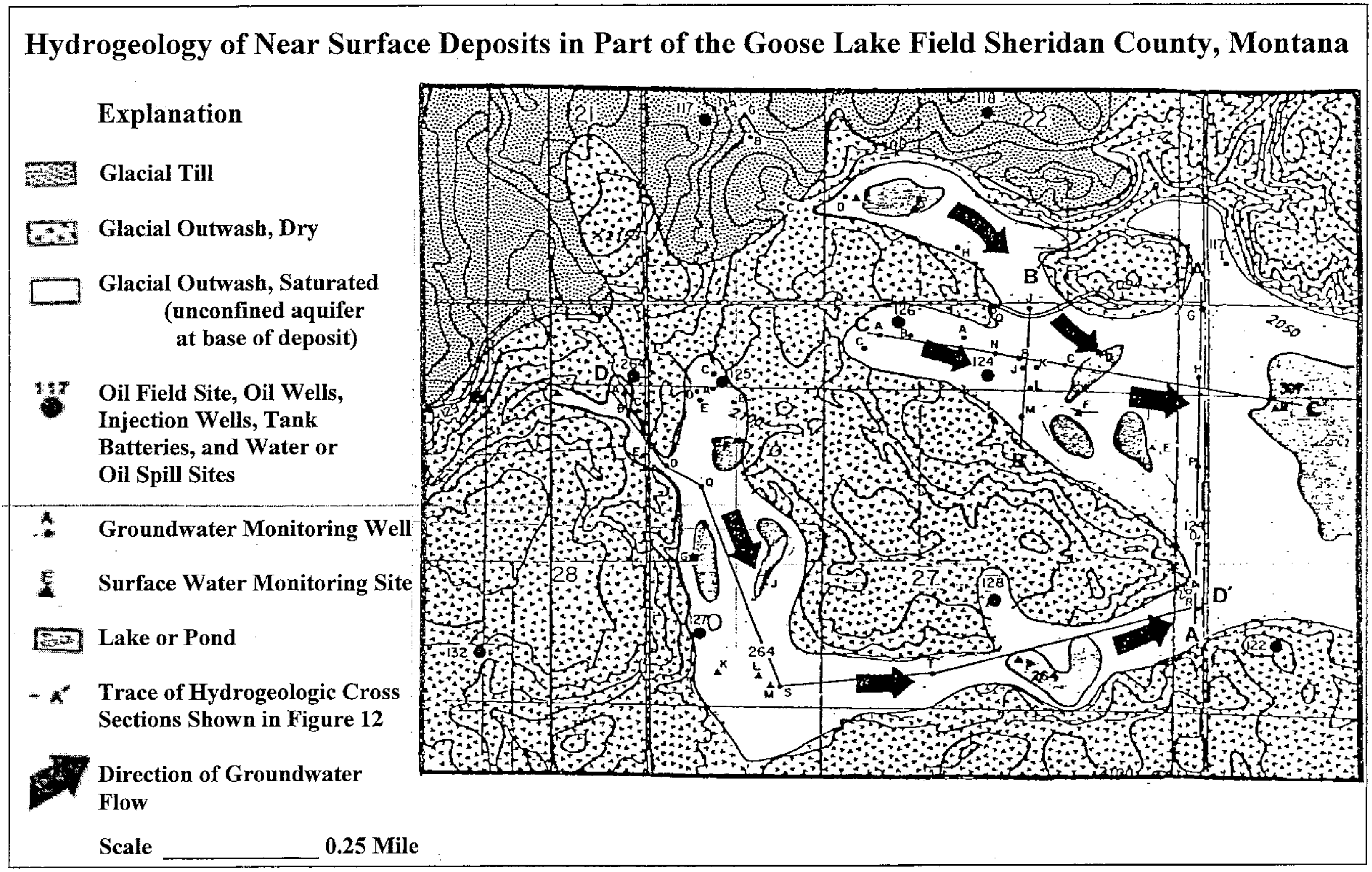

Figure 11. Hydrogeologic map of the Goose Lake detailed study area showing oil field sties, groundwater monitoring wells, and groundwater flow paths

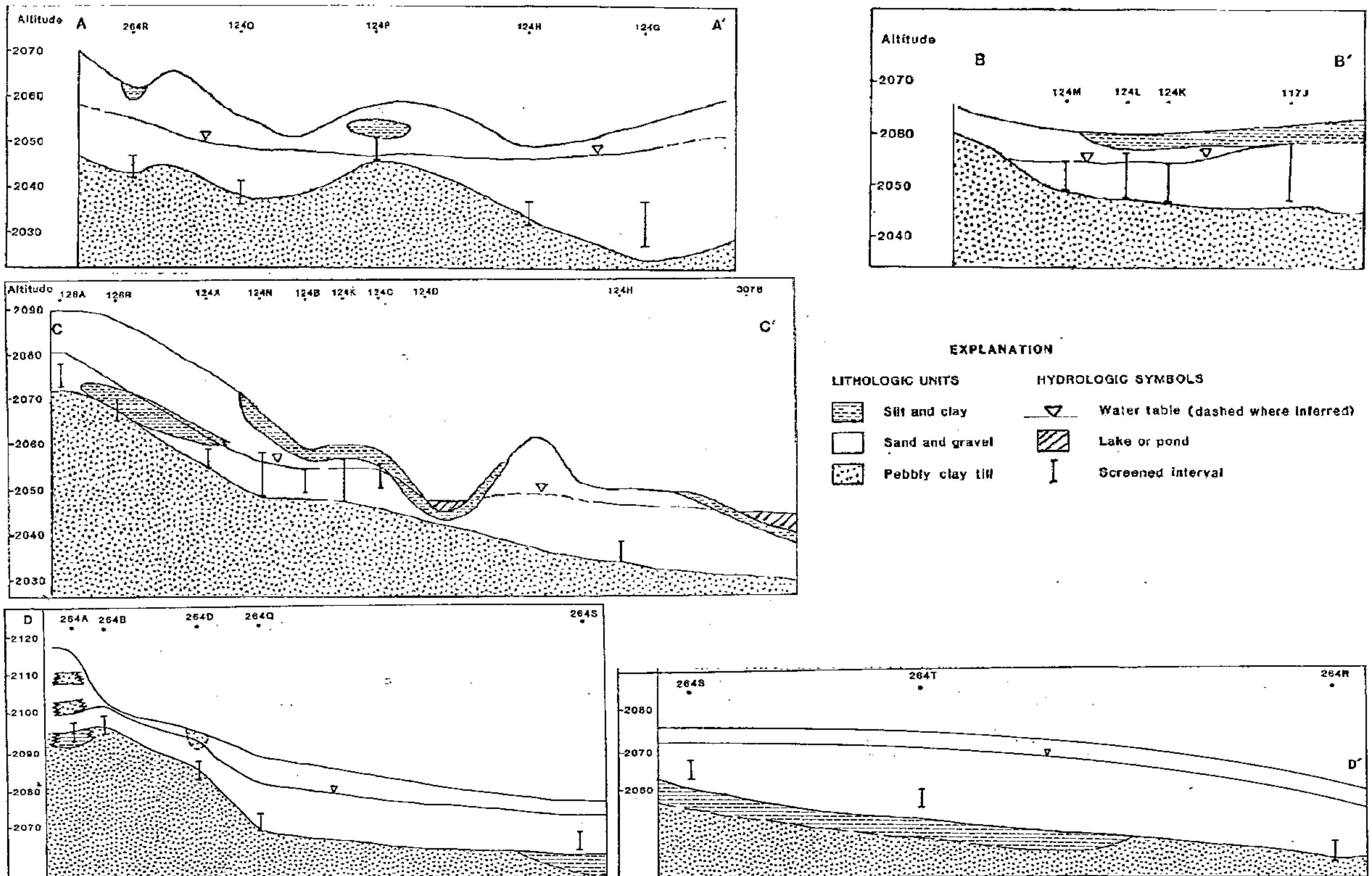

Figure 12. Cross sections showing surficial sediments and groundwater wells at the Goose Lake detailed study site. Locations of cross sections lines are shown in Figure 11 (From Reiten and Tischmak, 1993).

meltwater stream drainages that deposited the sediments but can be influenced by sediment slumping, controlled by the relative thickness and persistence of the underlying ice, and modern runoff dynamics.

At the Goose Lake detailed study area, these processes have resulted in an undulating blanket, averaging 15 ft thick, of moderately well-sorted to very poorly sorted silty sand and gravel outwash deposits overlying a basal confining bed of relatively impermeable, pebbly clay loam (till) (Reiten and Tischmak, 1993). The outwash deposits are saturated beneath the modern drainages and wetlands and unsaturated below topographic rises, creating unconfined aquifer conditions in the saturated outwash deposits (Reiten and Tischmak, 1993). A 200-minute aquifer test in the outwash deposits at oil field site 124 yielded a hydraulic conductivity of 100 ft/day and a transmissivity of 1400 ft^2/day, from a saturated thickness of 14 feet. The till was not exhumed to test for the presence of fractures, however, its position below saturated outwash, would likely reduce fracture permeability through swelling of saturated clays. Hydraulic conductivity is several orders of magnitude greater in outwash deposits compared to till, which promotes the lateral transport of groundwater and contaminants (Schwartz and Zhang, 2003).

To determine brine plume locations and assist in the placement of groundwater monitoring wells, geophysical surveys were performed in 1988 and 1989 using Geonics EM-31 and Geonics EM-34 equipment to locate areas of high apparent conductivity (Fig. 13). Reiten and Tischmak (1993) produced contoured apparent conductivity maps for four areas determined to be sources of saline contamination within the Goose Lake

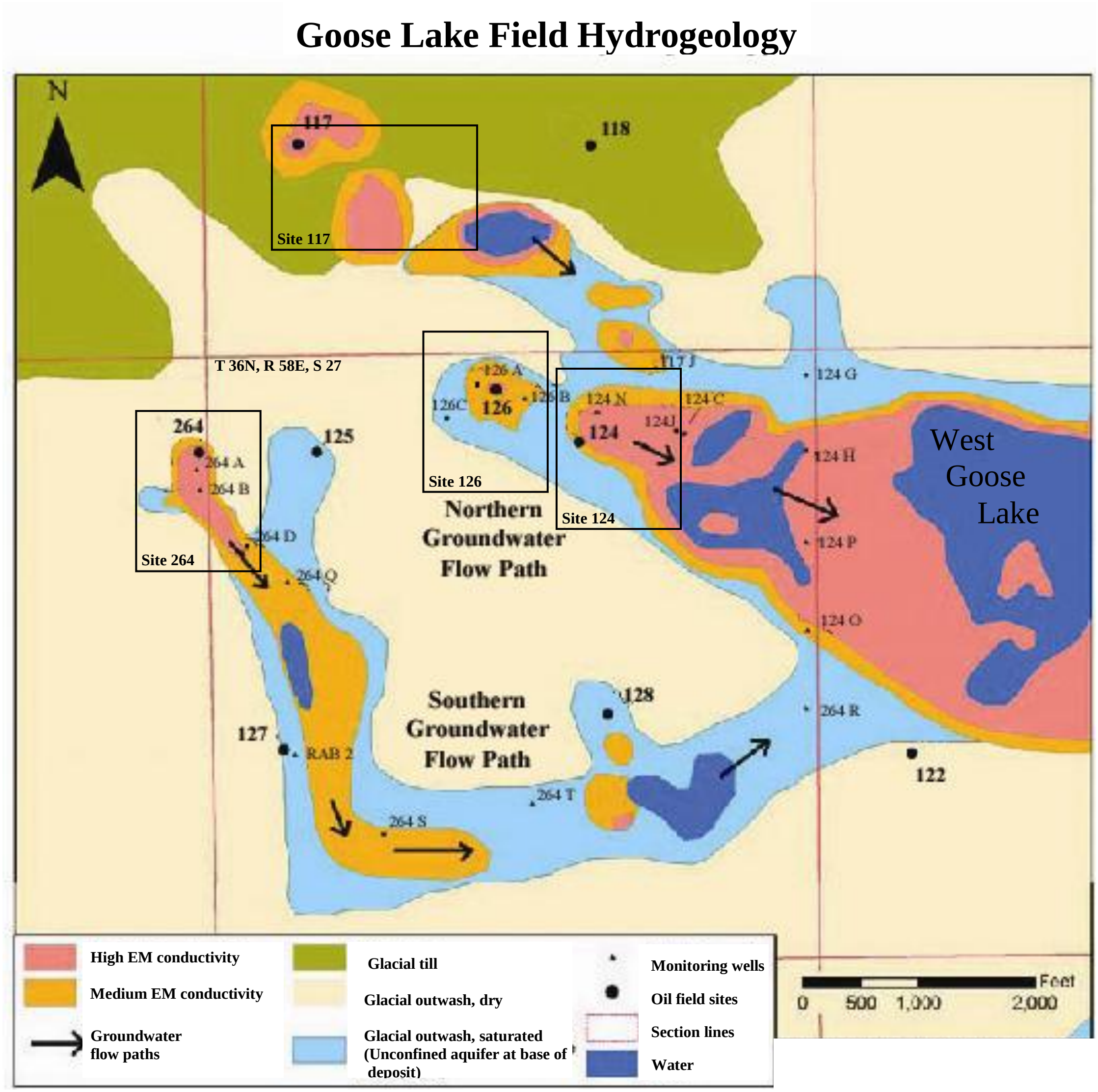

Figure 13. Hydrogeologic and apparent conductivity map of the Goose Lake detailed study area (modified from Reiten and Tischmak, 1993). Black boxes show the approximate locations of EM-31 and EM-34 surveys shown in Figure 14.

melted, the sediments on top collapsed onto the till below the stagnant ice. Thus, the modern drainage systems in the collapsed outwash sediments may not follow the detailed study area: Site # 117, Site # 124, Site # 126, and Site # 264 (Fig. 14). Unfortunately, this data was recorded in now lost field notebooks and attempts to georeference the images in Figure Fourteen proved unsuccessful. Based on the EM surveys, several

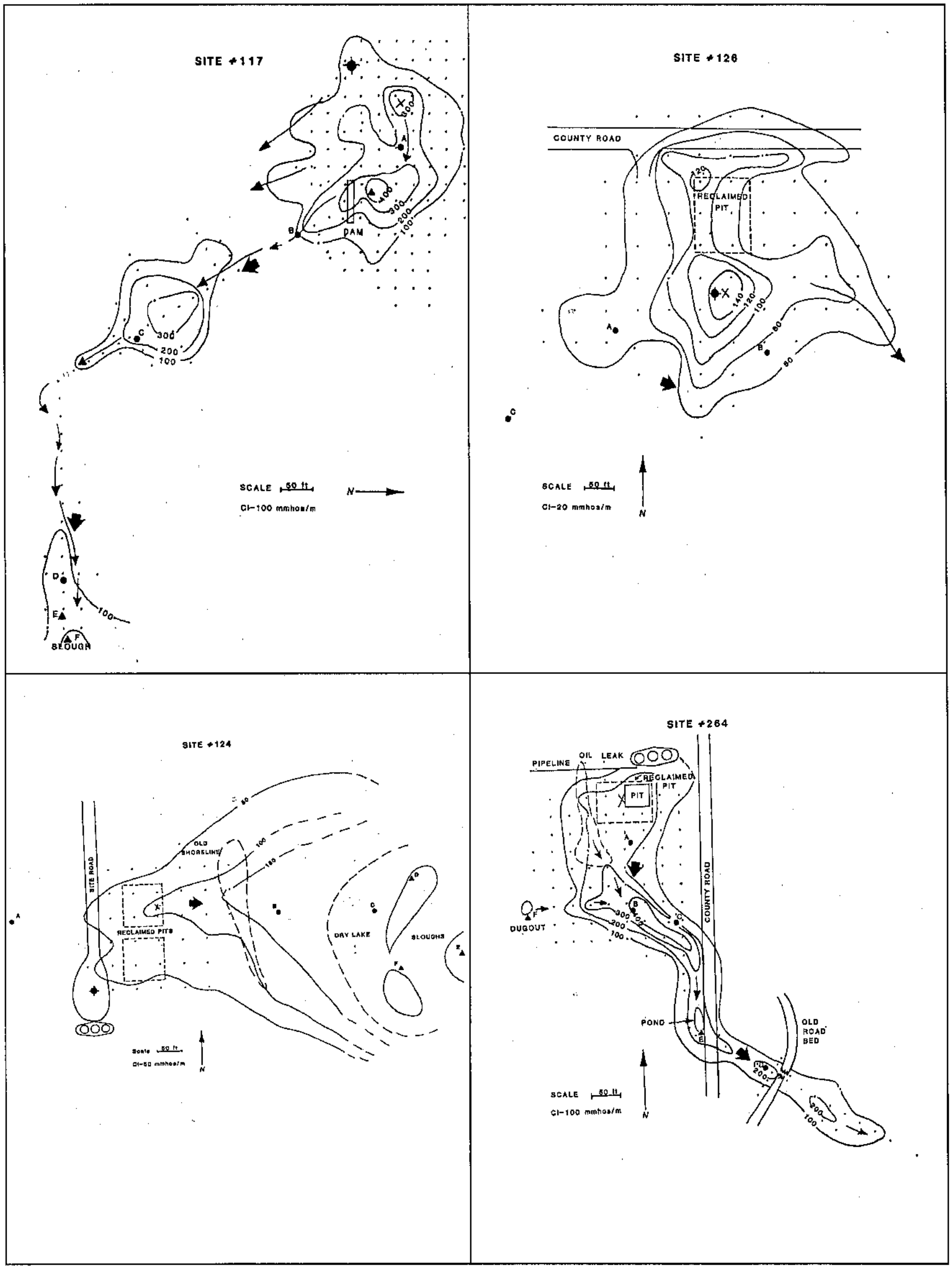

Figure 14. Contoured apparent conductivity maps compiled from 1989 EM-31 and EM-34 surveys at four oil field sites within the Goose Lake detailed study area contaminated by co-produced water. The arrows within each panel denote groundwater flow directions and site numbers correspond to oil field sites in Figure 8 (from Reiten and Tischmak, 1993).

additional groundwater monitoring wells were installed, with samples collected and submitted for analytical chemistry in 1989. Results from the geophysical surveys and groundwater samples identified several areas of chloride contamination within and adjacent to T36N, R58E, S27, with CI values from the vast majority of groundwater samples indicating saline contamination from co-produced waters (> 0.035).

During several site visits in 1989 and 1990, Reiten and Tischmak also recorded field parameters (chloride concentration and specific conductance) at numerous wetlands and West Goose Lake to observe seasonal changes in surface water chemistry that occur due changes in the water level (data in Reiten and Tischmak, 1993). Most of the surface water sites sampled were wetlands that have been delineated by the National Wetland Inventory (NWI) and classified according to the Cowardin classification system (1979). All of the sampled wetlands in the Goose Lake detailed study area in the NWI are classified as palustrine, and are either seasonally or temporally flooded, with the exception of West Goose Lake which is lacustrine and intermittently exposed. As with the groundwater samples, high levels of chloride contamination were observed in the surface water sites with the CI values often increasing as the water levels in the wetlands decreased.

Only samples from two groundwater wells had CI values below 0.035 (0.005 and 0.016). These two samples also had very similar chloride concentrations (7 and 12 mg/L) and were considered chemically representative of uncontaminated groundwater at the Goose Lake detailed study area. In contrast, the chloride concentrations of contaminated samples (CI > 0.035) ranged from 80 – 66,900 mg/L, with twenty samples having concentrations above 1,000 mg/L. The high chloride concentrations (> 1,000 mg/L) seen in the majority of contaminated samples demonstrated the high degree of co-produced water contamination in 1989.

Additionally, Reiten and Tischmak (1993) delineated two groundwater flow paths within the saturated outwash deposits at the Goose Lake detailed study area (Fig. 13). The two flow paths are referred to hereafter as the northern and southern groundwater flow paths. Saline contamination sources for the northern groundwater flow path are from oil field sites 124 and 126 in collapsed outwash deposits. Additional sources are from the reserve pit, a leaky co-produced water pipeline and breaching of an earth-filled dam at oil field site 117 in glacial till to the north. Fluids released from the leaky pipeline and the dam breach appeared to have traveled overland until reaching the more permeable outwash sediments and infiltrated to the water table (Reiten and Tischmak, 1993). All water sample locations in the northern groundwater flow path have prefixes of 117, 124, or 126. Contamination in the southern groundwater flow path came from the Hammer M tank Battery (oil field site 264) and mixed with additional sources from oil field sites 125 and 127. The contamination in the lower portion of the flow path is associated with oil field site 128. Hydrogeologic investigations in 1989 showed evidence of saline contamination from reserve pits and supported land owner reports of frequent oil and co-produced water pipeline leaks, and uncontained discharge of saline co-produced waters into the slough south of the tank battery. Water sample locations in the southern groundwater flow path all have a 264 prefix, with the exception of monitoring well RAB 2. West Goose Lake (307B) is considered the convergence point of the two flow paths.

2005/2006 Hydrogeologic Investigations

The Goose Lake Detailed study area was one location in a hydrogeologic investigation conducted by MBMG and USFWS researchers examining the potential of saline contamination to USFWS managed lands that used similar techniques as those employed by Reiten and Tischmak in 1989 (Nelson pers. comm., 2009). Three of the groundwater monitoring wells installed by the MBMG in 1989, and one other well installed in August of 2005 (RAB 2), were sampled in the fall of 2005 for water chemistry. Water samples from eight additional wells were collected in the spring of 2006. Geophysical surveys using a Geonics EM-31 were also performed on numerous USFWS lands and within the Goose Lake detailed study area in 2004 as part of this study. All EM surveys conducted on USFWS lands in 2004 were spatially interpolated by the author in 2009.

2009 Hydrogeologic Investigations

In 2009, rising head slug tests were performed to determine the hydraulic conductivity of the sediments surrounding six groundwater wells, three in each groundwater flow path (Fig. 13). These wells were then outfitted with capacitance rods to continually monitor groundwater elevations, and resulting hydraulic gradient, from July of 2009 to June of 2010. In the northern groundwater flow path wells tested and outfitted were 126 C, 124 J, and 124 H. Wells 264 B, 264 S, and 264 R were tested and monitored in the southern groundwater flow path. Well numbers for both flow paths are listed from highest to lowest topographic position.

Water sampling and geophysical surveys were both repeated within the Goose Lake detailed study area. Water samples from eighteen groundwater wells, ten wetlands, West Goose Lake, and the co-produced water held in the Hammer M tank battery were collected and submitted for analytical chemical analysis in the spring. The thirty sample locations selected in 2009 included all sampled sites in 2005/2006, and twenty six from 1989; seventeen groundwater wells, eight wetlands, and West Goose Lake. Geophysical surveys were conducted in the fall using a combination of Geonics EM-31 and EM-34 equipment over areas of high conductivity observed in previous EM surveys, and at additional sites selected from the locations of contaminated water samples in May.

Hypotheses

Two hypotheses are examined in this thesis: 1a) Saline groundwater plumes have continued to migrate down-gradient, towards West Goose Lake, increasing the lateral extent of contamination within the study area. 1b)Despite the expected increase in the lateral extent of contamination, the natural attenuation of contaminated surface and groundwater resources have resulted in a decrease in the CI in areas identified as contaminated in 1989. 2a) Contaminated groundwater plumes have migrated laterally, mainly confined within the permeable outwash sediments mantling the underlying less permeable glacial till. 2b)It is expected that the till is acting as a leaky aquitard, allowing little penetration and downward movement of saline contaminated groundwaters relative to lateral migration.

To test the first hypothesis, groundwater elevation data from 2009 – 2010 and water chemistry data from 1989 and 2009 were analyzed to determine groundwater flow directions and the changes in CI values in surface and groundwaters, respectively.

Hydrogeological investigations in 1989 delineated two groundwater flow paths that converge near West Goose Lake based on groundwater elevations in T36N, R58E, S27. To verify this analysis, and to examine potential seasonal variations in groundwater flow directions, water elevations were recorded for eleven months in groundwater wells outfitted with capacitance rods. The distributions of groundwater elevations allowed the hydraulic gradient and horizontal flow directions of the shallow groundwater system within the Goose Lake detailed study area to be monitored over a nearly one year period.

Thirty water samples were collected in 2009, with twenty six obtained from locations sampled in 1989. The 1989 field and laboratory results revealed that twenty four of these twenty six sites had CI values greater than 0.035, indicating contamination from co-produced waters (Reiten and Tischmak, 1993). Only two groundwater wells, 126 C and 264 T, had CI values (0.005 and 0.016, respectively) below 0.035, and neither well was sampled during 2005/2006. In 1989, well 126 C was up-gradient of a saline groundwater plume contaminating wells 126 A and 126 B, while well 264 T was down-gradient from the contamination plume emanating from the Hammer M tank battery (Figs. 10 and 13). Groundwater well 264 S is less than a quarter mile up-gradient from well 264 T, and was the furthest groundwater well showing contamination (CI = 0.32) in the southern groundwater flow path associated with the tank battery in 1989. If contaminant movement continues down the established groundwater flow paths, well 126

C is expected to remain uncontaminated, whereas well 264 T is expected to exhibit chloride contamination.

Additionally, groundwater well 264 R, which is approximately one half mile down-gradient of well 264 T, yielded the lowest CI value (0.09) of any contaminated site in 1989. This well is the last groundwater monitoring well in the southern groundwater flow path. Any increase in the CI value for well 264 R would further confirm that saline groundwater plumes are migrating down-gradient, towards West Goose Lake. However, an area of high apparent conductivity was identified between wells 264 T and 264 R (Fig. 13), while the wetland between the wells yielded a water sample with a CI value of 0.10 in 1989. Oil field site 128 is situated above the coulee separating wells 264 R and 264 T, and is likely the source of contamination seen in the coulee. Therefore, any increase in the CI value for well 264 R would confirm contamination is moving down-gradient, however, the source (i.e. the Hammer M tank battery or oil field site 128) would remain uncertain.

Since additional sources of co-produced waters to the environment (uncontained co-produced water discharges and new reserve pits) are thought to have been eliminated prior to 1989, it is expected that natural attenuation has decreased the magnitude of contamination. Surface and groundwater resources that exhibited CI values above 0.035 in 1989 were compared to CI values from 2009, using a non-parametric Wilcoxon signed rank test. A significant reduction in CI values from 1989 to 2009 would confirm that contamination is being naturally attenuated. Selected groundwater wells were also sampled in 2005/2006. If reductions in the CI value are evident, data from the three sampling periods may allow an estimation of the time required for remediation in similar geologic environments.

To test the second hypothesis results from geophysical surveys conducted in 2009 using a Geonics EM-31 and Geonics EM-34 were analyzed. The lateral extents of areas contaminated from co-produced waters were delineated using a Geonics EM-31. Depth profiles of contaminated groundwater plumes were created from EM-34 surveys, which allow six measurements from three intercoil spacings in both the horizontal and vertical dipole orientations. Changing the intercoil spacing and dipole orientation of the EM-34 yields different exploration depths, and consequently can produce a vertical profile of contaminated groundwater plumes. If saline contamination is moving laterally, within the permeable outwash deposits, this should be evident by a reduction in apparent conductivity values in successively deeper readings. Finally, the results from the two pieces of equipment will be compared to further characterize the saline contaminated groundwater plumes present at the Goose Lake detailed study area.

METHODS

Hydrogeologic Investigations

Water samples were collected in the Goose Lake detailed study area to determine the extent of saline contamination from oil and gas development. Surface and groundwater samples were obtained by the MBMG in 1989, with the MBMG data presented in this thesis only including the twenty six sites resampled in 2009. Twelve groundwater wells were sampled during joint investigations by the MBMG and USFWS in the fall of 2005 and the spring of 2006. In May 2009, water samples were collected from eighteen groundwater wells, ten wetlands, West Goose Lake, and the co-produced water held in the Hammer M tank battery (Fig. 15). Samples collected in 2009 attempted

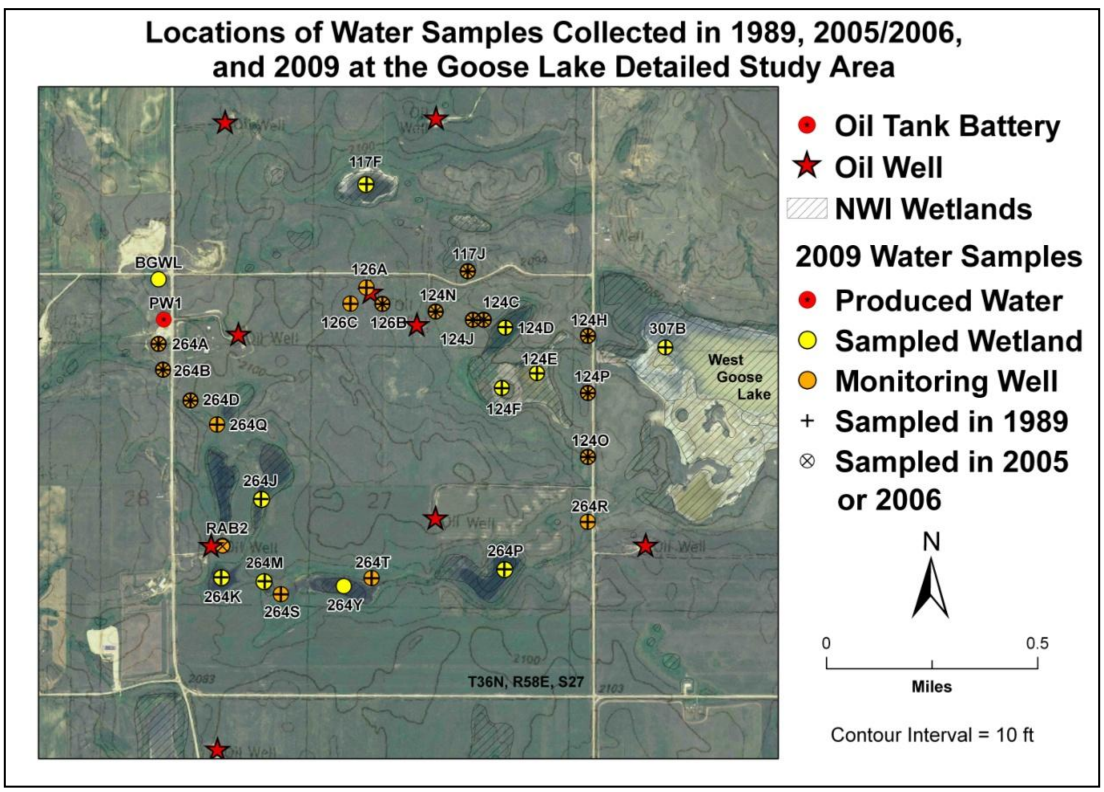

Figure 15. Map showing the locations of surface and groundwater samples collected at the Goose Lake detailed study area in 1989, 2005/2006, and 2009.

to cover the majority of the Goose Lake detailed study area, with the total number of samples collected dictated by analytical costs.

Information on all sampled groundwater wells is available through the MBMG Groundwater Information Center (GWIC) at http://mbmggwic.mtech.edu/, however, complete reports are not available for all wells. Well log information from the GWIC reports are summarized in Table 2. Groundwater monitoring wells installed by MBMG were completed in the fall of 1988 or the summer of 1989. During the MBMG/USFWS joint study, one additional groundwater well (RAB 2) was installed in the Goose Lake detailed study area in the summer of 2005. Groundwater monitoring wells are

Table 2. Hydrogeologic properties of sampled groundwater monitoring wells within the Goose Lake detailed study area. All measurements are in feet (Data from GWIC).

Well Name	Auger Depth	Bottom of Screen	Screen Length	Depth Interval and Geologic Material	
117J	18	15	10	5-17: Outwash	17-18: Unoxidized Till
124C	8	8	5	3-8: Outwash	
124H	17	17	5	12-16: Outwash	16-17: Unoxidized Till
124J	18	18	10	5-15: Outwash	15-18: Unoxidized Till
124N	20.5	18	10	8-18: Outwash	18-20.5: Unoxidized Till
124O	23	19	5	14-19: Outwash	19-23: Lake Clays
124P	17	12	5	7-12: Outwash	12-15: Oxidized Till
126A	17	17	5	12-17: Outwash	
126B	23	23	5	18-20 Lake Clay	20-23: Unoxidized Till
126C	19	16	5	11-15: Outwash	15-19: Lake Clays
264A	23	23	5	18-21: Outwash	21-23: Lake Clays
264B	8	8	5	3-6: Outwash	6-8: Transitional Till
264D	13	13	5	8-10: Outwash	10-13: Unoxidized Till
264Q	23	20	5	15-20: Outwash	20-23: Transitional Till
264R	22	20	5	15-19: Outwash	19-22: Unoxidized Till
264S	18	13	5	8-13: Outwash	13-18: Lake Clays
264T	27	20	5	15-23: Outwash	18-19/23-27: Lake Clay
RAB-2	13	12	5	7-12: Outwash	

constructed of 2 inch inside diameter, schedule 40 PVC pipe and utilize number-10-slot screens, a gravel pack, and bentonite seal.

1989 and 2005/2006 Surface and
Groundwater Chemistry

In1989 the MBMG sampled five of the seventeen reported groundwater wells in the spring (4/20 – 5/2/1989) with the remaining twelve wells sampled in the fall (10/12 – 10/17/1989). Samples collected by the MBMG and USFWS were obtained on September 15th, 2005 and the 4th, 5th and 10th – 12th of April, 2006. Different pumps and bailers were used during all hydrological investigations due to the low yield of some wells; however, all equipment used to collect groundwater samples were thoroughly cleaned using tap water, followed by Liqui-Nox cleaning solution, and finally deionized water rinses between sampling. In all studies, groundwater monitoring wells were purged of at least three casings full of water prior to sampling. After collection, bulk water samples were split into three bottles: a filtered and acidified sample, a filtered and non-acidified sample, and a non-filtered and non-acidified sample. A 0.45 micron filter was used to filter samples, with acidification done using 4 ml of HNO_3. Groundwater samples collected in 1989, 2005, and 2006 were analyzed for major ion chemistry, metals, and several other potential contaminants associated with drilling fluids used in the Williston Basin at MBMG's Analytical Laboratory in Butte, Montana. Procedures for all analyses conducted by the MBMG are described in EPA methods 200.7, 200.8, and 300.A (EPA, 1994).

The MBMG recorded field parameters, including chloride concentration and specific conductance, at numerous surface water sites within the Goose Lake detailed study site throughout 1989-1990. The field parameters reported, which were used in lieu of 1989 analytical chemistry results for all wetlands and West Goose Lake as no surface water samples were analyzed in the laboratory, were obtained in the spring of 1989 (April 21st and March 1st and 2nd). Field parameters were determined from water samples, collected in sterilized plastic cups, obtained directly from the surface water body (dipping of cup from shore/shallow water). Chloride concentrations were determined using QUANTAB Chloride Titrators numbers 1175 (range 60 – 480 ppm Cl^-) or 1176 (range 300 – 4,900 ppm Cl^-). Chloride concentrations in the range of 60 – 4,900 mg/L were read directly from the titrator strip, while samples with higher concentrations required dilution with deionized water using a volumetric flask. Specific conductance was determined with a YSI Model 30 probe.

2009 Hydrogeologic Investigations

Hydraulic Conductivity The hydraulic conductivities of the near surface sediments adjacent to six groundwater wells (124 H, 124 J, 126 C, 264 B, 264 R, and 264 S) were calculated from rising head slug tests performed on the 16th of July, 2009, using the methods of Bouwer and Rice (1976) and Bouwer (1989). Data was recorded on TruTrack WT-HR 1000 or 1500 capacitance rods (model number refers to effective measuring distance in mm) suspended by chain from the well cap. TruTrack capacitance rods contain a capacitive sensor capable of recording water heights, in mm, and an internal data logger to record data at user specified time intervals. All rising head slug tests utilized a 30 second recording interval. A Guzzler hand pump was used to bail groundwater wells, with the diameter of the pump hose requiring the rods to be removed prior to bailing. Therefore, a chain was used to hang the rods from a notch carved in the top of the well case for a minimum of five minutes prior to pumping to determine the initial groundwater elevation. Once removed, rods were placed in the shade during well bailing to minimize increases in temperature and reinserted immediately after the pump hose was removed. The same chain link was again placed in the notch on the well case and the wells left open during the test to ensure no changes in the water height occurred from moving the rod prior to the final measurement. Groundwater levels in all wells were allowed enough time for a complete recovery before the rods were removed to download the data.

Hydraulic Gradient The six groundwater wells listed above were then outfitted with a TruTrack capacitance rod to monitor groundwater levels, recorded at 30 minute intervals, from July 18th through September 17th 2009, when two capacitance rods failed (264 B and 124 H). Data collection continued unabated at the remaining four wells until removal of the rods on the 8th of June, 2010. To maximize the time of data collection during the winter months, when site access was unavailable, battery voltage and water and air temperatures were not collected in order to save memory space in the data logger, precluding the ability to temperature correct the data. TruTrack capacitance rods only measure the position of the water level on the capacitance rod. In order to convert this data to accurate groundwater elevations, the USGS surveyed the top of all wells and the land surface at their bases with a high precision, real time kinematic (RTK) GPS using two Trimble 5700 transceivers and an R8 receiver in September of 2009. Trimble 5700 transceivers have a reported measurement accuracy of ± ten mm for the horizontal position and ± 20 mm for the vertical position.

Surface and Groundwater Chemistry All surface and groundwater samples were collected over three days (5/14 – 5/16) in 2009. Surface water samples were collected in 1 gallon or 750 ml sterilized bottles directly from wetlands (walking into the water body and immersing bottle until full). Groundwater sample collection and bulk water sample preparation were done using the methods described previously. An additional, non-filtered and non-acidified, sample was also prepared at a selected subset of sample locations for strontium isotope analysis. Replicate samples were taken at well 264 A and two wetlands, BGWL and 117. Samples were analyzed for major ion chemistry and halides at the United States Geological Survey (USGS) National Water Quality Laboratory (NWQL) in Denver, Colorado. Strontium isotopes analysis was conducted at the USGS Yucca Mountain Project Branch Laboratory (YMPBL), also in Denver. Prepared bottles were stored in a cooler without ice during the remaining sampling and transport by car back to Helena, Montana, before being shipped overnight by FedEx to the respective laboratories within ten days of collection. Analytical chemical analysis at NWQL was performed following schedule 12, lab codes 1202 and 1258, using the methods described by Fishman (1993). Strontium isotope analysis performed at the YMPBL used the methods summarized by Peterman et al. (1985).

Hydrogeologic Data Analysis

The TruTrack capacitance rods used in this study are very precise but their accuracy drifts over time; therefore water height data required correction. Taking a manual measurement at the start and end of data collection determined the initial and final error in the rod (the true water height on the rod, calculated from the manual measurement, minus the water height recorded by the rod). A linear trend was created between these two points in Excel, spanning all data points, and the trend value is added to the corresponding measured rod value. This corrects the initial and final capacitance rod water heights to their respective manually measured water heights, and applies the change in the error over the entire measurement period, linearly adjusting the data in between.

Linearly adjusted capacitance rod data was used to create hydrographs for the six groundwater wells and a contoured potentiometric surface from groundwater elevations recorded at noon on the 25th of July, 2009. Hydrographs were produced in Excel and the potentiometric surface map was created using the Inverse Distance Weighted (IDW) weighted function in ArcGIS 9.3.1. The IDW function uses a linear weighted combination of sample points (the neighborhood) to interpolate the values of every point within the interpolated surface based on the inverse of the distance raised to a user selected power. This analysis utilized a neighborhood of six points (all wells) to ensure continuity across the entire surface, and used the ArcGIS 9.3.1 default power of 2. The horizontal flow directions in the shallow groundwater system from this potentiometric surface were compared to the groundwater flow directions reported by Reiten and Tischmak (1993) to determine potential differences in flow directions.

To monitor temporal changes in surface and groundwater chemistries contaminated by co-produced waters, the CI value for all water samples was determined by dividing the laboratory or field determined chloride concentrations in mg/L by the laboratory or field determined specific conductance in µS/cm. For sites with replicate samples in 2009 (BGWL, 117 F and 264 A), the chloride concentration and specific conductance values from the first water sample were used. The difference in the CI value between 1989 and 2009 was calculated for the twenty four sites that exhibited chloride contamination in 1989. Statistical analysis of the difference in CI values, and validation of the necessary sample distribution assumptions, was performed using the open source software package R 2.9.2, using a paired sample t-test.

Geophysical Surveys

2004 Geophysical Surveys

A Geonics EM-31 terrain conductivity system, owned by USGS Crustal Imaging and Characterization Team, was used to conduct geophysical surveys to determine the lateral extent of saline contamination plumes on the 23rd and 26th of July 2004. The EM-31 has a fixed coil array spaced 3.66m apart, housed in a 4 meter boom and operates at 9.8 KHz. This device continuously monitors apparent conductivity as the boom is carried overland, with readings made at user selected locations. When carried by the shoulder strap, the dipole is orientated vertically and provides an exploration depth (ED) of roughly 4-6 meters. Apparent conductivity measurements were recorded in a Trimble GeoXT GPS using ArcPad software. Survey lines followed no predetermined arrangement, but station (measurement) locations generally were within 30 m.

2009 Geophysical Surveys

Similarly, EM-31 surveys were performed using the aforementioned set-up and equipment source on the 15th and 16th of September and the 8th and 9th of October 2009. The spacings between measurements were variable in coulees and draws but never exceeded 30 m. Otherwise, measurements were taken in a grid pattern at approximately 30 m spacings that were determined from pacing and relative position to the previous survey line as seen on the GPS display.

Additionally, a Geonics EM-34 terrain conductivity system, owned by the USGS Montana Water Science Center, was used to determine the depth profile of contamination plumes on the 15th and 16th of September 2009. The EM-34 is comprised of a receiver and transmitter coil connected by a 40 meter reference cable and allows up to 6 measurements to be taken at each recording site by utilizing three different intercoil spacings in two dipole orientations. The frequencies used for the three spacings of 10, 20 and 40 meters are 6.4, 1.6, and 0.4 KHz respectively. Apparent conductivity readings were taken with the dipole in both the vertical and horizontal orientation, using circular bubble levels affixed on the coils for proper alignment. Depending on the dipole orientation and coil spacing, the EM-34 can provide variable exploration depths of approximately 7-60 meters (Table 3). EM-34 measurements were collected at 60 m spacings, measured with a “wheel tape” in the field, along selected transects in both groundwater flow paths and an area of overlap with the EM-31 to allow direct comparison between the two types of equipment.

Table 3. Geonics EM-34 exploration depths for different intercoil spacings and dipole orientations (from Geonics TN-6).

Intercoil Spacing (meters)	Exploration Depth (meters)	
	Horizontal Dipoles	Vertical Dipoles
10	7.5	15
20	15	30
40	30	60

Geophysical Data Analysis

After completing the EM surveys, the apparent conductivity data needed to be spatially interpolated for visual display. The Arcpad software used to record the apparent conductivity data created shapefiles from the latitude and longitude coordinates, creating the foundation for the spatially interpolated surfaces. However, the construction of these maps involved several steps to accurately display the apparent conductivities recorded, all of which were performed with ArcGIS 9.3.1. First, a buffer was created around each set of survey points at a distance of 25 meters for EM-31 data and 75 meters for EM-34 data. The IDW function was used to interpolate the electrical conductivity data, in raster format, to the same rectangular extent as the buffer layer using a neighborhood of three points and default power of 2. A series of data transformations was then required to create a shapefile from the IDW raster file. Finally, the shapefile was clipped to the extent of the buffer layer, providing data only for areas that had good geophysical control.

The EM-31, operated in the vertical dipole mode, has a similar exploration depth to the EM-34, when spaced at 10 meters, with the dipole horizontal. In order to compare data collected from the two types of equipment, the spatially interpolated conductivity values were normalized relative to background conductivity. The background conductivity for each piece of equipment was determined from the average of multiple readings taken in uncontaminated areas. This resulted in the apparent conductivity data being expressed as a response ratio, relative to the background conductivity measured by the respective instrument at the Goose Lake detailed study area.

RESULTS

Hydrogeologic Investigations

Hydraulic Conductivity

The hydraulic conductivity (K) values of the near surface sediments, determined from rising head slug tests, exhibited variation across the Goose Lake detailed study area (Table 4, Appendix A). K values ranged from 0.18 ft/d to 13.50 ft/day, with a geometric mean of 2.06 ft/d. The majority of wells tested have part of the screened interval penetrating into the underlying till and the calculation of K values for those wells only used the portion of the screen in the outwash deposits. Groundwater well 124 J has a different screen length (10 ft); however, the K value is within the range of the other tests. Additionally, there appears to be no obvious relationship between K and the ratio of outwash to till or lake clay in the screened interval of sampled wells.

Table 4. Hydraulic conductivity (K) values from rising head slug test (RHST) for sediments adjacent to six groundwater wells within the Goose Lake detailed study area from July 17, 2009.

Well Name	GWIC ID	Screen Length (ft)	Percent of Screen in Given Geologic Material	Hydraulic Conductivity (K)		
				ft/d	cm/s	m/d
124 H	890935	5	80/20 - Outwash/Till	13.50	4.74E-03	4.10
124 J	890928	10	70/30 - Outwash/Till	5.01	1.77E-03	1.53
126 C	890942	5	80/20 - Outwash/Lake Clay	0.18	6.40E-05	5.53E-02
264 B	890445	5	60/40 - Outwash/Till	4.50	1.59E-03	1.37E-03
264 R	890936	5	80/20 - Outwash/Till	1.29	4.54E-04	0.39
264 S	890938	5	100 - Outwash	1.07	3.77E-04	3.26
Geometric mean				2.06	7.26E-04	0.29
Minimum				0.18	6.40E-05	1.37E-03
Maximum				13.50	4.74E-03	4.10

Hydraulic Gradients

Hydrographs from the six groundwater wells outfitted with capacitance rods are stable, with changes in groundwater elevations generally less than two feet (Fig. 16-A). Since the groundwater elevations in the four wells that have complete records did not change relative to each other throughout the entire monitoring period, no seasonal reversals in the horizontal flow directions in the shallow groundwater system are indicated. Despite two of the rods yielding incomplete records, they would have required larger fluctuations in water levels than those seen in wells with complete records to change the observed hydraulic gradient. Depth to the water table varied between wells, and individual wells showed both seasonal and event induced changes in groundwater elevations below land surface (16-B). In the northern groundwater flow path, the depth to the water table decreases along the flow path (Table 5). The significance of the different water table configuration in the southern and northern groundwater flow paths will be addressed in the Discussion.

Although only six wells and an automatic contouring program (IDW) were used to examine hydraulic gradient, the results show broad agreement with those present by Reiten and Tischmak (1993). Due to the consistent nature and relative position of groundwater elevations seen in Figure 16-A, groundwater flow directions appear annually constant. Groundwater flow directions are perpendicular to the equipotential lines on the potentiometric surface map created from groundwater elevations on the 25th of July, 2009 (Fig. 17). The potentiometric surface map clearly shows groundwater flow directed along both the northern and southern groundwater flow paths. The bulls-eye pattern surrounding the highest and lowest wells, 264 B and 124 H, is an artifact of the

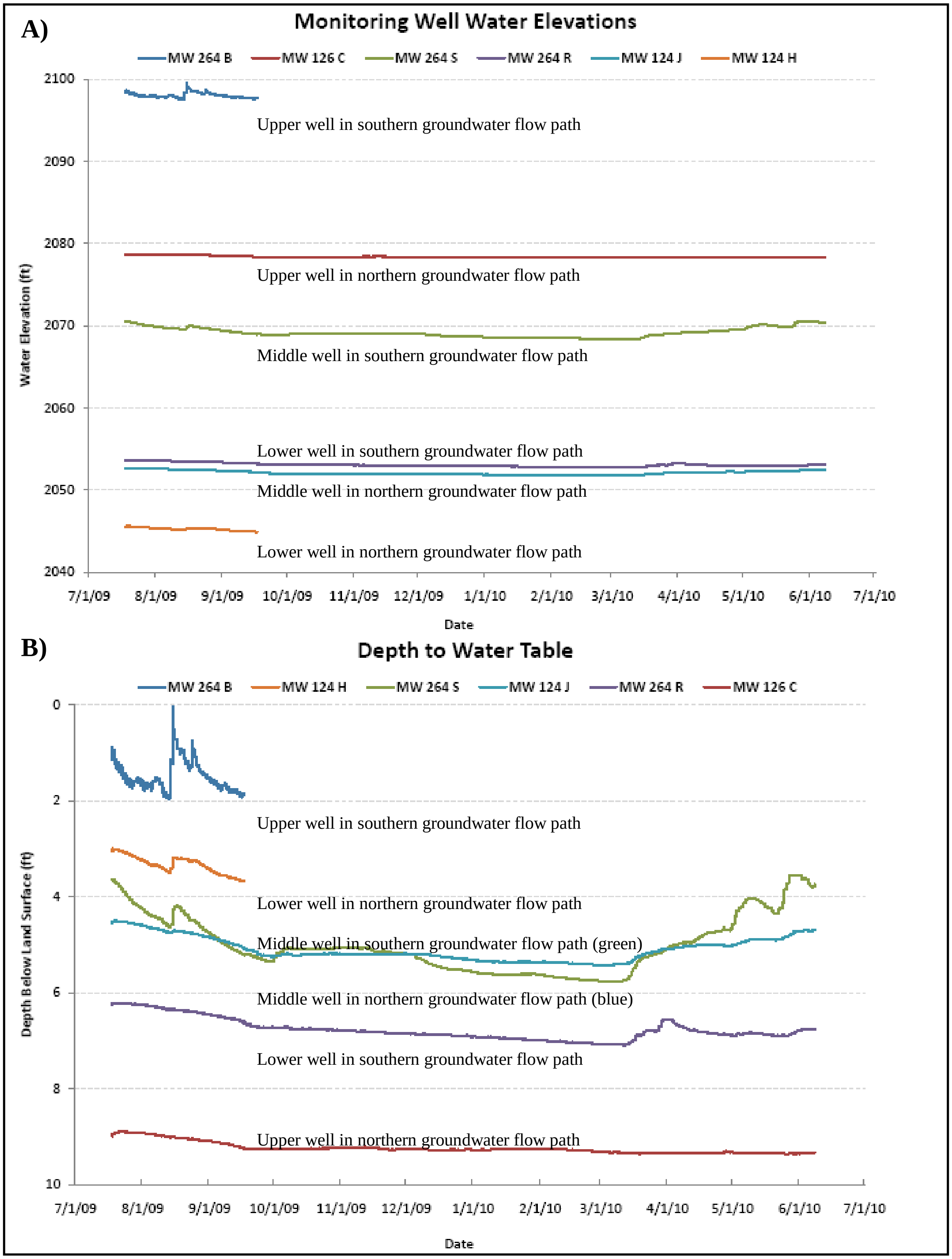

Figure 16. Panel A shows water level elevations in groundwater monitoring wells (MW) from 7/18/2009 – 6/8/2010. Panel B shows the depth to the water table for the same time period. Shorter recording periods for wells 124 H and 264 B is due to instrument failure after 9/17/2009.

Table 5. Land surface elevations and average depths to the water table for the six wells outfitted with capacitance rods in the two groundwater flow paths. Wells are listed from highest to lowest topographic position in each groundwater flow path. Italics indicate average value calculated from incomplete records.

Northern Groundwater Flow Path			Southern Groundwater Flow Path		
Well Name	**Land Surface Elevation (ft)**	**Average Depth to Water Table (ft)**	**Well Name**	**Land Surface Elevation (ft)**	**Average Depth to Water Table (ft)**
126C	2087.61	9.23	264B	2099.52	*1.48*
124J	2057.18	5.07	264S	2074.15	4.96
124H	2048.58	*3.33*	264R	2059.85	6.74

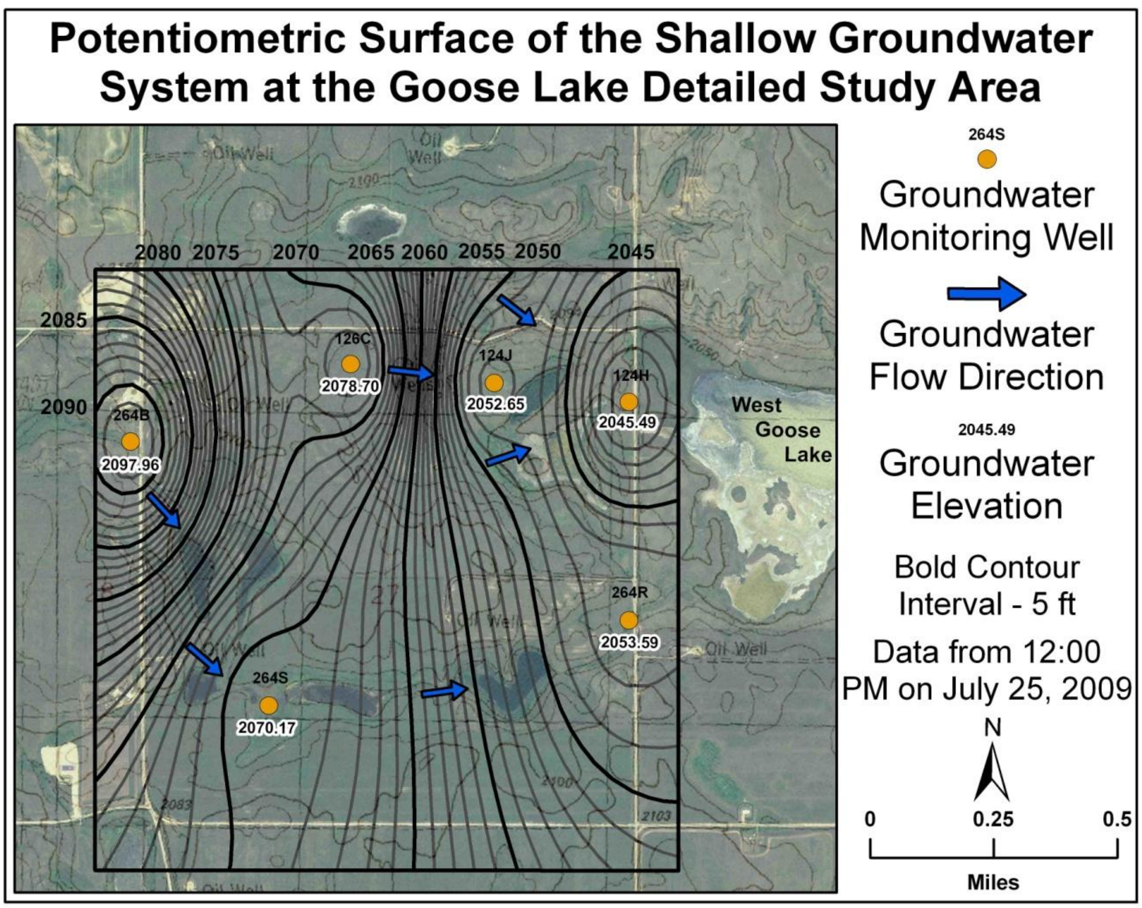

Figure 17. Potentiometric surface map of the Goose Lake detailed study site. Elevations of equipotential lines, in feet above sea mean level, are listed outside the black box.

interpolation method and not representative of the local hydraulic gradient as the contours to not reflect local topography as would be expected in the unconfined sand and gravel aquifer. The steep gradients south of well 264 B and east of well 126 C reflect the hummocky landscape which promotes topographically driven groundwater flow.

Surface and Groundwater Chemistry

The surface and groundwater chemistry results from 1989, 2005/2006, and 2009 (Appendix B) document widespread contamination to aquatic resources from oil and gas development within the Goose Lake detailed study area. Analysis of all water samples were performed in an analytical chemical laboratory with the exception of surface water sites sampled in 1989, where chloride concentrations and specific conductance values were calculated in the field. Field and laboratory determined chloride concentrations in 1989 compared well, with the log-transformed data from roughly thirty samples having an R^2 value of 0.994 (Reiten and Tischmak, 1993). The accuracy of laboratory determined chloride concentrations and specific conductance values from MBMG's Analytical Laboratory are reported to be within 10 %, but could be as high as 20 % due to dilution (Steve McGrath, pers. comm., May 2009). Similarly, measurement accuracies were likely within 5% for 2009 water samples, as seen in data from the USGS Branch of Quality Systems inorganic blind sample program, which continuously runs double blind, environmental matrix based (i.e. from natural water samples as opposed to a DI water based synthetic matrix samples) samples with known concentrations on most of the analytical equipment at NWQL. During June of 2009, when water samples from the Goose Lake detailed study area were analyzed, the maximum standard errors for twelve chloride and fourteen specific conductance blind samples were 4.9 and 4.0 %, respectively (data available from http://bqs.usgs.gov).

Additionally, well 264 B had to be repaired prior to sampling in 2009, and based on the water sample chemistry, was clearly in hydrologic connection with surface waters. Therefore, results from well 264 B were compared to field parameters from wetland 264 E (the slough adjacent to well 264 B) obtained on 4/20/89, and the site will be referred to hereafter as wetland 264B/E. Results from 1989, 2009, and the temporal change in chloride concentrations and specific conductance are displayed in Figures 18 and 19, respectively. From these data, maps of the 1989, 2009, and temporal change in CI values were produced (Fig. 20). All values are summarized in Table 6.

Decreases in chloride concentrations between 1989 and 2009 were observed in four wetlands, West Goose Lake, and twelve groundwater wells. Increased chloride concentrations were observed in four wetlands (117F, 264J, 264M, and 264P) and five groundwater wells (124 O, 126 A, 126 C, 264 R, and 264 T). These four wetlands and five groundwater wells, along with West Goose Lake and well 264 A, had increased specific conductance values in 2009. The remaining seventeen surface and groundwater sites had decreased specific conductance values. Wetlands and groundwater wells with greater chloride concentrations in 2009 compared to 1989 were also accompanied by increased CI values, with the exception of wetland 264 P. Decreased CI values were observed in West Goose Lake and the remaining five wetlands and twelve groundwater wells (Table 7).

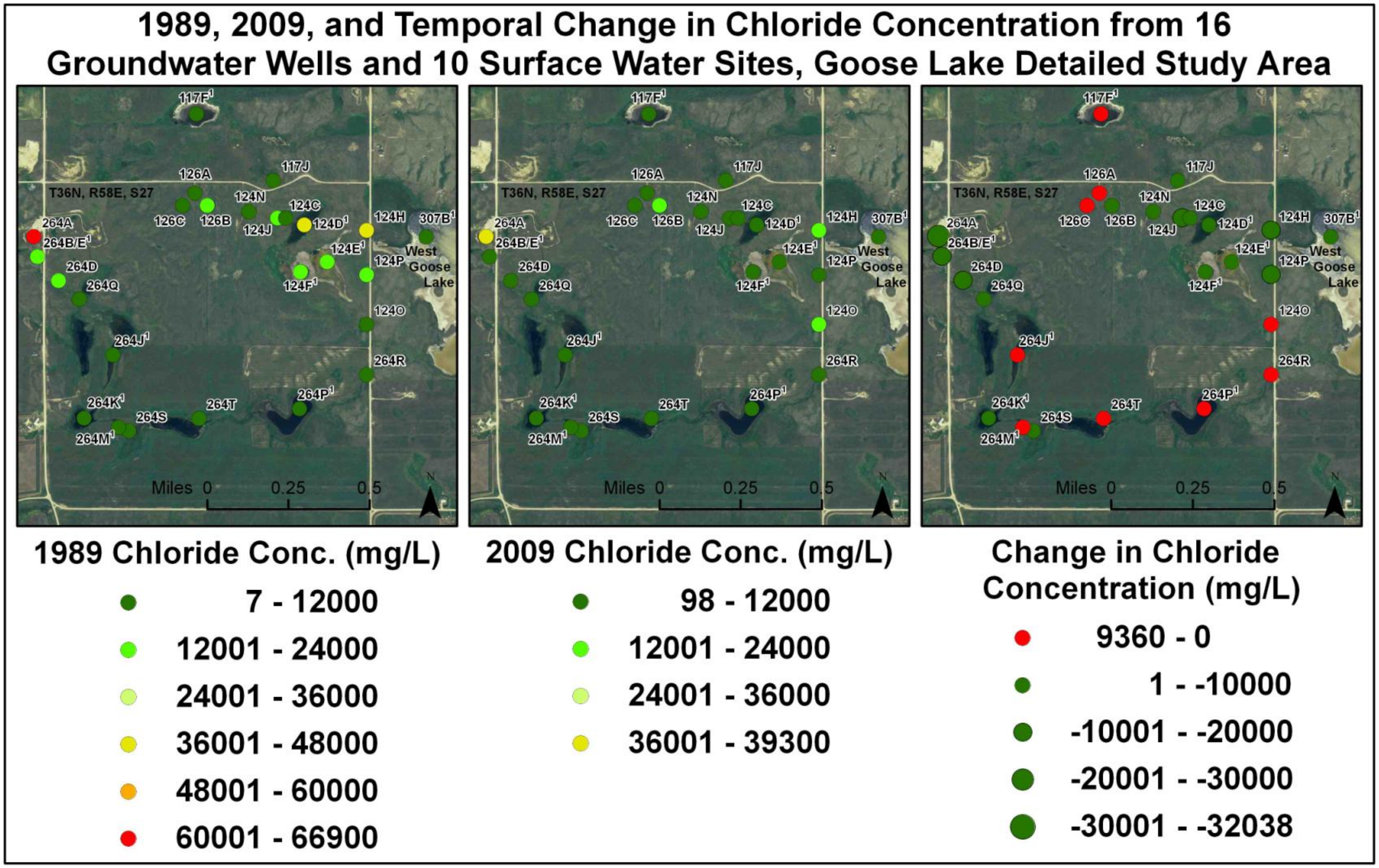

Figure 18. Maps showing the specific conductance (C.C.) of water samples from 16 groundwater wells and 10 surface water sites within the Goose Lake detailed study area. First panel shows 1989 results. Second panel shows 2009 results. Third panel shows the temporal change, calculated as the 2009 C.C. minus the 1989 C.C., with negative values indicating a lower C.C. in 2009. Note: the max and min values in each legend are respective values form that panel.

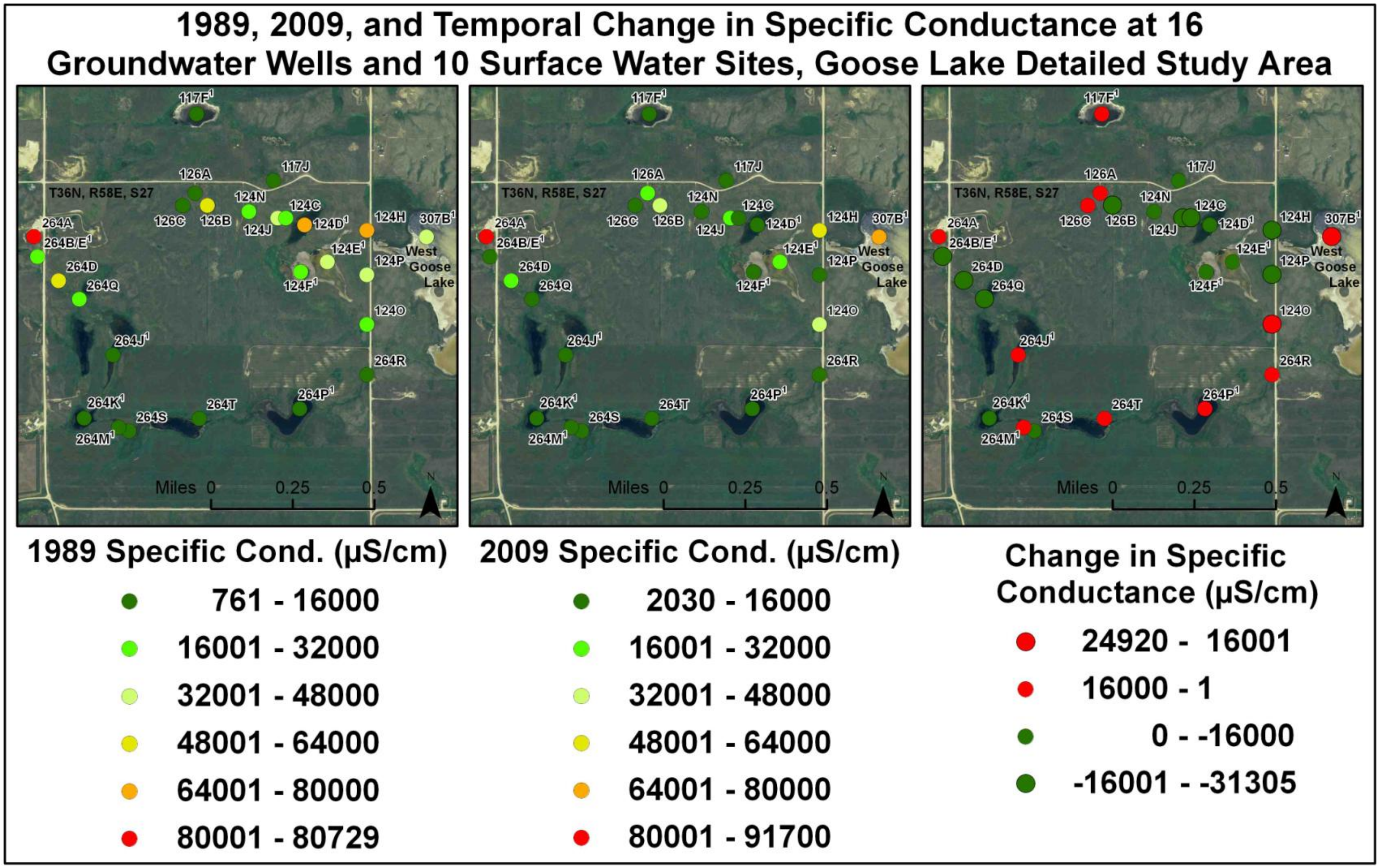

Figure 19. Maps showing the specific conductance (S.C.) of water samples from 16 groundwater wells and 10 surface water sites within the Goose Lake detailed study area. First panel shows 1989 results. Second panel shows 2009 results. Third panel shows the temporal change, calculated as the 2009 S.C. minus the 1989 S.C., with negative values indicating a lower S.C. in 2009. Note: the max and min values in each legend are respective values form that panel. A superscript 1 denotes surface water samples, i.e. 117^1.

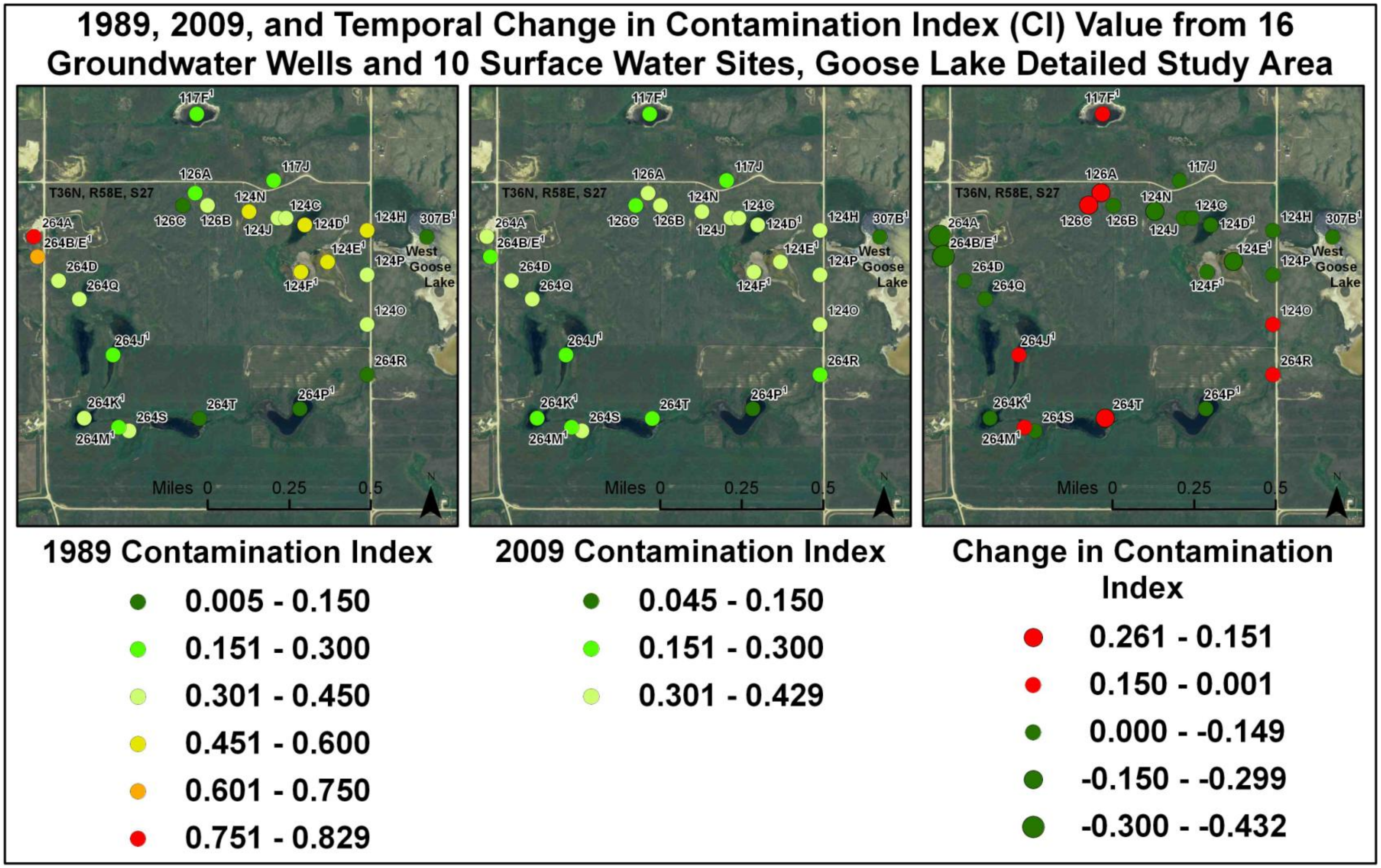

Figure 20. **Maps showing the specific conductance (C.I.) of water samples from 16 groundwater wells and 10 surface water sites within the Goose Lake detailed study area. First panel shows 1989 results. Second panel shows 2009 results. Third panel shows the temporal change, calculated as the 2009 C.I. minus the 1989 C.I., with negative values indicating a lower C.I. in 2009. Note: the max and min values in each legend are respective values form that panel. A superscript 1 denotes surface water samples, i.e. 117^1.**

Table 6. The chloride concentrations, specific conductance, and contamination index values from sampled surface water sites and groundwater wells in 1989 and 2009. Surface water sites are denoted by a superscript 1, i.e. 117F[1]. Contamination index values in bold are considered uncontaminated. Locations of all samples are shown in Figure 15.

Site Name	GWIC ID	Chloride Concentration (mg/L)		Specific Conductance (μS/cm)		Contamination Index	
		1989	2009	1989	2009	1989	2009
117F[1]		450	2,941	2,080	10,750	0.216	0.275
117J	890941	2,560	1,550	9,211	5,980	0.278	0.259
124C	890429	10,600	2,330	29,096	7,540	0.364	0.309
124D[1]	220933	5060	3,280	11,090	10,600	0.456	0.309
124E[1]	220934	16,870	7,020	34,700	22,000	0.486	0.319
124F[1]		12,790	4,870	27,030	14,700	0.473	0.331
124H	890935	36,500	19,500	78,617	50,000	0.464	0.390
124J	890928	18,000	6,150	44,480	17,900	0.405	0.344
124N	890929	12,000	3,340	22,557	10,300	0.532	0.324
124O	890932	7,840	17,200	21,375	44,500	0.367	0.387
124P	890933	16,300	4,370	40,603	13,600	0.401	0.321
126A	890432	1,060	6,100	6,233	17,900	0.170	0.341
126B	890433	23,600	15,800	59,538	41,300	0.396	0.383
126C	890942	7	964	1,203	3,630	0.005	0.266
264A	890940	66,900	39,300	80,729	91,700	0.829	0.429
264B	890445	32,800		78,626		0.417	
264B/E[1]			762		3,400	0.656	0.224
264D	890446	18,600	5,330	48,105	16,800	0.387	0.317
264E[1]		15,000		22,880		0.656	
264J[1]	220987	350	661	1,740	2,680	0.201	0.247
264K[1]		2,190	396	6,800	2,030	0.322	0.195
264M[1]		140	1,270	890	4,500	0.157	0.282
264P[1]	220993	80	98	840	2,160	0.095	0.045
264Q	890939	10,300	3,940	28,810	12,500	0.358	0.315
264R	890936	195	732	2,182	3,460	0.089	0.212
264S	890938	2,810	1,910	8,800	6,290	0.319	0.304
264T	890937	12	1,030	761	4,650	0.016	0.222
264Y[1]			120		1,080		0.111
307B[1]	221011	4,480	4,050	41,680	66,600	0.107	0.061
BGWL[1]			9		394		0.022
RAB2	221723		975		3,970		0.246

Table 7. Change in specific conductance (S.C.), chloride concentration (C.C.) and contamination index (C.I.) values for repeat water samples between 1989 and 2009, with negative values indicating a lower value in 2009. Surface water sites are denoted by a superscript 1, i.e. 117F[1].

Site Name	Change in:			Site Name	Change in:		
	S.C.	C.C.	C.I.		S.C.	C.C.	C.I.
117F[1]	8,670	2,491	0.059	**126C**	2,427	957	0.261
117J	-3,231	-1,010	-0.019	**264A**	10,971	-27,600	-0.400
124C	-21,556	-8,270	-0.055	**264B/E[1]**	-19,480	-14,238	-0.432
124D[1]	-490	-1,780	-0.147	**264D**	-31,305	-13,270	-0.070
124E[1]	-12,700	-9,850	-0.167	**264J[1]**	940	311	0.046
124F[1]	-12,330	-7,920	-0.142	**264K[1]**	-4,770	-1,794	-0.127
124H	-28,617	-17,000	-0.074	**264M[1]**	3,610	1,130	0.125
124J	-26,580	-11,850	-0.061	**264P[1]**	1,320	18	-0.050
124N	-12,257	-8,660	-0.208	**264Q**	-16,310	-6,360	-0.043
124O	23,125	9,360	0.020	**264R**	1,279	537	0.123
124P	-27,003	-11,930	-0.080	**264S**	-2,510	-900	-0.015
126A	11,667	5,040	0.171	**264T**	3,890	1,018	0.206
126B	-18,238	-7,800	-0.013	**307B[1]**	24,920	-430	-0.046

The two largest decreases in CI values observed in the detailed study area occurred at wetland 264 B/E and well 264 A (-0.431 and -0.400, respectively), which are immediately below the Hammer M tank battery and had the greatest CI values in 1989. The two sites that were considered uncontaminated in 1989, 126 C and 264 T, had the largest increases in CI values, with both sites exhibiting saline contamination from co-produced waters in 2009. Despite the decreases in CI values observed in eighteen of the twenty four sites identified as contaminated in 1989, all twenty six sites were above the 0.035 threshold established by Reiten and Tischmak (1993) in 2009. Additionally, wetland 264 Y in the southern groundwater flow path (Fig.15), which was not sampled by the MBMG in 1989, had a CI value indicating chloride contamination in 2009 (0.111, Appendix B). Only a water sample and replicate from BGWL (Background Wetland,

Fig. 15), taken from a small wetland up-gradient from the Hammer M tank battery, had CI values below 0.035 in 2009. With average chloride concentrations and specific conductance values of 9 mg/L and 390.5 µS/cm, the two samples from BGWL had CI values of 0.022 and 0.023, and are considered chemically representative of natural waters recharging the sand and gravel aquifer at the Goose Lake detailed study area.

Contamination index results from surface and groundwater samples collected in 1989 documented saline contamination from co-produced waters at twenty four of the sites that were resampled in 2009. To determine if a significant reduction in the CI values of contaminated sites had occurred between 1989 and 2009, only these sites were analyzed. Data from three additional sites were excluded due to differences in sample location (wetland B/E) and potential influences from a transport line break (wells 126 A and 126, see Discussion), resulting in a sample size of twenty one. The histogram and box plot of the difference in CI values between 2009 and 1989 (Fig. 21) show the data are long tailed due to the presence of one outlier (well 264 A), and a Kolmogorov-Smirnov normality test determined the data was not normally distributed. Therefore, a non-parametric analysis was required. There is convincing evidence that the difference in median CI values between 2009 and 1989 is non-zero (two-sided p-value = 0.014 from a Wilcoxon signed rank test with a V stat of 46). The estimated reduction in median CI value is -0.057, with a 95% confidence interval from -0.106 – -0.011.

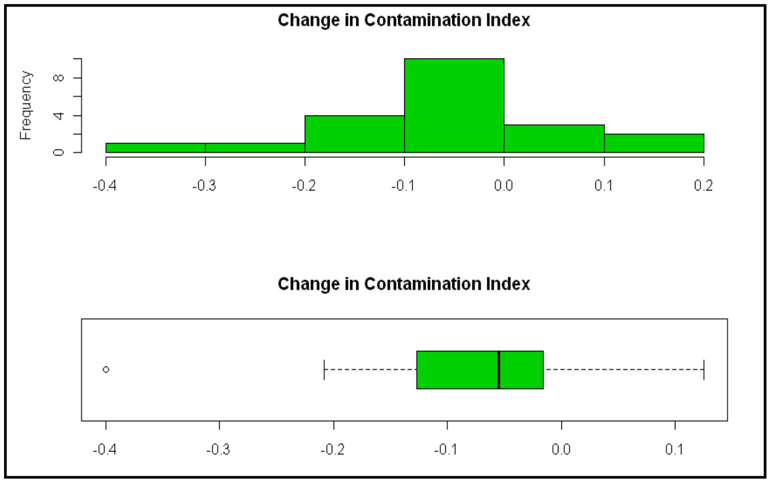

Figure 21. Histogram and box plot of the change in contamination index values from 1989 to 2009.

Geophysical Surveys

Several spatial interpolation methods were examined to display the geophysical data (krigging, natural neighbor, spline, and IDW) with IDW chosen since it is an exact interpolation method, forcing the produced surface to pass through all data points. Additionally, conductivity values changed rapidly in places in the field due to the geometry of the saline groundwater plumes. Therefore, a neighborhood of three points was required to accurately display the data.

EM-31 Surveys

EM-31 surveys in 2004 and 2009 measured elevated apparent conductivities near and down-gradient of several oil field sites within the Goose Lake detailed study area. The spatially interpolated surfaces from the 2004 surveys were created from 589 apparent conductivity readings and cover an area of 43.8 acres (Fig. 22). Results from the 765 measurements taken in 2009 are shown in Figure 23 and cover an area of 195.7 acres. Surveys from both years identified areas of elevated conductivities in the southern groundwater flow path in the slough below the Hammer M tank battery and at oil field site 128. The greatest conductivity measurements in 2004 and 2009 occurred below the tank battery, with measured values of 450 and 445 mS/m, respectively. Additionally, elevated conductivity measurements were measured in and around wetland 264 P and in the slough between wells 264 R and 264 T in 2009. Elevated conductivities were also seen in the northern groundwater flow path associated with oil field sites 124 and 126 in 2009, with a maximum value of 420 mS/m recorded west of wetland 124 F. Near West Goose Lake, conductivity readings were also elevated in 2009, with a high of 240 mS/m.

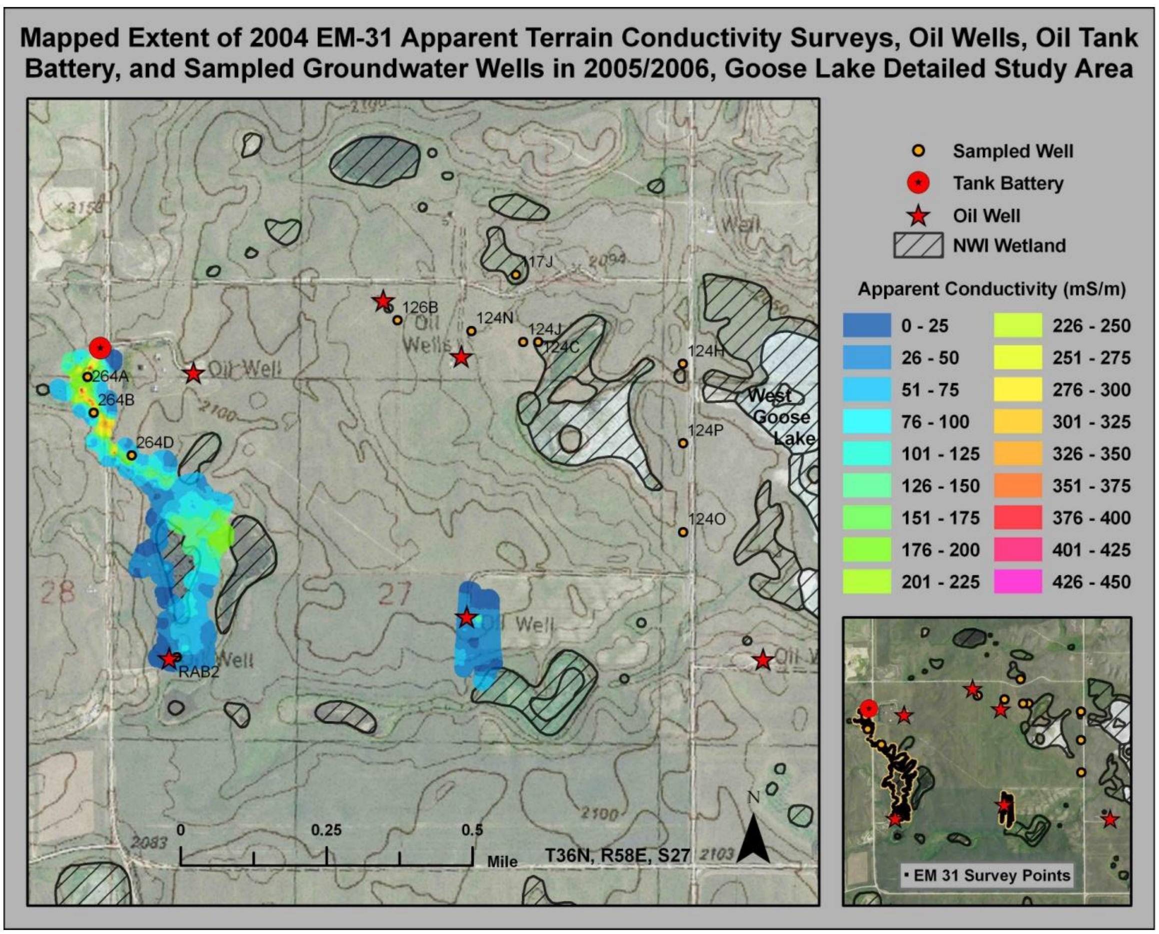

Figure 22. Map showing the locations of 2004 EM-31 survey points, the mapped extent of 2004 EM-31 surveys, oil wells, NWI wetlands, and type and location of water samples collected in 2005/2006. Inset map shows locations of EM-31 survey points.

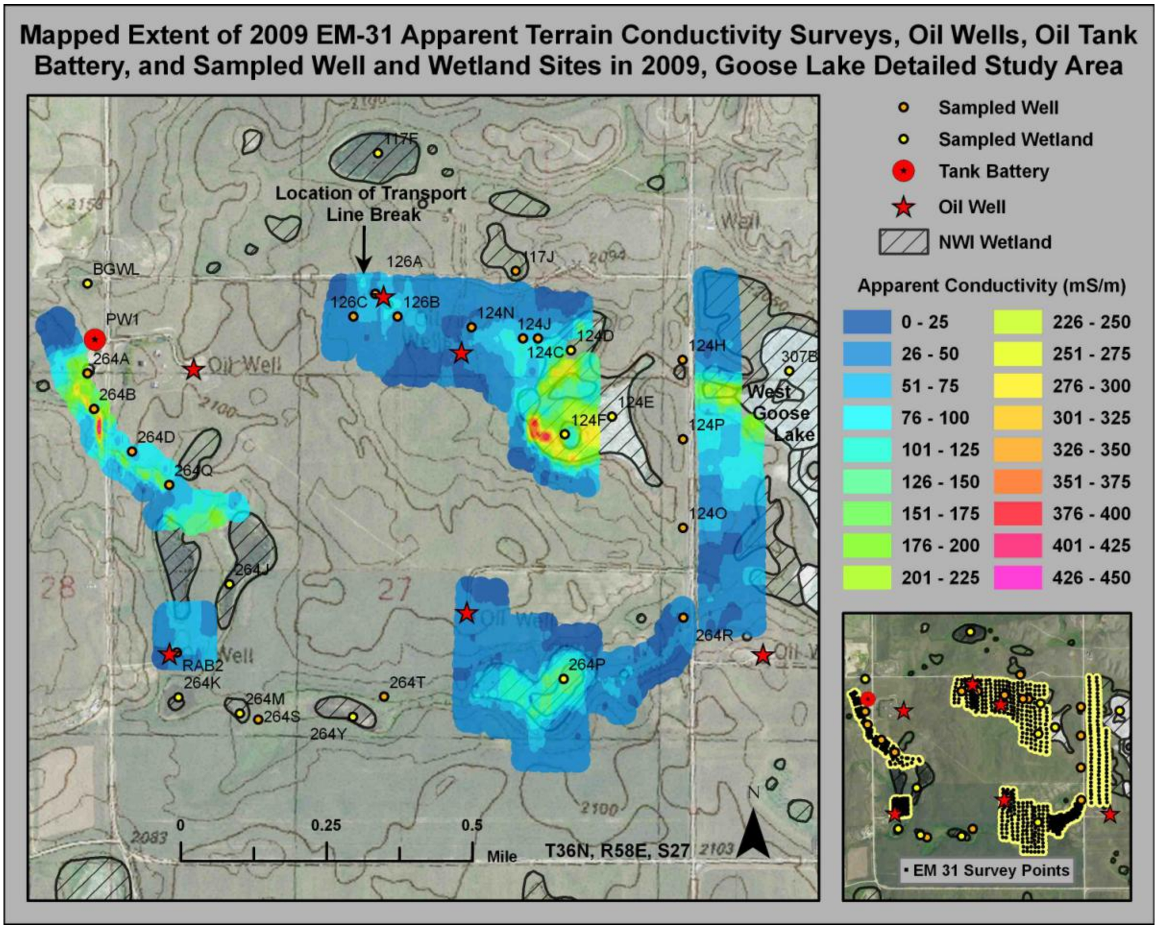

Figure 23. Map showing the locations of 2009 EM-31 survey points, the mapped extent of 2009 EM-31 surveys, oil wells, oil tank battery, NWI wetlands, and type and location of water samples collected in 2009. Inset map shows locations of EM-31 survey points.

<u>EM-34 Surveys</u>

During the 2009 geophysical surveys, the EM-34 was operated with the dipole positioned horizontally and vertically, with apparent conductivity measurements taken in each orientation at 10, 20, and 40 m intercoil spacings. Although six measurements were attempted at all 125 sampling sites, due to various factors (interference from pipelines, fences, data entry, soil properties, etc) not all readings were obtained. Spatially interpolated surfaces were produced only from successful readings, with 124 measurements used for the 10 and 20 m horizontal dipole maps and 125 measurements for the 40 m horizontal dipole map (Fig. 24). The spatially interpolated surfaces from the vertical dipole surveys (Fig. 25) were created from 119, 118, and 115 measurements in the 10, 20, and 40 m intercoil spacings, respectively. The spatially interpolated surface for the EM-34 surveys covers 228.7 acres. To maintain consistency between the EM-31 and EM-34 data, the same color ramp was used for all geophysical data.

The EM-34 apparent conductivity values were lower when measured in the vertical dipole orientation than in the horizontal dipole orientation. Maximum recorded values for the horizontal dipole were 280, 250, and 260 mS/m for the 10, 20, and 40 meter spacings, respectively. Corresponding maximum values in the vertical dipole position were 102, 93, and 108 mS/m. Unlike the previous EM-31surveys, only the 40 meter, horizontally aligned dipole position recorded the highest apparent conductivity value in the slough below the Hammer M tank battery data. The largest conductivities at both 10 meter spaced dipole positions, as well as the 20 meter spaced, horizontal dipole orientation, were recorded along the western edge of the contiguous measurements in the northern groundwater flow path, near wetland 124 E. For the 20 and 40 meter spaced, vertical dipole orientations, maximum conductivities were reported in the strip of measurements in the northern groundwater flow path, near wetland 124 F.

Background apparent conductivities at the Goose Lake detailed study area were determined from multiple readings and varied by equipment and operation mode. The locations used to determine background conductivities were selected from the survey points in areas expected to be uncontaminated; however due to the widespread extent of groundwater contamination and minimal overlap in uncontaminated areas, different

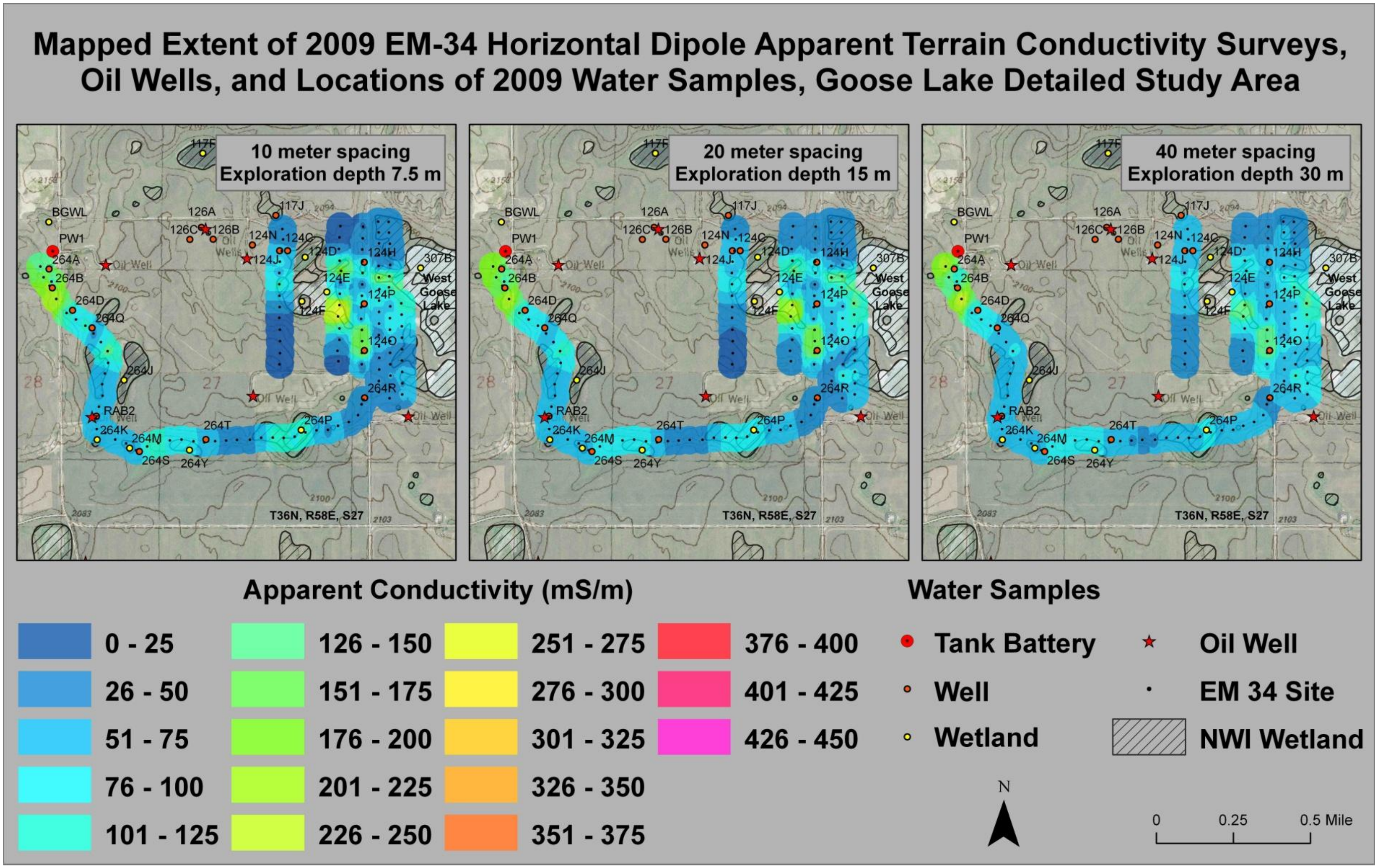

Figure 24. Maps showing the locations of 2009 EM-34 horizontal dipole survey points, mapped extent of 2009 EM-34 horizontal dipole surveys, type and location of water samples, oil wells, oil tank battery, and NWI wetlands. Left panel shows 10 m horizontal dipole spacing results. Middle panel shows 20 m horizontal dipole spacing results. Right panel shows 40 m horizontal dipole survey results.

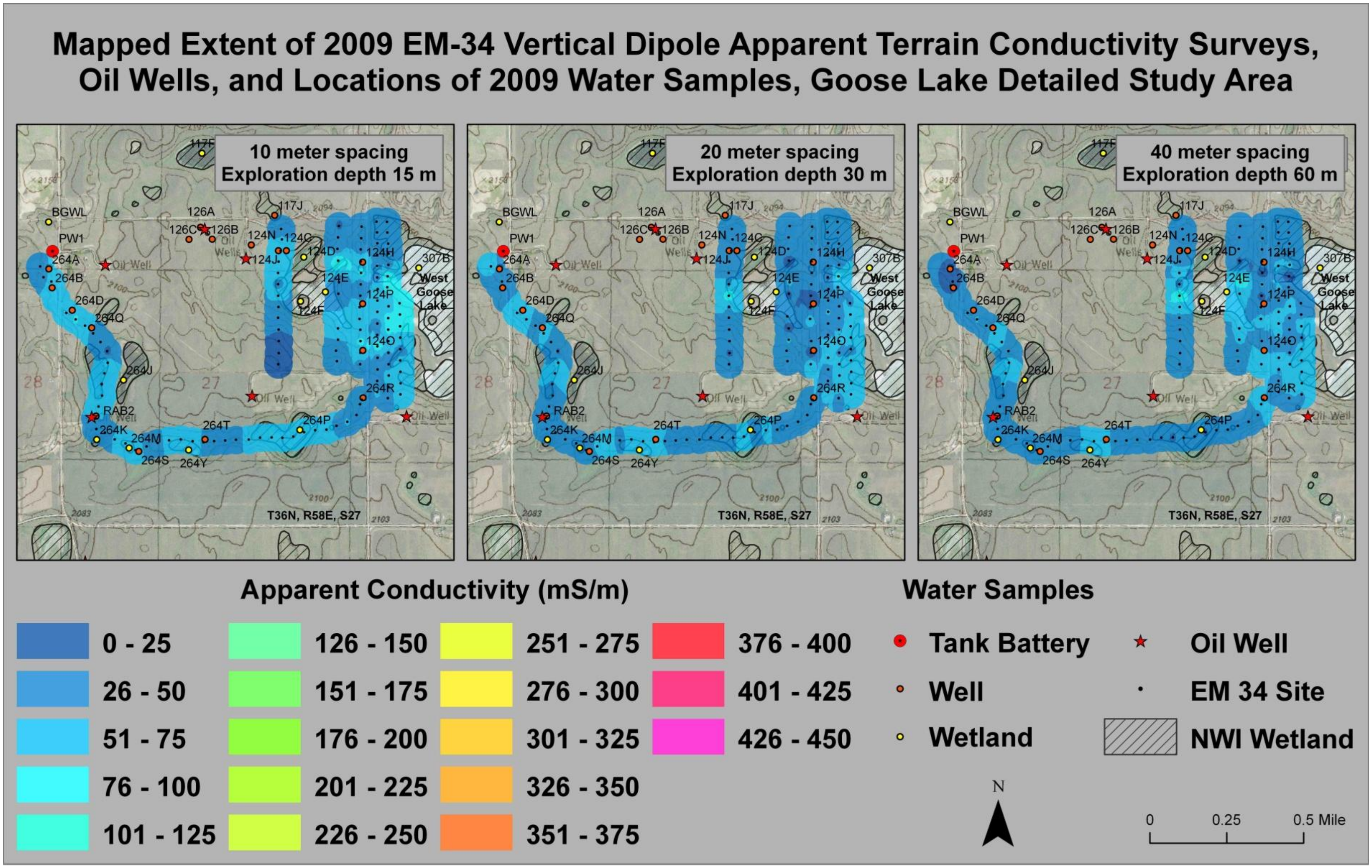

Figure 25. Maps showing the locations of 2009 EM-34 vertical dipole survey points, mapped extent of 2009 EM-34 vertical dipole surveys, type and location of water samples, oil wells, oil tank battery, and NWI wetlands. Left panel shows 10 m vertical dipole spacing results. Middle panel shows 20 m vertical dipole spacing results. Right panel shows 40 m vertical dipole survey results.

locations were used for the EM-31 and EM-34 measurements. Fifty measurements south of oil field site 126 and north of the Hammer M tank battery were used to determine the background conductivity measured by the EM-31, with a mean of 28 mS/m. Average background conductivities for the EM-34 were calculated from the four southwestern most survey points in the northern groundwater flow path. These four measurements were obtained on a ~15 m topographic rise where the overlying outwash deposits are likely unsaturated. Mean EM-34 conductivities in the horizontal dipole position, listed by increasing intercoil spacings, were 16, 22.25, and 36.75 mS/m. Respective EM-34 vertical dipole background conductivity values were 22.75, 30.5, and 42.5 mS/m.

EM-31 and EM-34 Comparison

Normalizing the recorded apparent conductivity values by dividing by the background conductivity results in the data being expressed as a response ratio, and allows comparison across different geophysical equipment with equivalent exploration depths. Figure 26 shows response ratio maps, calculated by normalizing the polygon values in the spatially interpolated maps, from the EM-31 and EM-34 ten-meter, horizontal dipole surveys. The maximum normalized conductivity measurement was smaller for the EM-31 compared to the EM-34 (15.8 and 17.5, respectively). The larger value for the EM-34 is likely caused by a low estimate of background conductivity, due to measurements taken in unsaturated outwash deposits. Areas with high response ratios were identified in the southern and northern groundwater flow paths.

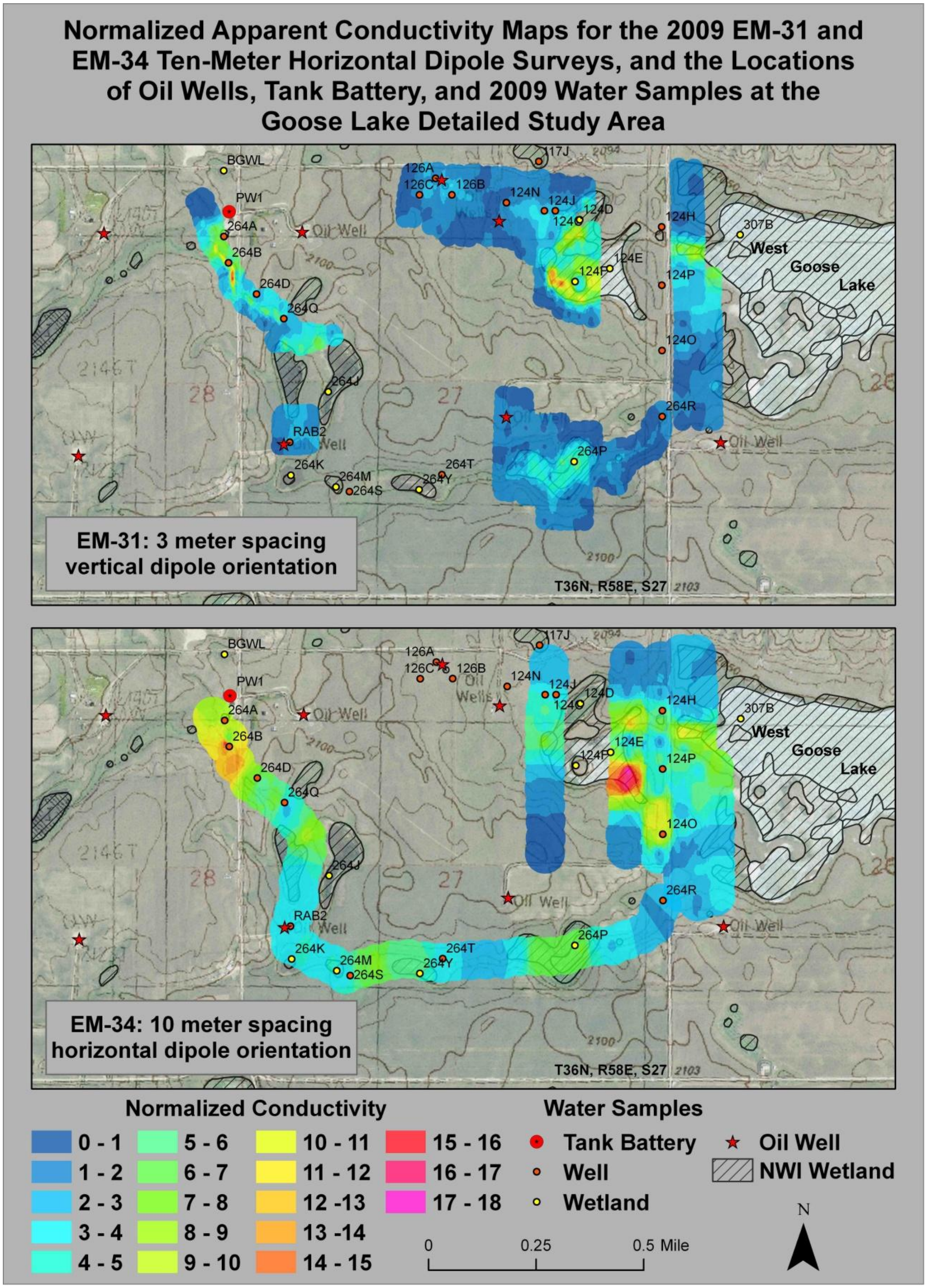

Figure 26. Normalized conductivity maps for 2009 EM Surveys. Top panel shows EM-31 vertical dipole results. Bottom panel shows EM-34 results from 10 meter spacings, with a horizontal dipole.

DISCUSSION

Hydrogeologic Investigations

Initial studies in 1989 established the hydrogeological framework of the Goose Lake detailed study area, documenting two major groundwater flow paths that converge near West Goose Lake in the saturated outwash deposits mantling the underlying glacial till. Surface and groundwater samples from 1989 documented widespread co-produced water contamination from several oil field sites in and adjacent to T36N, R58E, S27. Repeat sampling in 2005/2006 and 2009 also recorded extensive saline contamination to aquatic resources, with the temporal change in water chemistry illustrating the continued migration of saline contamination within the study site. Additionally, hydrogeological investigations in 2009 measured the hydraulic conductivities in sediments adjacent to six groundwater wells and then continually monitored groundwater elevations to allow comparisons to the hydraulic conductivity value and lateral hydraulic gradients, respectively, reported in 1989.

Hydraulic Conductivity

Hydraulic conductivities of near surface outwash deposits determined from the six rising head slug tests in 2009 (range: 0.18 – 13.50 ft/d and a geometric mean of 2.06 ft/d) were drastically lower than that determined by the 200-minute aquifer test conducted by Reiten and Tischmak at oil field site 124 in 1989 (100 ft/d, well name not published). Although the K value in 1989 was likely calculated from a well not tested in 2009, there are several other potential reasons for the differences in results. The slug tests remove only a fraction of the water volume of a long term aquifer test, resulting in much less of the aquifer is being subjected to pumping stress and hydrogeological characterization, with K values often being scale dependent (Rovey and Cherkauer, 1995). As a result the cone of influence from the 1989 pump test was much larger than the 2009 slug test and could have extended into higher permeability outwash deposits (i.e. a gravel stringer). Additionally, changes in the outwash thickness due to the hummocky topography would have a greater effect with the larger cone of influence. Thus, the difference in K values between 1989 and 2009 are likely due to differences in test design.

Despite the discordance of the 2009 rising head slug test results to the 1989 aquifer pump test, the rising head slug test data are useful in illustrating the wide range of variability in K across the Goose Lake detailed study area. Although the wells tested had screens completed in different ratios of outwash to till/lake clay (Table 4), there is no relationship between K and the length of screen in the outwash. In fact, the well completed entirely in outwash had the second lowest K value of 1.07 ft/day. As a result, the range in K values observed in the slug test data likely reflect the local heterogeneity in textural composition expected in outwash sediments, with lower K values related to lower degrees of sorting and increased silt content. Therefore, the slug test results illustrate the difficulty in applying a single K value to even a small area of glacial outwash deposits.

Hydraulic Gradient

Seasonal groundwater elevation changes were small, less than 2.5 ft, as seen in the hydrographs for the four wells with complete capacitance rod records (Fig. 16-A).

Although two of the capacitance rods failed to record complete data sets, they occupied the highest topographic position along the southern groundwater flow path (264 S) and the lowest topographic point in the northern flow path (124 H) illustrated in Figure 13. Groundwater elevation differences between these two wells and the nearest well equipped with a capacitance rod were approximately 19 and -7 feet, respectively, on the 25th of July, 2009. As all wells are completed in the same glacial outwash aquifer, it would seem unlikely that groundwater elevations in the wells with failed capacitance rods would have changed by such large magnitudes as to alter the observed local flow dynamics. The placement of only six rods across the entire study area precluded the ability to monitor local scale changes in groundwater flow that can occur in response to surface and groundwater interactions around wetlands (Winter, 2003). However, this would have little effect on the overall flow regime. Therefore, the shallow groundwater flow directions monitored within the Goose Lake detailed study area were consistent throughout the year and in agreement with the results of Reiten and Tischmak (1993).

Individual wells exhibited variations in the depth below land surface (Fig. 16-B). Rapid decreases in the depth to water signify a rise in the water table and are indicative of recharge events. There was a large spike in three of the six wells on August 15th in response to a large precipitation (1.25 inches reported approximately 20 miles southwest in Plentywood, MT) event not evident in the remaining wells. Here, meteoric waters appear to have only been able to recharge the underlying aquifer where the depth to the water table was not too great. Although the depth to the water table was nearly identical at wells 264 S and 124 J, only well 264 S shows this recharge event. This may relate to differences in vegetative cover due to prescribed burning by the USFWS in the southern half section in May 2009, but is more likely due to the hydrogeologic position of the groundwater wells. Both wells are roughly in the middle of their respective groundwater flow path, yet the depth to the water table is configured differently along the two flow paths (Table 5). In the northern groundwater flow path (124 J), the depth to the water table decreases down-gradient in contrast to the southern groundwater flow path (264 S) where the depth to the water table increases down-gradient. Thus, meteoric water from this precipitation event was able to infiltrate to the water table in the slough up-gradient of well 264 S, where the water table is close to the land surface, but did not occur above well 124 J, leading to a rise in the groundwater elevation at well 264 S and not 124 J.

Recharge events likely remobilize salts that have accumulated from evaporation above the water table (Reiten pers. comm., May, 2009). During the summer months and drought cycles, declines in the water table result in the evaporitic deposition of salts in the capillary fringe as groundwater is lost to evapotranspiration. As the water table in the aquifer rises in response to spring recharge, seen in early March in Figure 16-B, or precipitation events described above, these salts may be dissolved back into the shallow groundwater. Additionally, the highly variable climate in the PPR produces wet and dry cycles which can affect groundwater levels, with higher water tables during wet periods and lower levels during drought conditions (Winter, 2003). Therefore, individual precipitation events along with seasonal and climatic driven water level fluctuations may have the potential to remobilize salts that have been evaporatively concentrated above the saturated zone. This likely produces pulses of enhanced saline contamination during wet periods when the water table is relatively high; however, the addition of enough water may dilute the and the temporary hydrologic isolation and storage of some salts during dry phases when the water table is low. The highly variable response of the groundwater table to a single precipitation event, and the minor variations in seasonal recharge, illustrate the inhomogeneity in hydraulic response to these events that can be exhibited over a small geographic area.

Surface and Groundwater Chemistry

Fifteen of the seventeen groundwater monitoring wells, all eight wetlands, and West Goose Lake sampled in 1989 had CI values above the threshold of 0.035 indicating contamination from co-produced waters (Reiten and Tischmak, 1993). Two monitoring wells, 126 C and 264 T, had CI values representative of uncontaminated waters in 1989 (CI = 0.005 and 0.016, respectively). Relative to the positions of known saline groundwater plumes in 1989, well 126 C was up-gradient from chloride contaminated monitoring wells 126 A and 126 B in the northern groundwater flow path, while well 264 T was located down-gradient from the furthest extent of saline contamination in the southern groundwater flow path associated with the Hammer M tank battery. The topographically lowest sample collected in 1989 was surface water from West Goose Lake, which is assumed to be the convergence point of the southern and northern groundwater flow paths, and had a CI value of 0.107.

Changes in water chemistry were observed at all sampled locations between 1989 and 2009 and confirm the hypothesis that the lateral extent of saline contamination in the Goose Lake detailed study area has increased. Water samples collected in 2009 from the twenty six repeat sampling locations, and two additional sites (wetland 264 Y and well RAB 2), all yielded CI values greater than 0.035. Decreases in CI values were observed in twelve of the sixteen groundwater wells, six wetlands, and West Goose Lake between 1989 and 2009. CI values increased in three wetlands and five wells, including 126 C and 264 T. Between 1989 and 2009, the relative change in the chloride concentration (+/-) was followed by the same relative change in the CI value, with the exception of wetland 264 P. Here, the small increase in chloride (18 mg/L) was offset by a much larger increase in specific conductance (1,320 µS/cm). Well 264 A and West Goose Lake were the only sample locations to have an increase in specific conductance not accompanied by an increase in chloride concentration between sampling periods. These results show the value of the CI in determining the severity of chloride contamination as opposed to tracking chloride concentration or specific conductance alone, which would have improperly classified one (264 P) and three (264 A, 264 P, and 307 B) sites, respectively, as exhibiting increased contamination in 2009 relative to 1989.

Only three water samples, a surface water site in 2009 and two groundwater wells in 1989 had CI values below 0.035, and are considered representative of recharging and uncontaminated groundwaters in the surficial sand and gravel outwash aquifer at the Goose Lake detailed study area. A surface water and replicate sample obtained from a small wetland above the Hammer M tank battery in 2009 (BGWL, Fig. 15) had CI values of 0.023 for both samples. Similarly, the CI values for the two uncontaminated groundwater wells were 0.005 for well 126 C and 0.016 for well 264 T. Furthermore, the chloride concentrations from BGWL in 2009 (9 mg/L both samples) compare well to the

1989 chloride concentrations recorded in the two uncontaminated wells: 126 C and 264 T (7 and 12 mg/L, respectively). The specific conductance values seen in 1989 from wells 126 C and 264 T (1,293 and 761 µS/cm, respectively) are greater than those seen in BGWL (390 µS/cm). This is probably caused by the chemical evolution of groundwaters that increases TDS with sulfate salts, but does not concentrate chloride salts (Custer, 1976). Therefore, the water chemistry seen in the uncontaminated samples can be used a baseline for the expected natural water chemistry at the Goose Lake detailed study area.

Southern Groundwater Flow Path The change in CI values in the southern groundwater flow path monitoring wells between 1989 and 2009 follow the classic pattern of decreasing contamination up-gradient (in highly contaminated areas) and increasing contamination down-gradient (in minimally or uncontaminated areas) expected if the saline groundwater plume is migrating laterally in the shallow groundwater flow system (Figs. 20 and 27-A). Error bars shown in Figure 20 are calculated as the square root of the sum of the squared measurement accuracies of chloride concentrations and specific conductance measurements, with 10% accuracies for MBMG data and 5 % accuracies for NWQL data. The Hammer M tank battery, in the upper part of the flow path, was the source of much of the contamination through reserve pits and discharges of produced water (Reiten and Tischmak, 1993). Although leachates are still being generated from salts present at this site, additional inputs of chloride likely ceased prior to 1989. However, the simple relationship observed in the groundwater chemistry between 1989 and 2009 is not seen when combined with surface water data (Fig. 27-B).

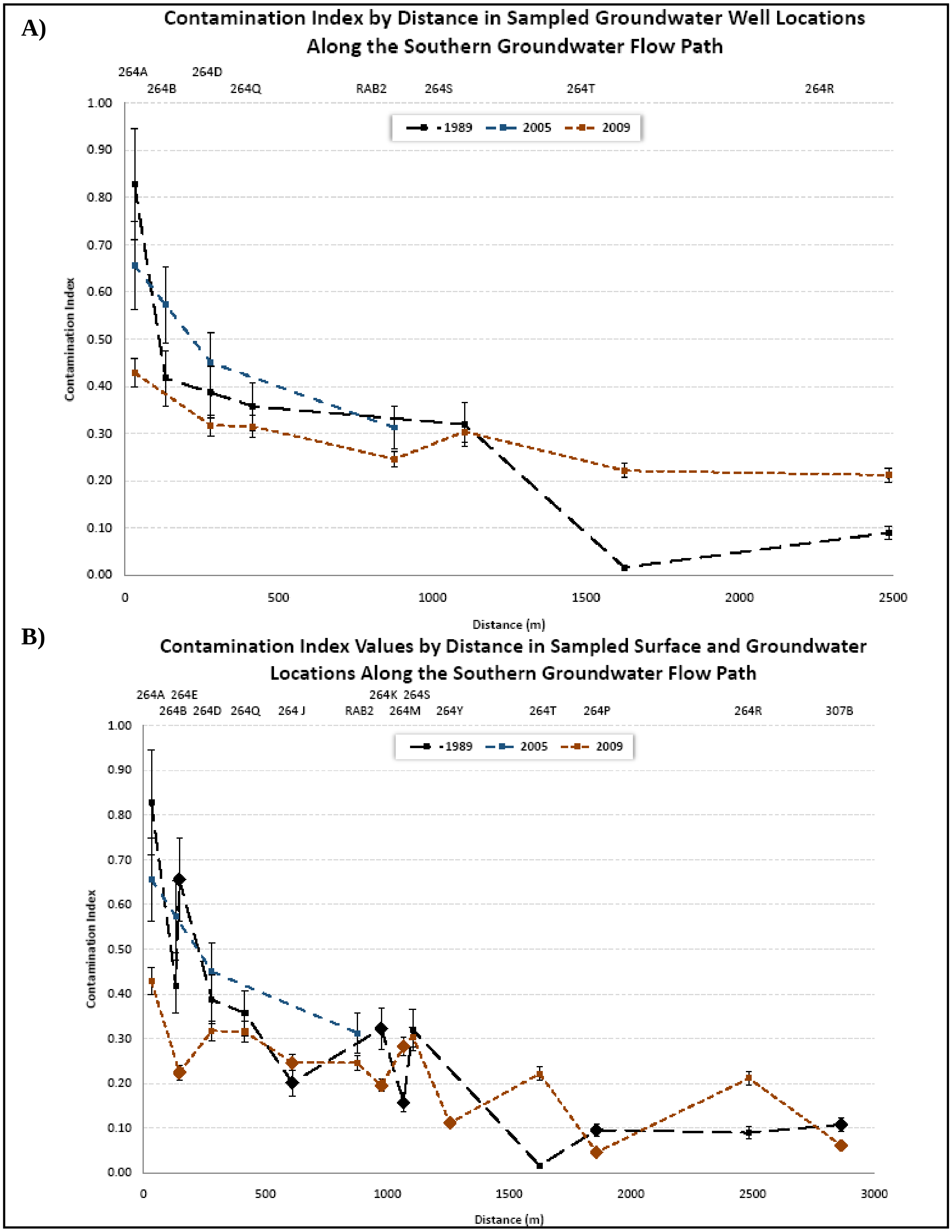

Figure 27. Panel A: 1989, 2005, and 2009 CI values in sampled groundwater wells in the southern groundwater flow path by distance from the Hammer M tank battery. Panel B: 1989, 2005, and 2009 CI values for sampled surface water sites (larger diamond symbols) and groundwater wells (smaller square symbols) in the southern groundwater flow path by distance from the Hammer M tank battery.

The changes in the groundwater chemistry in the southern flow path support the down-gradient transport of contaminants. Notice the steady decline in the CI value at well 264 A (immediately below the Hammer M tank battery) between the three sampling periods (Fig 27-A). This is the result of fresher waters recharging the aquifer in the slough above and flushing the saline contamination down-gradient. Well 264 A had the highest chloride concentrations of all sampled sites in 1989 (66,900 mg/L), 2005/2006 (30,841 mg/L) and 2009 (39,300 mg/L), and the second largest reduction in CI value (-0.400) between 1989 and 2009. The lateral transport of this highly contaminated groundwater is likely the cause of increased CI values observed at wells 264 B and 264 D in 2005 relative to 1989. This pulse of saline groundwater appears to not have reached well RAB2 or had been diluted back to 1989 levels past well 264 D, as the 2005 CI value at well RAB 2 was below that of the two closest wells (264 Q and 264 S) in 1989. Continued flushing of the highly contaminated groundwater by lower salinity recharging waters is illustrated by the reduction in CI values between 2005 and 2009 for the three groundwater wells sampled in 2005 (264 A, 264 D, and RAB 2). The next three wells down-gradient from well 264 A that were sampled in 1989 and 2009 (264 D, 264 Q, and 264 S) also showed a reduction in CI values, however, the reductions steadily decline (-0.070, -0.043, and -0.015, respectively). Further down-gradient, the precipitous drop in the 1989 CI values between wells 264 S and 264 T (dashed black line, Fig. 27-A) clearly shows the saline groundwater plume associated with the Hammer M tank battery had yet to reach well 264 T. In 2009, well 264 T had a CI value nearly fourteen times greater than 1989, for a total change of 0.206. The last well in the southern groundwater flow path, well 264 R, which had the lowest CI of any contaminated water sample in 1989 (0.089), had an increase in the CI value of 0.123. Between 1989 and 2009, the leading edge of the saline groundwater plume associated with the Hammer M tank battery moved past well 264 T, and possibly well 264 R, as evidenced by the large increases in CI values in these wells.

The surface water chemistry observed in wetlands within the Goose Lake detailed study area, and broader PPR, are influenced by hydrologic position and climatic conditions (Winter et al., 2003), which helps explain the scatter in CI values seen in Figure 27-B relative to Figure 27-A. Figure 28 shows the regional groundwater flow conditions that commonly develop in the PPR, with local and intermediate flow regimes probably dominating in the southern groundwater flow path. From the water table configuration and aerial photos, wetlands in this flow path likely occupy all hydrologic positions, with recharge wetlands up-gradient, to flow-through and discharge wetlands down-gradient. Additionally, annual precipitation is highly variable (Fig. 29), with evaporation exceeding precipitation in the PPR causing many of the temporary and seasonal wetlands to go dry during the summer (Winter et al., 2003). During this drawdown phase, surface waters often experience evaporative concentration of salts. Therefore, the interaction of the hydrologic position and climates can result in much higher variability in surface water chemistry relative to groundwater chemistry.

The greater variability of surface water chemistry to groundwater chemistry is illustrated by examining CI values from field parameters collected over 1989-1990 for well 264 S and wetland 264 M, less than 50 m apart (Fig. 15 and Table 8). Wetland

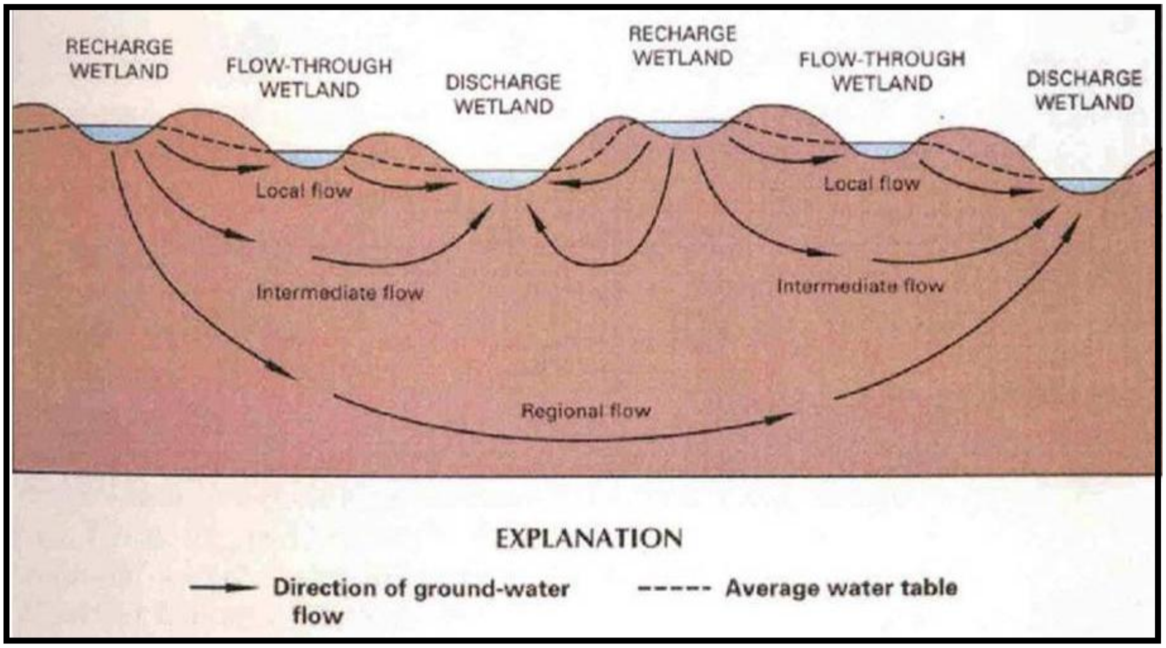

Figure 28: Illustration of groundwater flow paths that commonly develop in the PPR (Created by Donald Rosenberry pers. comm., 2010, unpublished).

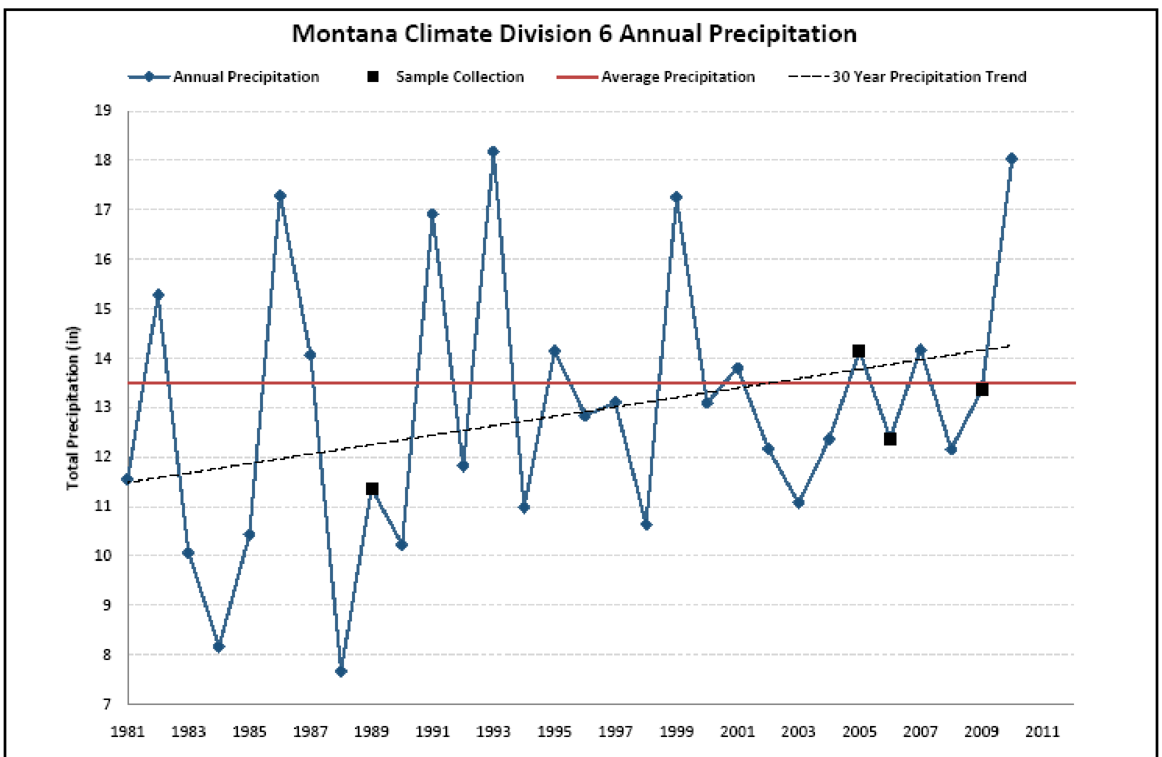

Figure 29. Annual precipitation totals for Montana climate division 6. Data from http://www.ncdc.noaa.gov/temp-and-precip/time-series/index.php and accessed on 2/4/2011.

Table 8: Field parameters from site visits in 1989 and 1990 for well 264 S and wetland 264 M (data from Reiten and Tischmak, 1993).

Well 264 S				Wetland 264 M			
Sample Date	**Chloride (mg/L)**	**Spec. Cond. (µS/cm)**	**CI**	**Sample Date**	**Chloride (mg/L)**	**Spec. Cond. (µS/cm)**	**CI**
Jun-89	4000	10000	0.400	**May-89**	140	890	0.157
Jun-89	3040	8630	0.352				
Jul-89	3380	8460	0.400				
Oct-89	2850	8470	0.336	**Oct-89**	2560	7270	0.352
Oct-89	2940	8640	0.340				
Jun-90	4182	10930	0.383	**Jun-90**	5946	8190	0.726
Average	3399	9188	0.369	**Average**	2882	5450	0.412
Std Dev	568	1035	0.029	**Std Dev**	2916	3976	0.289

264 M is a dugout created for stock water purposes and acts like a discharge wetland, receiving much of its input from groundwater. This wetland contains water when many of the surrounding temporary and seasonal wetlands have gone dry. The water levels in the temporary and seasonal wetlands are often dependent on precipitation and runoff. Additionally, precipitation totals were less in 1990 than 1989, with both years below average (Fig. 29), limiting groundwater recharge up-gradient and meteoric dilution of surface waters in wetland 264 M. The CI values at wetland 264 M steadily increased from 0.157 to 0.726 (std dev = 0.289) but there was no discernable pattern in well 264 S which fluctuated over a range of 0.336 – 0.400 (std dev = 0.029). Thus, the increased CI values in wetland 264 M probably relate more to evaporative fluxes on surface water than changes in groundwater chemistry, which remained fairly constant at well 264 S.

The observed variability of the surface water chemistry in the southern groundwater flow path makes it difficult to relate the changes seen in surface waters to the changes observed in groundwater chemistry; however, two observations are noteworthy. First, the 2009 CI values from sampled wetlands below 264 B/E (0.224) decrease down-gradient (264 J = 0.247, 264 K = 0.195, 264 Y = 0.111, 264 P = 0.045), with the exception of wetland 264 M (0.282, Figs. 15 and 27-B). Discordance of wetlands 264 B/E and 264 M are likely the result of sampling and groundwater flow position, respectively. The sample obtained from wetland 264 B/E was clearly connected with surface waters, but this was mainly runoff flowing in the slough, and not the water impounded behind the culvert (beneath the county road between wells 264 B and 264 D) where field parameters for wetland 264 E were obtained in 1989. Wetland 264 M is a dugout and water in this wetland likely had a much larger input from the shallow groundwater system than the other wetlands in this flow path which are more dependent on precipitation and runoff, explaining the elevated chloride concentration and CI value. Second, although both wells 264 T and 264 R had increased CI values in 2009, the wetland between (264 P) yielded a decrease in CI and had the lowest chloride concentration (98 mg/L) and CI (0.045) of any contaminated sample, supporting the greater influence of meteoric water on the spring chemistry of wetlands in the southern groundwater flow path, relative to groundwater.

<u>Northern Groundwater Flow Path</u> The northern groundwater flow path consists of two smaller flow paths that converge along the northern boundary of T 36N, R 58E, S27 (Figs. 11 and 13). Groundwater chemistry north of the bifurcation is influenced by oil field site 117, whereas groundwater chemistry westward is influenced by oil field sites 124 and 126. Groundwaters from the two limbs combine northeast of oil field site 124, and flow towards West Goose Lake. Figure 30 shows the 1989, 2006, and 2009 CI

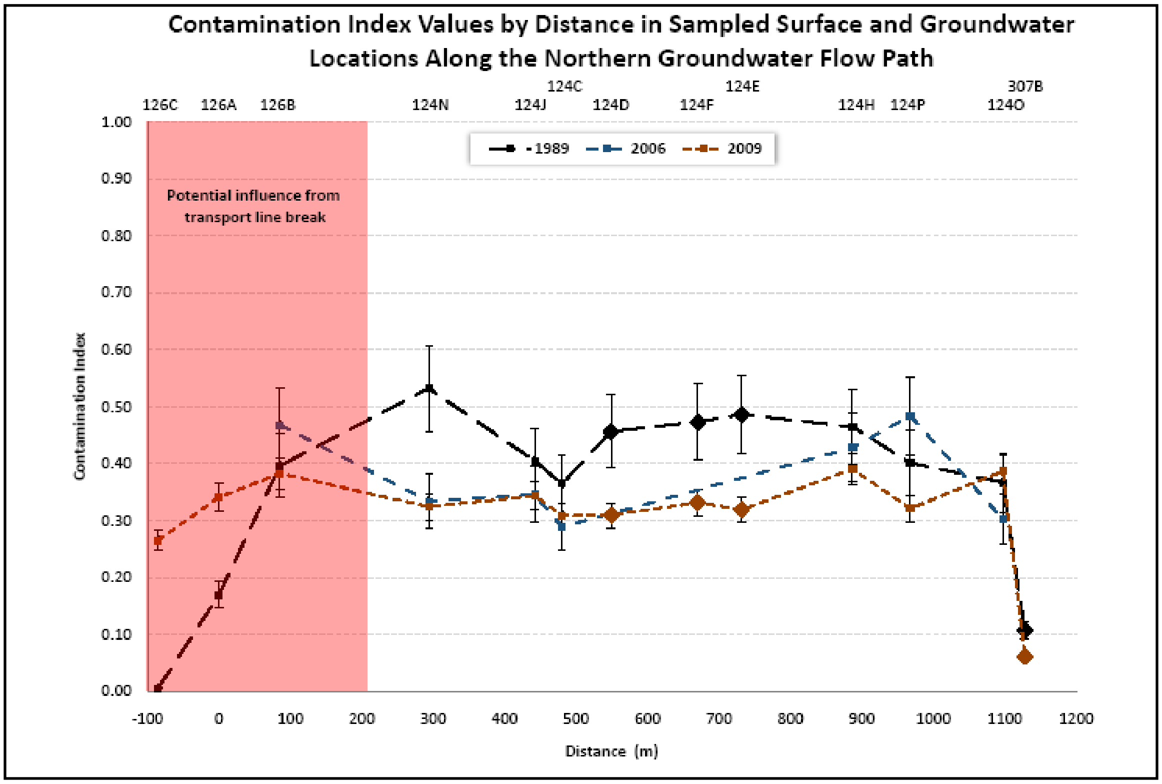

Figure 30. 1989, 2006, and 2009 CI values for sampled surface water sites (larger diamond symbols) and groundwater wells (smaller square symbols) in the northern groundwater flow path and by distance from well 126 A. Wetland 117 F and well 117 J are not shown.

values from surface and groundwater samples plotted by distance from well 126 A, the furthest up-gradient well contaminated in 1989. Error bars shown are calculated as the square root of the sum of the squared measurement accuracies of chloride concentrations and specific conductance measurements, with 10% accuracies for MBMG data and 5 % accuracies for NWQL data. Wetland 117 F and well 117 J are not shown as they are not influenced by the contamination from oil field sites 124 or 126, but rather the hydraulic gradient channels groundwater from these sites into the main flow path above wells 124 C and 124 J (Fig. 13). Although wetland 117 F showed an increase in the CI value between 1989 and 2009, the CI value at well 117 J decreased (Fig. 20). A dugout and groundwater well (117 D and 117 E, respectively; Fig. 14, site #117) up-gradient of wetland 117 F had elevated CI values in 1989 (0.360 and 0.193, respectively) and are likely the source of increased chlorides and CI value in wetland 117 F in 2009. However, the 2009 chloride concentration (1,550 mg/L) and CI value (0.274) from well 117 J were lower than wells 124 C (2,330 mg/L and 0.309) and 124 J (6,150 mg/L and 0.390), indicating the influx of groundwater from the north is less contaminated than the groundwater in the main flow path. As in the southern groundwater flow path, flow is likely local and intermediate; however, the three wetlands shown in Figure 30 (124 D, 124 E and 124 F) are primarily discharge wetlands as evidenced by salt tolerant vegetation (*Salicornia*) and aerial photos. These three wetlands had very similar CI values in both 1989 (CI avg = 0.472, std dev = 0.015) and 2009 (CI avg = 0.320, std dev =0.011) with values comparable to nearby groundwater wells, and will be discussed in conjunction with the groundwater results.

The increases in CI value for wells 126 A and 126 C from 1989 to 2009, and well 126 B from 1989 to 2006 (Fig. 30), may not be due to leachates generated from the buried reserve pit present at oil field site 126, but from a ruptured transport line that occurred in March of 2006 (Fig. 31). A break in a transport line leading to the Hammer M tank battery that parallels the county road resulted in an unknown volume of co-produced water and oil being discharged immediately northwest of oil field 126 (Fig. 23). These fluids proceeded to flow overland before filling the cattle guard on the site access road, and likely infiltrating into the near surface sediments and groundwater. For reference, a co-produced water sample from the Hammer M tank battery in 2009 had a

Figure 31. Photo A shows discharged oil and co-produced water from a transport line rupture north of oil field site 126 in March 2006. Photo B shows were oil and co-produced water flowed overland before filling the cattle guard immediately north of oil field site 126 (Photos courtesy of Mickey McCall, SCCD).

chloride concentration of 121,000 mg/L and a CI value of 0.543, although this sample had been separated from the oil. Figure 31-A was taken following the spill, whereas Figure 31-B was taken after repairs had been completed on the line. The location of the spill was up-gradient from wells 126 A, 126 B, and 126 C, with only well 126 B sampled in 2006. Well 126 B is also down-gradient of the buried reserve pit at the site and had an increased CI value between 1989 and 2006 (0.396 and 0.468, respectively), but this resulted from a greater reduction in specific conductance (-26,638 µS/cm) than chloride concentration (-8,218 mg/L). The reduction in the CI value at well 126 B between 2006 and 2009 (- 0.085) was due to a much larger relative increase in the specific conductance (8,400µS/cm) than in the chloride concentration (418 mg/L). Wells 126 A and 126 C experienced increases in chloride concentration (5,040 and 957 mg/L, respectively) and CI value (0.171 and 0.260, respectively) between 1989 and 2009. As the extent of contamination from the transport line break cannot be assessed, results from oil field site 126 are considered unrepresentative of preexisting saline groundwater migration.

When oil field site 126 is removed, the changes observed in CI values in Figure 30 follow the expected pattern of decreased values up-gradient and increased values down-gradient if saline groundwater plumes are migrating laterally. The CI values decreased in all sampled wells between 1989 and 2006, and in all sampled sites between 1989 and 2009, with the exception of wells 124 P and 124 O, respectively. However, in contrast to the southern groundwater flow path, groundwater wells in the northern groundwater flow path that showed reduced CI values between 1989 and 2009 exhibited no obvious relationship between the magnitude of reduction and distance from the contamination source. Changes in CI values between 2006 and 2009 were more subtle, with 2009 values from wells 124 C, 124 J and 124 N comparable to 2006 values, decreased CI values at wells 124 H and 124 P, and an increase in the CI value at well 124 O. A consistent trend of decreasing CI values through 1989, 2006, and 2009 was observed in the up-gradient wells (124 N – 124 H), with the exception of 124 C, where the 2009 CI value (0.309) was slightly higher than the 2006 CI value (0.289). Additionally, the three wetlands (124 D, 124 E and 124 F) show similar reductions in CI between 1989 and 2009. This is interpreted as recharging meteoric and groundwater with much lower salinity than the contaminated groundwater flushing the saline contamination down hydraulic gradient.

The changes in CI values observed in the lowest wells down-gradient, 124 P and 124 O, are explained by examining water chemistry from 1989, 2006, and 2009 (Fig. 30). Well 124 H had the greatest chloride concentration (36,500 mg/L) of any sample in the northern groundwater flow path in 1989, and a CI value of 0.464. Additionally, wetlands 124 D, 124 E, and 124 F, had comparable CI values to well 124 H (0.456, 0.473, and 0.486, respectively), albeit with lower chloride concentrations. In 2006, well 124 P recorded the greatest chloride concentration (20,126 mg/L) and CI value (0.484) in this flow path. The increased chloride concentration and CI value at well 124 P between 1989 and 2006 (3,826 mg/L and 0.082, respectively) are the result of the highly saline groundwater up-gradient, evidenced by well 124 H and wetlands 124 D, 124 E, and 124 F, moving past well 124 P but not yet reaching well 124 O, where chloride concentration and CI value decreased. Well 124 O recorded the second highest chloride concentration and CI value in 2009 (17,200 mg/L and 0.387, respectively), only slightly less than well 124 H (19,500 mg/L and 0.390). Between 2006 and 2009, this pulse of contamination moved past well 124 O, evidenced by chloride concentration and CI value increases of 14,626 mg/L and 0.084, respectively.

West Goose Lake West Goose Lake, which has no surface outlest, is the lowest of three large lakes east and southeast of the detailed study area (Fig. 10) and is a major discharge basin, evidenced by abundant mirabilite and thernardite (sodium sulfate salts) deposits in and around the lake bed. Despite the down-gradient increases observed in chloride concentrations and CI values in the lowest groundwater wells in the southern and northern groundwater flow path between 1989 and 2009, West Goose Lake exhibited a minor decrease in chloride concentration from 4,480 to 4,050 mg/L and a decrease in the CI value of -0.047. The reduction in CI value is more likely due to the large increase in specific conductance (24,920 μS/cm) than the small reduction in chloride concentration (-430 mg/L), which remained fairly constant. Including 2009, precipitation was below average for seven of the ten years prior to sample collection in 2009 (Fig. 29), and may explain the sharp increase in specific conductance. Long intervals of below average precipitation can result in increased specific conductance values in PPR wetlands (LaBaugh and Swanson, 2003). West Goose Lake and the lakes above, which probably provide a significant contribution to West Goose Lake via groundwater, would likely have experienced evaporative concentration of salts and increased specific conductance during this period of reduced precipitation, explaining the large increase between 1989 and 2009. Additionally, evaporatively deposited salts can also be removed from lake basins through aeolian deflation, especially during drought periods when the water levels decrease, exposing more of the lake bed (LaBaugh and Swanson, 2003). The author witnessed such an event in the spring of 2008 at a naturally saline lake north of Westby, Montana, where plumes of aeolian suspended sediments continually blew from the basin. Although not observed, similar deflation likely occurred at West Goose Lake which is less than five km away. Therefore, due to the potential influences from the surrounding lakes and climate on the chemistry of West Goose Lake, the reduction in CI value between 1989 and 2009 (Fig. 20) is not viewed as providing conflicting evidence to the down-gradient transport of saline contamination in the shallow groundwater system.

<u>Groundwater Flow Path Comparison</u> Although the changes in CI values followed the same general trends in the northern and southern groundwater flow paths, distinct differences exist in the down-gradient CI profiles (Figs. 27 and 30). Assuming the increased contamination seen at oil field site 126 is related to the transport line break, the remainder of surface and groundwater sites in the northern flow path exhibited decreases in CI values between 1989 and 2009, with the exception of the lowest groundwater well (124 O). Similarly, reductions in CI values occurred in all up-gradient wells in the southern flow path, with increases in the last two down-gradient wells (264 T and 264 R). In contrast, wetlands in the southern flow path exhibited a mixture of increased and decreased CI values that were independent of the CI changes in nearby wells. Wetlands in the northern flow path are likely discharge wetlands compared to the predominantly recharge and flow-through wetlands in the southern flow path as evidenced by salt tolerant vegetation (*Salicornia*) and aerial photos. Additionally, many of the wetlands in

the southern flow path are in hydrological connection with each other through overland flow in the slough during the spring. The concordance of CI values between wetlands and nearby wells in the northern flow path (Fig. 30) not seen in the southern flow path (Fig. 27-B), may be due to differences in wetland type, with the discharge wetlands receiving a stronger influence from groundwater on spring chemistry relative to runoff and precipitation dependent recharge and flow-through wetlands. However, the consistency of the CI values in the discharge wetlands and groundwater wells in the northern flow path is likely a temporal phenomenon as shown by the seasonal variability in wetland 264 M (Table 8).

Unlike the southern flow path, where the CI values of contaminated sites were highly variable (0.045 – 0.429) and generally decreased with distance from the contamination source (Fig. 27), the CI values in the northern groundwater flow path were less variable (0.259 – 0.390) and showed no obvious relationship to distance from source (Fig. 30). In fact, one of the striking features of the 2009 CI results is the consistency in CI values throughout the northern groundwater flow path. From the hydrograph results (Fig. 16), there was potential for more frequent recharge events due to the shallower depth of the water table in the up-gradient end of the southern flow path relative to the northern flow path. The repeated fluctuations in the water table near the Hammer M tank battery likely result in enhanced leachate production during recharge events, and frequent pulses of highly saline surface and groundwater. The local topography channels runoff from the spring thaw and intense precipitation events into the slough were it can infiltrate into the shallow groundwater system or flow into lower wetlands and dilute the pulses of contamination, explaining the decreasing CI trend with distance in the southern flow path. In contrast, near the oil field sites in the northern groundwater flow path, the greater and more consistent depth of the water table likely reduces the variability of leachate production. Additionally, no major drainage system exists to collect and channel runoff waters near the main contaminant sources. As a result, there is likely a more constant release of saline contamination, transported almost entirely by groundwater, yielding the relatively consistent CI values seen in the northern groundwater flow path.

Natural Attenuation in Contaminated Sites The differences in CI values observed in previously contaminated surface and groundwater sites were statistically significant using the Wilcoxon sign rank test, with a twenty year median reduction -0.057. Without knowing the initial CI values when the last of the co-produced water contamination was introduced, the nature of reductions in CI values (i.e. linear, exponential…) cannot be ascertained from the two measurement points; however, assuming that the reduction in CI continues at the same linear rate, these values can be used to estimate the minimum amount of time required for CI values to drop below 0.035 using the following equation, where T is time in years, CI_C is the current CI value, and M is the median reduction in CI value:

$$T = ((CI_C - 0.035) / |M|) \times 20 \quad (1)$$

The largest CI values in the northern and southern groundwater flow paths in 2009 were recorded at wells 124 H (0.390) and 264 A (0.429), respectively. Applying equation 1, wells 124 H and 264 A would continue to exhibit chloride contamination for another 125 years and 138 years. The above equation does not consider the fate of contaminants; however, the increases in down-gradient CI values evidenced by wells 124 O, 264 T, and 264 R demonstrated that contaminants are transported laterally (Fig. 20). Therefore, reductions in the CI values in the most contaminated sites would likely be offset by down-gradient increases. Since the majority of co-produced water contamination at the Goose Lake detailed study area was introduced between 1967 and 1974, this implies that saline contamination from oil and gas development to surface and groundwaters in outwash deposits within the PPR could persist for 180 years after initial contamination (2009 – 1967 = 42, 42 + 138 = 180 years). These results are similar to those of Murphy et al., (1998), who concluded leachates would be generated for tens to hundreds of years at a North Dakota oil field site in glacial till, and further attest to the longevity of contamination to aquatic resources from co-produced waters in the PPR.

Geophysical Surveys

Geophysical results in 2004 and 2009 compared well to groundwater chemistry results, with elevated apparent conductivity readings (> 50 mS/m) measured near the majority of groundwater wells with high chloride concentrations and CI values, illustrated by the EM-34 10 m horizontal and vertical survey results (Fig. 32). The linear correlations between groundwater chemistry and apparent conductivity were not excellent, likely due to the influence of changes in the outwash thickness and depth to the water table on the conductivity measurements. However, the presence of elevated conductivities associated with co-produced water contamination validates the use of spatially interpolated EM data to delineate saline contaminated groundwater plumes.

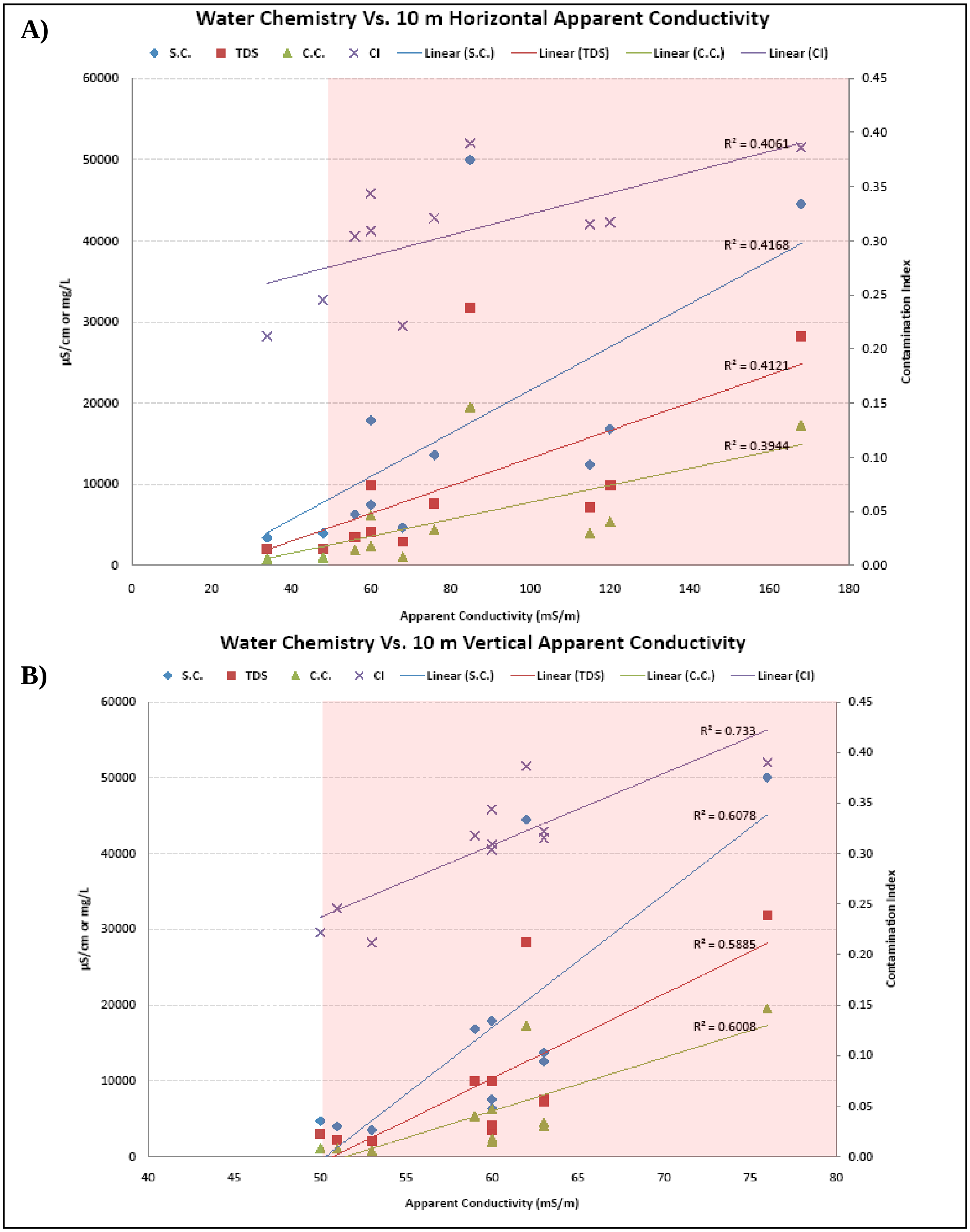

Figure 32. Groundwater well chemistry wells plotted against apparent conductivity measurements from the EM-34 10 m horizontal (A) and vertical (B) dipole surveys. S.C. – Specific conductance (µS/cm), TDS – Total dissolved solids (mg/L), and C.C. – Chloride concentration (mg/L). Red shading shows apparent conductivity values above 50 mS/m.

Although, it should be noted that apparent conductivity increases with clay content and soil saturation, often resulting in elevated apparent resistivity in and around wetland basins.

EM-31 Surveys

The EM-31was used to delineate the lateral extent of groundwater contamination within the Goose Lake detailed study area in 2004 and 2009 (Figs 22 and 23). Conductivity readings ranged from less than 10 mS/m to roughly 450 mS/m in both years, with values < ~ 30 mS/m considered representative of the background conductivity. A saline plume of contaminated groundwater emanating from the Hammer M tank battery and migrating down the slough below was suggested from both surveys. There is a rapid drop to background conductivity readings in areas hydraulically up-gradient, northeast and northwest of well 264 A in 2004 and 2009, respectively, confirming initial work documenting the tank battery as the source of contamination. Elevated conductivities associated with this plume were evident as far down-gradient as surveyed, near well RAB 2. Additionally, an area of elevated conductivity was measured near oil field site 128 (the furthest east survey) in 2004. In 2009, the surveyed area at oil field site 128 was expanded to include the lower end of the southern flow path.

Although the EM-31 spatially interpolated conductivity maps from 2004 and 2009 are similar (Figs. 22 and 23), some subtle differences are worth noting. First, several small areas of apparent conductivity values greater than 350 µS/cm were measured in the slough below the tank battery in 2004 (Fig. 22) compared to one larger area behind the culvert under the county road between wells 264 B and 264 D in 2009 (Fig. 23). The culvert is elevated above the slough, allowing flow though only when the water reaches a certain depth, otherwise runoff is impounded in the slough and infiltrates into the shallow groundwater or evaporates. Precipitation was less in 2004 than 2009 and followed a longer period of lower than average precipitation (Fig. 29). The consistently greater and closer to average precipitation preceding 2009 likely resulted in more runoff events capable of impounding water in the slough. The subsequent infiltration of dissolved salts to the groundwater may explain the single area of high apparent conductivities above the culvert in 2009 (Fig. 23). Second, a tongue shaped lobe of elevated conductivities (50 – 106 mS/m) was visible north of well RAB 2 in 2004, with conductivity values decreasing around the edges (Fig. 22). In 2009, elevated conductivities (50 -77 mS/m) were recorded running throughout the small area surveyed north of well RAB 2 (Fig. 23). The reduction in magnitude and lateral expansion of the area of elevated conductivities may be due to flushing of contaminated groundwater between 2004 and 2009 supported by the reduction in chloride concentration and CI value at well RAB 2 (Table 6). Finally, the area of elevated conductivity (150 – 180 mS/m) eastward of oil well 128 in 2004 (Fig. 22) was not recorded in 2009 (Fig. 23); however this is likely due to differences in survey lines and measurement locations (inset maps in Figs. 22 and 23). Sampling density was much greater near this site in 2004 than in 2009 and did not follow any standardized pattern, resulting in 135 data points used to create the spatially interpolated surface near oil field site 128 in 2004 compared to only 38 data points for the same area in 2009. The closest survey point in 2009 to the maximum reading in 2004 had the highest conductivity recorded near the oil pad (46 mS/m), but was ~ 12.5 meters away with a value comparable to the two nearest surveyed points in 2004 (45 and 50 mS/m, respectively). Therefore, the differences between Figures 22 and 23 show evidence of contaminant movement and the effects of different sampling locations.

Expanded surveys at oil field site 128 in 2009 documented elevated conductivities (maximum reading of 155 mS/m) in and around wetland 264 P and in the slough leading to well 264 R, with a maximum value of 82 mS/m (Fig. 23). The spatially interpolated surface shows only a thin strip of elevated conductivity in the slough, however, this is due to limitations of the equipment. The shallow exploration depth of the EM-31 (~6 m) can make it highly susceptible to changes in the depth to the water table that commonly develop in areas of undulating topographic relief (Reiten and Tischmak, 1993). During the 2009 geophysical surveys, the water table at well 264 R was approximately 2 meters below the land surface. Although the groundwater beneath the slough is contaminated as evidenced by water chemistry from well 264 R (Table 6), the relatively deep water table combined with the steep sided slough, allows the elevated conductivities to be measured only when the water table is within the exploration depth of the EM-31 (i.e. when measurements were taken in the bottom of the slough). The expansion of the elevated conductivities with the loss of topography from the slough near and eastward of well 264 R (Fig. 23) support the conclusion that groundwater beneath the slough and east of the county road is contaminated, yet elevated conductivities are only being measured when the water table in within the exploration depth of the EM-31.

EM-31 surveys were also conducted in the northern groundwater flow path and measured elevated conductivities associated with oil field sites 124 and 126 in 2009 (Fig.

23). A saline contaminated groundwater plume is clearly defined expanding southeastward from oil file site 124, with maximum conductivities in and around the wetland basins. Without samples of the groundwater chemistry near the peak conductivity (420 mS/m, west of wetland 124 F), it is difficult to determine if the extremely elevated readings here are due to the most contaminated groundwaters surveyed in this flow path or are due to differences in soil texture (clays and silts) and saturation in the wetland basins. However, the high chloride concentrations and CI values seen in discharge wetlands 124 D, 124 E, and124 F (Table 6) in the spring would require highly contaminated recharging waters, furthering the hypothesis that the elevated conductivities are due to extremely contaminated groundwater beneath and up-gradient of the wetlands. Elevated conductivities associated with saline contamination in the northern groundwater flow path were also observed east of the county road, near West Goose Lake. The maximum conductivities west of the road (220 – 240 mS/) were measured southeast of well 124 H which had the highest chloride concentration and CI value in this flow path in the spring (Table 6). Finally, the larger area of elevated conductivity eastward of wells 124 H and 124 P relative to 124 O (Fig. 23) corroborate groundwater results documenting a later arrival of highly contaminated groundwater at well 124 O.

The EM-31 survey results at oil field site 126 in 2009 support the conclusion that the increased chloride concentrations and CI values seen in wells 126 A and 126 C are due to the transport line break (Figs. 23 and 31). In 1989, an area of elevated conductivity was measured directly northwest of well 126 B associated with the buried reserve pit (Fig. 13). Apparent conductivities returned to background levels near the site access road. Elevated conductivity measurements define the saline groundwater plume associated with the reclaimed (buried) reserve pit in 2009, and an additional area directly north of well126 C, at the approximate location of the transport line break. The elevated conductivity values in 2009 defined a new plume that emanates from the transport line break, and had migrated south to wells 126 A and 126 C. These results illustrate the continued potential of co-produced water contamination to aquatic resources in the PPR from routine oil field operations.

EM-34 Surveys

An EM-34 was used to measure apparent conductivity in the study area and characterize the three dimensional geometry of saline contamination plumes and map stratigraphic controls on the transport of contaminated groundwaters in glacial outwash deposits. A brief overview of the principles of electromagnetic induction and EM-34 apparent conductivity measurements is necessary to understand the survey data presentation and results. The EM-34 uses a variable magnetic field (primary field) to induce electrical current flow and secondary magnetic fields in a hemispheric shaped sampled volume of subsurface material that are measured along with the primary magnetic field. At the low induction numbers used by the EM-34, the exploration depth is a function of intercoil spacing and orientation (McNeill, 1980b). Figure 33 shows the relative and cumulative response curves for the horizontal and vertical dipole orientations, illustrating two key points. First, the apparent conductivity readings are an integration of subsurface material beneath the instrument, with the exploration depth

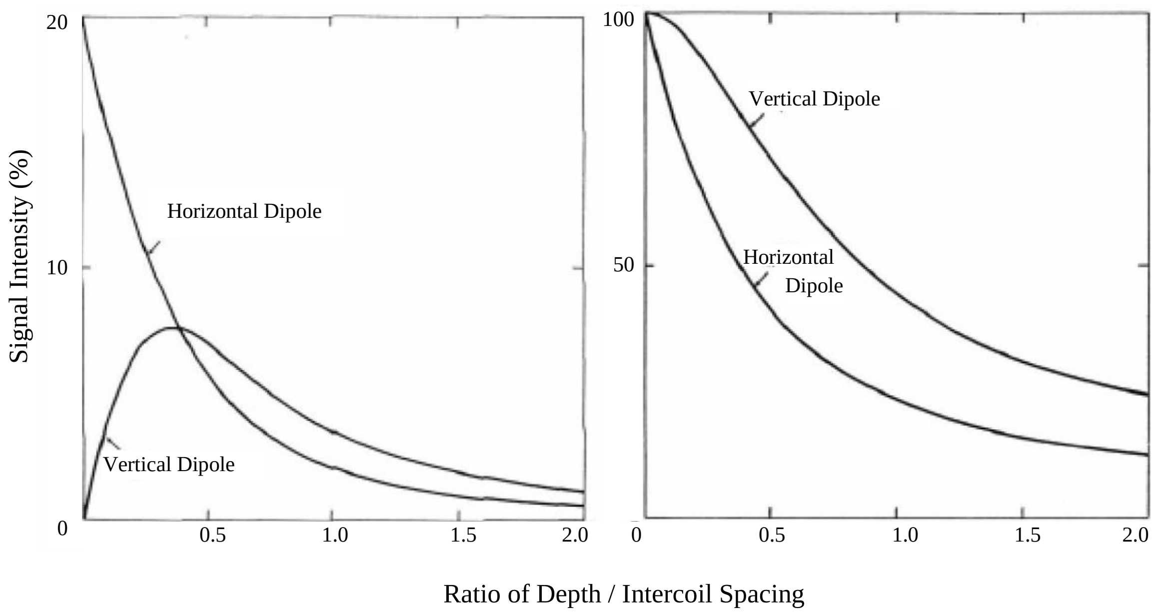

Figure 33. EM-34 relative response curves (left) and cumulative response curves (right) for horizontal and vertical dipole orientations (modified from Geonics TN-6).

being the depth at which 66% of the cumulative response is recorded. Second, the horizontal dipole mode receives the majority of the signal from immediately below the land surface compared to ~ 0.4 intercoil spacings in the vertical dipole mode. Therefore, despite the overlap in exploration depths between the two dipole orientations, the differences in electromagnetic induction results in the horizontal dipole mode receiving much more of its signal from shallower depths in the subsurface relative to the vertical dipole mode.

Furthermore, the interpretation of the EM-34 apparent conductivity data requires a hydrostratigraphic framework for control, as different geologic materials have different ranges of apparent conductivities (Fig. 34). Notice the relative positions of the glacial sediments, with conductivities generally decreasing from (lake) clays to tills to gravel and sand (outwash). The stratigraphic setting at the Goose Lake detailed study area is based

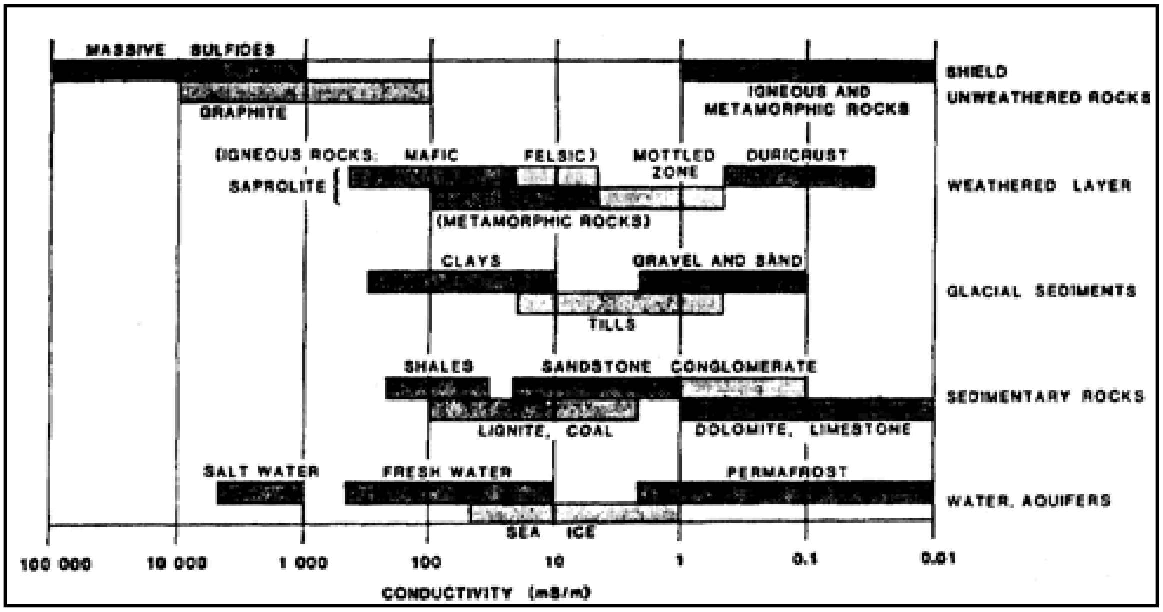

Figure 34. Electrical conductivity ranges for common geologic materials (from Sheriff, 1991).

on the 1989 hydrogeologic investigations (Reiten and Tischmak, 1993), and consists of saturated glacial outwash deposits mantling glacial till. Although glacial tills generally have higher conductivities than gravel and sand deposits, the presence of highly saline groundwater in either the outwash or glacial till results in much higher conductivities in these sediments relative to uncontaminated glacial till. Since the EM-34 data is an integration of conductivities from a volume of subsurface material, increases in the pore water conductivity (increasing TDS or contamination) or the presence of lacustrine (lake clay) deposits will increase conductivity readings. However, due to the difference in response curves, the vertical dipole is less sensitive to changes in near surface conductivities, with this influence decreasing with increased intercoil spacing (Fig. 33).

The EM-34 surveys were conducted following the approximate line of cross section D – D' in the southern groundwater flow path, roughly along cross section B – B' in the northern groundwater flow path, and following and on both sides of cross section

A – A' which crosses both the northern and southern flow paths (Figs. 11 and 12). The profiles were located to allow comparison with groundwater chemistry and an interpretation with a more detailed stratigraphic framework. From the groundwater well logs (GWIC, Fig. 12 and Table 2), the thickness of the outwash deposits in the southern groundwater flow path along D – D' increases down-gradient from 1.8 m at well 264 B to 6.1 m at well 264 Q and then decreases to 4.0 m at well 264 S. Continuing down-gradient, thickness increases to a maximum of 7.0 m at well 264 T before decreasing to 5.8 m at well 264 R. Additionally, a minimum of 1.5 m of lake clays are present below the outwash deposits at wells 264 S and 264 T and were mapped as a continuous unit in 1989. In the northern groundwater flow path along B – B', the thickness of the outwash deposits decrease from 5.2 m at well 117 J to 4.0 m at wells 124 K and 124 L, and finally to 3.7 m at well 124 M. The EM-34 surveys were continued south of well 124 M, up a large hill (~15 m elevation gain). A layer of clay-rich organic (slough) sediments was observed on the surface, with a thickness of 1.2 m at well 117 J and 0.9 m at wells 124 K and 124 L. Along A – A', the thickness of outwash deposits decrease from 9.1 m at well 124 G to a 4.9 m at well 124 H and then to 3.7 m at well 124 P, where 1.2 m of lake clay is present at a depth 0.9 m. Outwash thickness increases southward to 5.2 m at well 124 O and 5.8 at well 264 R, with a minimum of 1.8 m of lake clay existing beneath the outwash at well 124 O. With this refined stratigraphic framework, the EM-34 results (Figs. 24 and 25) strongly support the hypothesis that contaminated groundwater plumes have migrated laterally; mainly confined within the permeable outwash sediments mantling the glacial till. Additionally, the geophysical results illustrate that the down-gradient transport of contaminates is further controlled by the topography of the underlying till.

Southern Groundwater Flow Path The largest conductivity values (range: 230 – 260 mS/m) from all horizontal dipole intercoil spacings within the southern groundwater flow path were measured in the slough below the Hammer M tank battery (Figs. 24 and 35-A). Elevated conductivities were also recorded with all intercoil spacings throughout much of the flow path, with the differences between conductivity profiles largely explained by the subsurface geology. Conductivity profiles are very similar from the Hammer M tank battery to south of well 264 D, where the outwash thickness increases from 3.1 to 6.1 m between well 264 D and 264 Q. The concordance of conductivity values here is due to the thin outwash deposits, with a thickness much less than the smallest exploration depth, causing the cumulative response of the underlying till to be very similar for all intercoil spacings. As the thickness of the outwash sediments increases down the flow path, the influence of the underlying till on the cumulative response decreases with decreased intercoil spacings, producing the deviation observed in conductivity profiles down-gradient of well 264 Q.

The drop in horizontal conductivity measurements to between 40 – 58 mS/m in all intercoil spacings at well 264 Q (Figs. 24 and 35-A) is likely related to the influx of fresher water from the wetland and saturated outwash sediments to the north (Figs. 11 and 13), in addition to the increased thickness in outwash deposits. The reason for the greater increase in conductivity values in the measurements below well 264 Q in the 10 and 20 m relative to the 40 m intercoil spacing is unclear, but may be due to a thinning

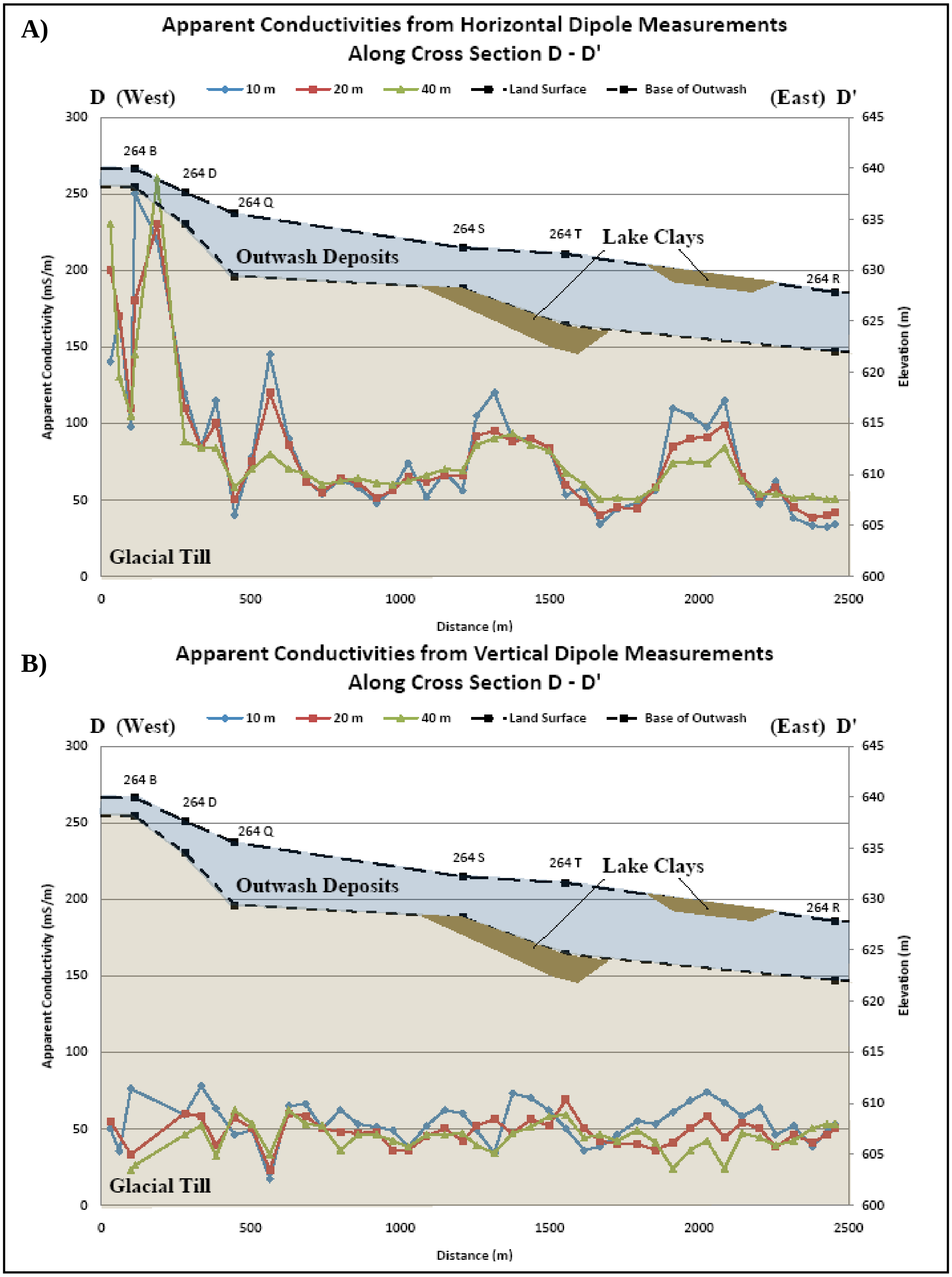

Figure 35. Apparent conductivity measurements from the EM-34 horizontal (A) and vertical (B) dipole surveys along cross section D – D' in the southern groundwater flow path.

of the outwash deposits, with more influence of the underlying till seen in the 40 m spacing. Below this, conductivity values dropped, but remained elevated and similar (51 – 74 mS/m) in all intercoil spacings until well 264 S, where conductivity values increased (Figs and 35-A). The increased conductivity readings can be explained by the presence of a minimum of 1.5 m of lake clay deposits underlying the outwash deposits between wells 264 S and 264 T, where outwash thickness also increases from 4.0 to 7.0 m. This interpretation is supported by the 10 m survey results, where the depth to the buried lacustrine deposits increases to near the exploration depth. The conductivities in the 10 m spacing decreased with increased outwash thickness, implying the observed values are being controlled by the decreasing influence of the relative response of the clay layer at depth as opposed to near surface conductivities.

Horizontal conductivity values decreased in all intercoil spacings down-gradient of well 264 T until wetland 264 P (surficial clay deposit in Fig. 35), where conductivities increased (Figs. 24 and 35-A). The visual disparity between the intercoil spacings immediately down-gradient of well 264 T in Figure 24 is more related to the map symbology than differences in conductivity readings, as several readings are at or near the 50 mS/m demarcation (Fig. 35-A). Conductivities were largest in the 40 m intercoil spacing at the five points below well 264 T reflecting the greater outwash thickness, but had similar values (range: 50 – 60 mS/m) as the 10 m (range: 34 – 58 mS/m) and 20 m (range: 40 – 58 mS/m) intercoil spacings. The increased conductivity values above wetland 264 P are due to the near surface clay deposits in the wetland as seen from a comparison of the intercoil spacing profiles (Figs. 24 and 35-A). Conductivity values were largest in the 10 m intercoil spacing for measurements taken above the wetland and decreased with increased intercoil spacing, as would be predicted if the high conductivity is due to the near surface sediments. Additionally, the highest measurement from all intercoil spacings was taken at the lowest point in the wetland where it would be expected to find the thickest wetland sediments. Finally, down-gradient of wetland 264 P, conductivities return to similar values (~50 mS/m) as seen on the up-gradient end in all intercoil spacings, further supporting the conclusion that the increased conductivity values above the wetland are due to the increased clay content of the near surface sediments (Figs. 24 and 35-A).

Immediately below wetland 264 P, conductivity values were similar in all intercoil spacings; however, conductivity values diverge in the slough down-gradient moving towards well 264 R (Figs. 24 and 35-A). With the exception of one point taken near a small NWI classified wetland, the conductivity values in the slough were constantly largest in the 40 m intercoil spacing and decreased with decreased intercoil spacing. Additionally, the 40 m results were much more consistent (range: 50 – 54 mS/m) than either the 10 m (range: 32 – 47 mS/m) or 20 m (range: 38 – 52 mS/m) intercoil spacing results, where conductivities generally decreased down-gradient. The greater and less variable conductivity readings observed in the 40 m relative the 10 and 20 m intercoil spacing is likely explained by the relatively thick outwash deposits and deeper water table at well 264 R (5.8 m and 2.0 m, respectively). East of the county road, conductivity values were similar to those recorded near well 264 R implying that the groundwater chemistry here is similar to that at well 264 R (Figs. 24 and 35-A). Finally, the increased conductivities near West Goose Lake (almost due east from well 264 R) are likely due to the increased clay content of the lacustrine sediments and presence of sodium sulfate salts; however the elevated conductivities seen up-gradient suggest that saline contaminated groundwater from the southern flow path is being discharged into the lake (Fig. 24).

In contrast to the horizontal dipole orientation, measurements in the vertical dipole orientation are primarily responsive to conductivity changes with depth (i.e. depth of contamination, Fig. 33). The vertical dipole surveys show the underlying till is acting as an aquitard allowing minimal penetration of saline groundwater. This is especially evident in the slough immediately below the Hammer M tank battery and the county road, despite only one of the five survey points in the slough successfully yielding a complete suite of vertical dipole measurements (Figs. 25 and 35-B). Three measurements were obtained from the 10 m intercoil spacing and two measurements in the 20 m and 40 m spacings. The survey locations that successfully obtained 40 m intercoil spacing measurements were very close together and were directly beneath the Hammer M tank battery. However, one was located on a bench above the slough, with the other in the slough, at 264 B which provides excellent subsurface control. Conductivity measurements were comparable for the two 40 m spacings, exhibiting a small decrease above the slough (26 and 23 mS/m, respectively), and are representative of uncontaminated glacial till (Fig. 34).

The survey location above the slough was the only point to record all vertical dipole measurements, with the 20 m conductivity of 33 mS/m and a 10 m conductivity of

76 mS/m (Fig. 35-B). Horizontal dipole survey results from this location, where the outwash thickness is slightly greater, were similar (98 – 110 mS/m); however at well 264 B, the thinnest documented outwash thickness (1.8 m), horizontal dipole conductivities varied considerably (Fig. 35-A). Here, the 10 m spacing yielded the largest value (250 mS/m) observed in the southern groundwater flow path in this setting, with conductivity decreasing to 180 mS/m and 145 mS/m in the 20 m and 40 m spacings, respectively. The elevated horizontal dipole conductivities above the slough and highly elevated, and rapidly decreasing horizontal conductivity values at well 264 B attest to the high degree of chloride contamination in the near surface; however the return of conductivities to background values in the 40 m vertical dipole survey, clearly show that very little of the contamination is present at depth. Therefore, it is evident that the saline contamination at the Goose Lake detailed study area is mainly confined to the saturated outwash deposits, with the underlying till acting as an aquitard, allowing little infiltration of saline groundwater. The similarity of the 20 m and 40 m vertical dipole measurements above the slough, where a 30 m increase in the exploration depth produced only a 10 mS/m reduction in apparent conductivity, and that the apparent conductivities at the 40 m spacing are background level suggest that depth of contamination is less than 30 m and clearly less than 60 m (Fig. 35-B).

East of the county road between wells 264 B and 264 D, the vertical dipole surveys continue to support minimal penetration of saline groundwater into the underlying till (Figs. 25 and 35-B). However, the increase in outwash thickness down-gradient from well 264 B to well 264 Q reduces the sharp contrast seen up-gradient. The reduction in the area of elevated conductivity with increased intercoil spacing seen near well 264 D, and the emergence of elevated conductivities east of well 264 Q, illustrates this increase in outwash thickness, with elevated conductivities only seen with the 40 m spacing in the thicker outwash deposits. The presence of a subsurface glacial till high, approximately three measurements southeast of well 264 Q, postulated from the horizontal dipole surveys (conductivity range: 80 – 145 mS/m, Fig. 35-A), is supported by the vertical dipole surveys, where conductivities were near background for all intercoil spacings (conductivity range: 17 – 33 mS/m, Fig. 35-B). South of the subsurface glacial till high to well RAB 2, elevated conductivities (> 50 mS/m) were documented in all vertical dipole intercoil spacings, with the conductivity values decreasing with increased intercoil spacing, furthering the interpretation of minimal saline contamination at depth (Figs. 25 and 35-B).

As in the horizontal dipole surveys, elevated conductivities (> 50 mS/m) were measured near wells 264 S and 264 T, and wetland 264 P (surficial clay deposit in Fig. 35) in the vertical dipole surveys (Figs. 25 and 35-B). Differences between the vertical dipole intercoil spacings survey results across these conductivity anomalies support the assumption that conductivity readings here are being influenced by the presence and stratigraphic position of lacustrine clays. A buried lacustrine deposit is present underneath wells 264 S and 264 T and is expressed by the persistence of elevated vertical conductivities in all intercoil spacings. In contrast, vertical conductivities at wetland 264 P are the influenced by clay-rich surficial sediments, expressed primarily in the 10 m survey, as elevated conductivities across the entire wetland and into the slough leading to well 264 R. The influence of the wetland sediments is greatly reduced in the 20 m spacing, as seen by the large reduction in the area of elevated conductivities (Fig. 25). In the 40 m spacing, changes in the near surface conductivity have little effect on the reading, evidenced by the lack of elevated conductivities and two measurements representative of uncontaminated till (both 24 mS/m) from the wetland basin. Additionally, the influence of wetland sediment is likely the cause of elevated vertical conductivities west of well 264 S documented in the 10 m intercoil spacing and not seen in 20 m and 40 m spacings, as two readings were taken within a NWI classified wetland (Fig. 35-B).

In the slough east of wetland 264 P, the horizontal and vertical dipole conductivity values were in closer agreement and remained generally above background levels (Figs.24, 25, and 35). Horizontal dipole conductivities ranged from 32 – 62 mS/m in the slough compared to 38 – 64 mS/m in the vertical dipole mode, with the lower values and concordance among dipole orientations due to the relatively small saline contamination of the groundwater and the outwash thickness, respectively (732 mg/L chloride and 5.8 m at well 264 R). Directly across the county road, vertical dipole conductivity values were similar to those observed in the slough, and show the same increasing trend towards West Goose Lake as observed in the horizontal surveys (Figs. 24 and 25). However, vertical dipole conductivity values were less variable in the 40 m intercoil spacing compared to the 10 m and 20 m spacing, illustrating the same relative response of the underlying till.

Northern Groundwater Flow Path EM-34 surveys in the northern groundwater flow path also documented larger conductivities in the horizontal dipole orientations relative to the vertical dipole orientations (Figs. 24, 25, and 36) and further support stratigraphical control on the transport of saline contaminated groundwater at the West Goose Lake detailed study area. The effect of topography on apparent conductivity measurements is seen in the horizontal and vertical dipole orientations in the expanded survey along transect B – B' (location shown in Fig. 11), where a steep hill results in a 15 m elevation gain south of wetland 124 F (Fig. 36). Notice the consistency in conductivities between the 20 m and 40 m vertical dipole and 40 m horizontal dipole results, the three greatest exploration depths, despite the elevation gain. In contrast, the three measurements from 10 m vertical and 10 m and 20 m horizontal dipoles all exhibit decreased conductivities with increased elevation, reflecting the increased depth or disappearance of the water table in the outwash deposits on the hill.

At the bottom of the hill, the outwash deposits are saturated and conductivity values increase in all dipole orientations and intercoil spacings due to the presence of saline groundwater contamination (Figs. 24, 25, and 36). The largest conductivities along the B – B' transect in all vertical and horizontal dipole orientations were recorded at the two survey locations at the bottom of the hill; however, conductivity profiles were very different for the two survey locations. At the base of the hill, conductivities in the horizontal dipole surveys were elevated but similar, while the vertical dipole surveys showed a marked increase with increased intercoil spacing. One measurement point north, the elevated conductivities showed minor decreases from increased intercoil

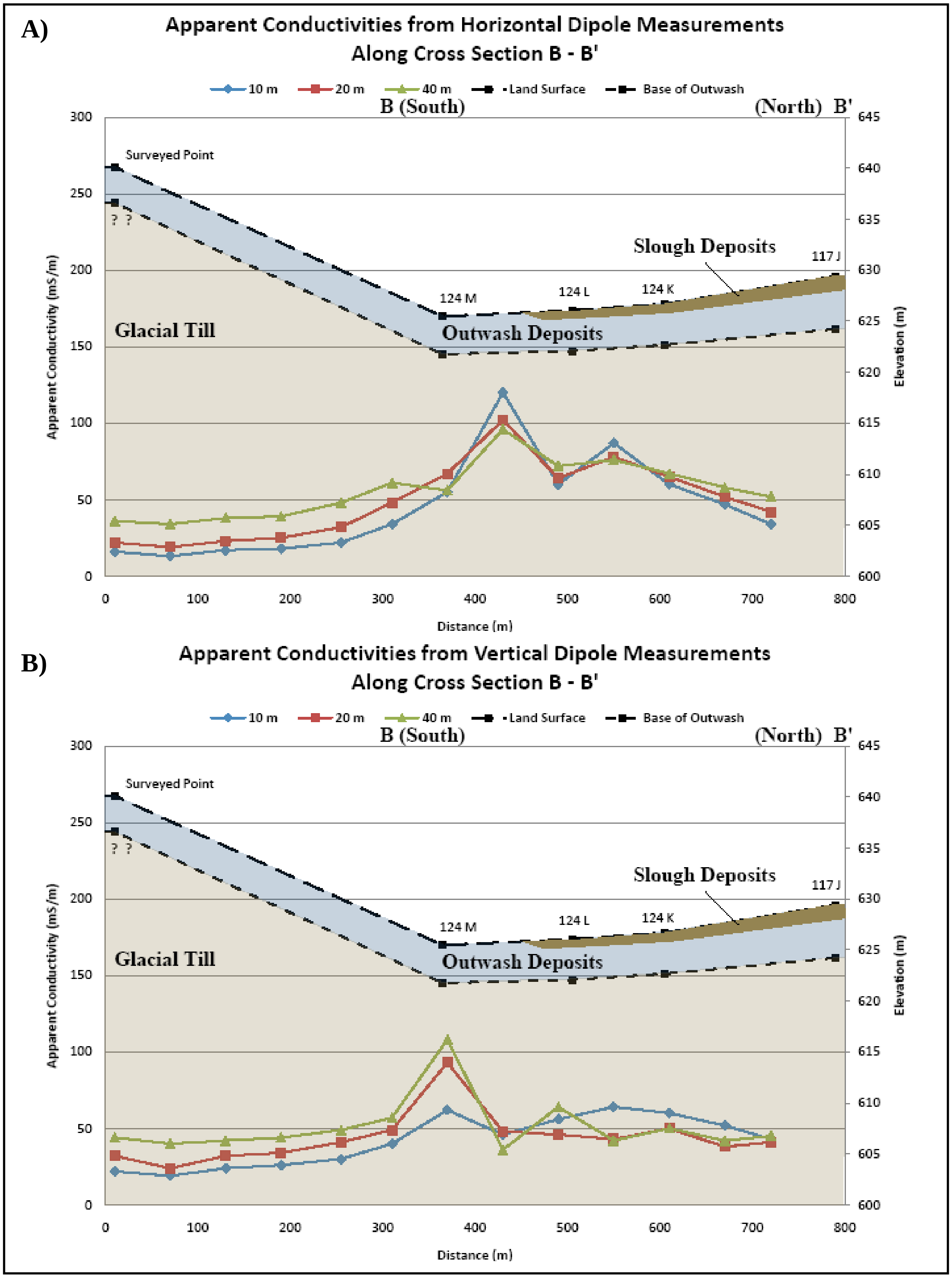

Figure 36. Apparent conductivity measurements from the EM-34 horizontal (A) and vertical (B) dipole surveys along cross section B – B' in the southern groundwater flow path.

spacing in the horizontal dipole surveys and much lower conductivities in the vertical dipole surveys (Fig. 36).

Greatly elevated conductivities in all EM-34 surveys at these two points attest to the high degree of contamination (Fig. 36), with the location at the base of the hill recording the largest 40 m vertical dipole conductivity and potentially representing the greatest depth of saline groundwater contamination observed in the EM-34 surveys. The comparable conductivity of the 10 m vertical dipole to the horizontal dipole surveys, the larger conductivity in the 20 m vertical dipole mode compared to the 40 m horizontal dipole mode, and similar values for the 20 m and 40 m vertical dipole surveys, suggest that the elevated conductivity source is between 15 and 30 meters below the land surface (the exploration depths of 10 m vertical and the 20 m and 40 m vertical and horizontal dipole orientations, respectively). After the outwash thickness at the nearest well is removed, penetration into the till is approximately 11 – 26 meters, which compares well to the value seen below the Hammer M tank battery (< 30 m).

Additionally, the horizontal dipole surveys record elevated conductivities (> 75 mS/m) northward past wells 124 J and 124 C (Figs. 24 and 36-A). After a second, but smaller peak in conductivities south of wells 124 J and 124 C (87, 78, and 76 mS/m at the 10, 20 and 40 m spacings, respectively) the final four horizontal dipole measurements steadily declined north towards well 117 J in all intercoil spacings. The slight expansion in elevated conductivities (> 50 mS/m) with greater intercoil spacing north of wells 124 J and 124 C is likely due to the increase in outwash thickness between these wells and well 124 J. Furthermore, the northward reduction in conductivities supports the water chemistry results that the groundwater entering the northern flow path from oil field site 117 is less contaminated than the groundwater down-gradient of oil field site 124. Vertical dipole conductivities were more variable with only the shallowest survey, the 10 m intercoil spacing, showing a northward reduction in the last four conductivity measurements. In the 20 and 40 m spacings, vertical dipole measurements were similar, with no obvious trend (Figs. 25 and 36-B). Interestingly, the 20 and 40 m vertical conductivity measurements are very comparable to the 40 m vertical dipole survey measurements taken across the hill to the south, in an area that should be uncontaminated, and therefore maybe more representative of the background conductivity in the underlying glacial till.

The EM-34 surveys along and surrounding cross section A – A' document much larger rates of lateral, relative to vertical, transport of saline groundwater contamination (Figs. 11, 24, 25, and 37). This is seen in the north-south expansion of elevated conductivities (> 75 mS/m) compared to cross section B – B' (Fig. 36) in the horizontal dipole surveys (Figs. 24 and 37-A), and the more consistent and generally lower vertical dipole conductivities (Figs. 25 and 37-B). Additionally, two peaks in conductivities were documented along B – B' (west of wetland 124 F and just south of wells 124 J and 124 C, Figs. 24 and 36), and defined the beginnings of two preferential groundwater flow paths that likely developed in response to a subsurface glacial till high beneath well 124 P surrounded by deeper outwash deposits at wells 124 H to the north and 124 O to the south (Figs. 12 and 37). The two flow paths are clearly expressed as elevated conductivities in the horizontal dipole surveys north and south of well 124 P (Fig. 24) and

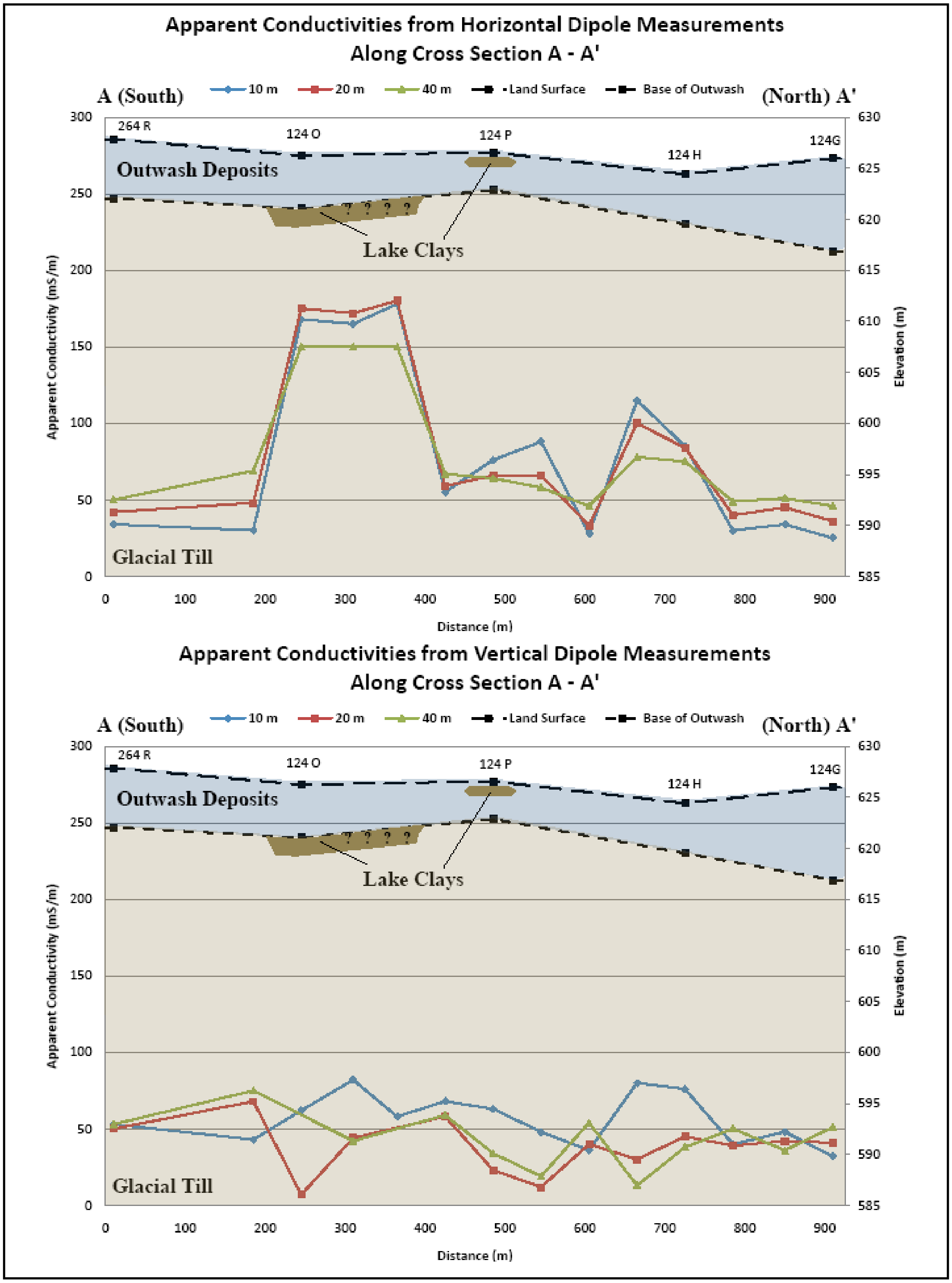

Figure 37. Apparent conductivity measurements from the EM-34 horizontal (A) and vertical (B) dipole surveys along cross section A – A' in the southern groundwater flow path.

are in agreement with 2009 groundwater chemistry results, were wells 124 H and 124 O had the highest chloride concentrations and CI values in the northern flow path.

The largest horizontal dipole measurements from all intercoil spacings (280, 250, and 160 mS/m in the 10, 20, and 40 m spacings, respectively) in the northern groundwater flow path were recorded west of well 124 P, in the area of elevated conductivities leading to well 124 O (Fig. 24). Unfortunately, no vertical dipole measurements were obtained at this point; however, the next point south had similar horizontal conductivities (262, 208, and 130 mS/m, respectively, Fig. 24) and recorded the 20 m vertical dipole measurement (33 mS/m, Fig. 25). The glacial till high near well 124 P likely channels this highly contaminated groundwater southwest, where elevated conductivities (≥ 150 mS/m) were observed in the three measurement points at and north of well 124 O in all horizontal dipole surveys (Figs. 24 and 37-A). A full suite of vertical dipole measurements was only obtained from the middle survey point and documented a sharp decrease in conductivity between in the 10 m (83 mS/m) and 20 m and 40 m (44 and 42 mS/m, respectively) intercoil spacings (Figs. 25 and 37-B). The large difference in conductivities between the horizontal and vertical dipole surveys (Fig. 37) and the rapid decrease in conductivity to comparable values in the deeper vertical dipole surveys (Fig. 37-B) illustrate the shallow depth of saline contamination.

Well 124 O experienced the largest increase in chloride concentration (14,626 mg/L) between 2006 and 2009, with the later arrival of highly contaminated groundwater evident in the EM-34 surveys. Conductivity measurements dropped drastically across the county road east of well 124 O, where only one survey location documented conductivities greater than 75 mS/m in the 10 m and 40 m horizontal dipole surveys (Fig. 24). Here, vertical dipole measurements were similar to those observed near well 124 O, with the 40 m spacing yielding 35 mS/m (Fig. 25). The rapid decrease in horizontal conductivity values implies that the highly saline groundwater evident from the 2009 water chemistry data, has yet to migrate much past well 124 O, while the low 40 m vertical dipole measurements confirms that the contamination has not migrated vertically. Although a new pulse of highly saline groundwater arrived at well 124 O after 2006, the chloride concentration had decreased from 1989 to 2006 (7,840 and 2,574 mg/L, respectively). The lateral migration of this contaminated groundwater is seen in the 10 m and 20 m vertical and all horizontal dipole surveys as an expanding strip of elevated conductivities (> 75 mS/m) trending northeast from well 124 O, with conductivities generally increasing towards West Goose Lake. The difference in the 20 m and 40 m vertical dipole surveys may be due to the presence of buried lake clays as documented at well 124 O; however all readings were less than 60 mS/m. Increased conductivities in the horizontal dipole surveys near West Goose Lake may be due to higher contamination in the groundwater (i.e. 1989 vs. 2006 water chemistry at well 124 O) or influences from the preferential groundwater flow path north of the glacial till rise near well 124 P.

The horizontal dipole measurements in the preferential flow path north of well 124 P (Fig. 24) documented lower conductivities than in the southern preferential flow path, with the largest conductivities recorded west of well 124 H (200, 150, and 102 mS/m in the 10, 20, and 40 m spacings, respectively). Horizontal conductivity measurements decreased in all surveys at well 124 H and were significantly lower than conductivities observed at well 124 O (Fig. 37-A), where chloride concentrations were slightly less in the spring of 2009. The reason for this disparity is unclear, but may be due to the presence of lake clays beneath well 124 O and not well 124 H. Across the county road towards West Goose Lake, conductivities greater than 100 mS/m were recorded in all horizontal dipole surveys (Fig. 24). Elevated conductivities (> 75 mS/m) were also documented along the western lake shore, and appear to show a convergence of the two preferential flow paths. Conductivities decreased to similar values (~ 40 mS/m) for the four northernmost measurement locations in the horizontal dipole orientations for all survey lines surrounding cross section C – C', and likely define the approximate northern edge of the saline groundwater plume in the northern flow path.

The vertical dipole measurements in the preferential flow path north of well 124 P exhibited a sharp contrast with depth, where elevated conductivities (> 100 mS/m) were only documented in the 10 m survey (Fig. 25). Conductivities in the 10 m vertical orientation followed a similar trend as the horizontal surveys, with decreased conductivities near well 124 H and increased values across the county road. In the 40 m spacing, conductivities were much lower with the exception of one survey location northeast of wetland 124 E (92 mS/m); however the 40 m measurement was much larger than the 10 m and 20 m measurements (40 and 56 mS/m, respectively) and is probably reflecting increased conductivity in the deeper stratigraphy and not groundwater contamination. Despite the elevated conductivities in the preferential flow path observed in the 10 m vertical and all horizontal dipole surveys, conductivities in the 20 m and 40 m vertical orientations were some of the lowest documented (4, 13, 22 and 23 mS/m) and furthering the hypothesis that limited penetration of saline contamination into the underlying till has occurred. Finally, the vertical conductivity measurements in the four northernmost survey points in all survey lines (~40 mS/m) were very similar to the horizontal dipole surveys, which further supports the interpretation that the drop in conductivities seen in the horizontal dipole surveys is delineating the approximate edge of the saline ground plume in the northern flow path.

EM-31 and EM-34 Comparison The maximum and background conductivities documented by the EM-31 (445 and 28 mS/m, respectively, Fig. 23) were much larger than the EM-34 10 m horizontal dipole survey (280 and 16 mS/m, respectively, Figs. 24 and 35-A). Additionally, separate survey locations were used to calculate the average background conductivity measured by each instrument, with the EM-34 measurements taken above unsaturated outwash deposits; likely resulting in a low background conductivity estimate. However, much of the difference observed between areas of overlap in the normalized conductivity profiles (ratio of apparent conductivity measurement to background conductivity) is due to the increased sampling density of the EM-31 relative to the EM-34 (Figs 23 and 24). This can be clearly seen in the southern groundwater flow path, where only one EM-34 measurement was taken down the flow path, compared to several across the slough with the EM-31. Additionally, the EM-31 was operated in the vertical dipole orientation, and therefore receives more of the signal from greater depths than the EM-34. Despite the potentially low EM-34 background conductivity and differences in dipole orientation, the maximum normalized conductivities for the EM-31 and EM-34 were similar (15.9 and 17.5, respectively), illustrating the effectiveness of normalizing the data to compare results from different types of equipment. When combined (and slightly cropped), the normalized apparent conductivity maps (Fig. 38) cover the majority of the southern and northern groundwater flow paths and illustrate the widespread saline contamination within the Goose Lake detailed study area.

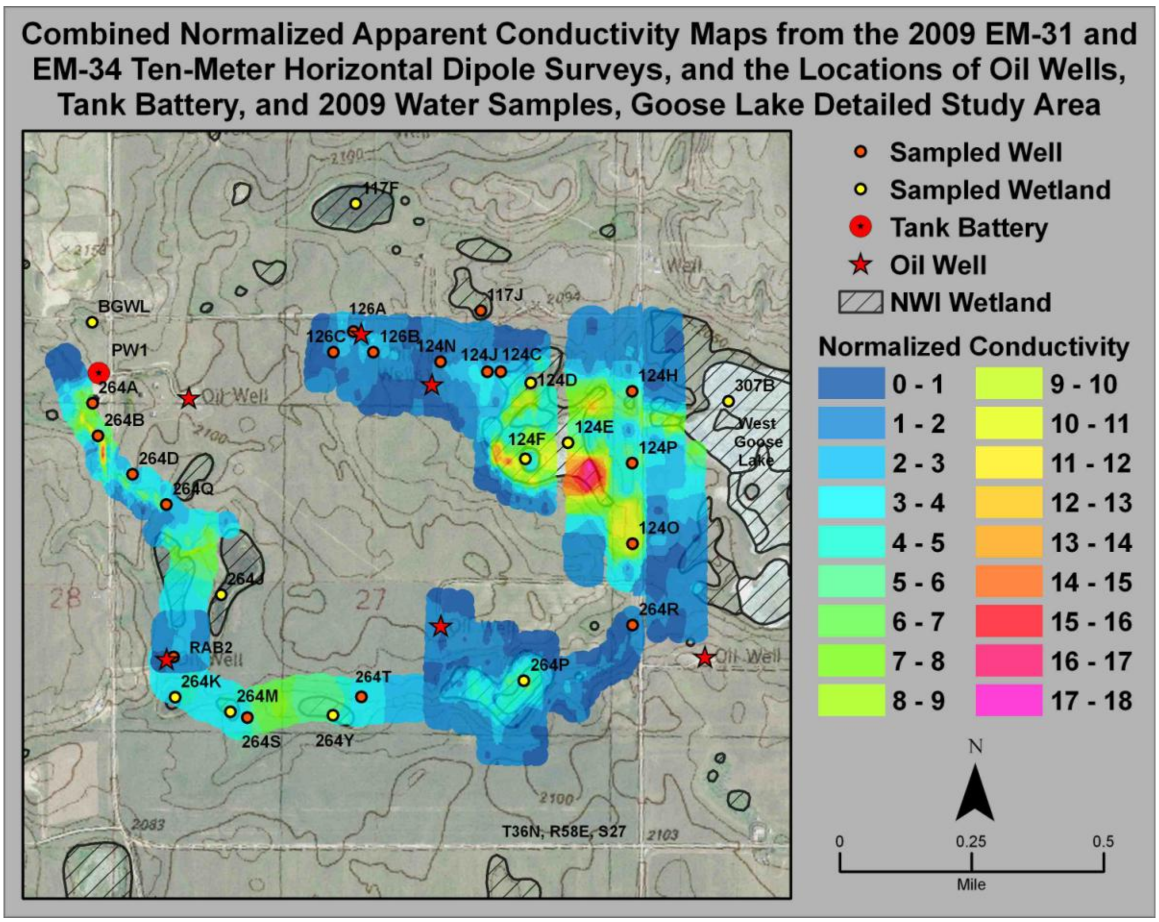

Figure 38. Combined normalized conductivity maps from the EM-31 and EM-34 10m horizontal dipole orientation surveys. EM-31 results were used in areas of overlap due to closer spacing and larger number of measurement locations.

<u>Lateral and Vertical Transport Rates</u> The geophysical data clearly showed a strong reduction in conductivities with depth, implying that minimal penetration of saline

contaminated groundwater into the underlying till has occurred. A simple comparison of measured EM-34 conductivities illustrates this conclusion, with much larger average and maximum conductivities and lower standard deviations in the horizontal dipole, relative to vertical dipole, orientations (Table 9). Estimates of the maximum depth of saline penetration were comparable for both groundwater flow paths; approximately 15 – 30 m. In contrast, widespread saline contamination was observed far down-gradient from likely contaminant sources in both flow paths, showing the much larger rate of lateral transport. Based on the EM-31 and EM-34 surveys, the majority of the contamination in the northern flow path is originating from oil field site 124 which was installed in 1967 (Reiten and Tischmak, 1993). Contamination has migrated laterally, past wells 124 H and 124 O, and into West Goose Lake; a distance of 765 and 1,150 meters, respectively. The contamination in the southern groundwater flow path associated with Hammer M tank battery was documented as far down-gradient as well 264 T, and possibly well 264 R (Fig. 20), where contamination could also be due to leachates generated from the buried reserve pit at oil field site 128. Measured down the flow path, the distance to these wells is approximately 1,625 and 2,485 meters, respectively.

Table 9. Summary statistics of all EM-34 apparent conductivity measurements in the Goose Lake detailed study area, with all values in mS/m.

	Vertical Dipole Orientation			Horizontal Dipole Orientation		
	10 m	**20 m**	**40 m**	**10 m**	**20 m**	**40 m**
Minimum	17	7	4	13	19	34
Maximum	102	93	108	280	250	260
Average	53	46	47	73	75	74
Std Dev	17	13	13	52	44	34

Chloride concentrations at CI values of wells 124 H (36,500 mg/L and 0.) and 264 R (195 mg/L) in 1989 indicate that the saline contamination observed in West Goose Lake (4,480 mg/L) in 1989 was mainly derived from the northern groundwater flow path, requiring a minimum lateral transport distance of 765 meters in 22 years (~35 m/yr). In the southern groundwater flow path, the leading edge of the saline contamination plume from the tank battery was likely between wells 264 T and 264 R in 2009. The initial operational date of the tank battery is not published; however, installation likely occurred in 1967 when several of the nearby oil wells were drilled. A similar minimum lateral transport rate is seen when well 264 T and 1967 as the date of initial contamination are employed (1625/42 = ~39 m/yr). Using 1967 and a 30 m exploration depth, the vertical transport rate is ~ 0.7 m/yr. Therefore, it appears that the lateral rate of contaminant transport is at least 50 times greater than the vertical transport rate.

Additionally, the stratigraphic position of the glacial till beneath the saturated outwash deposits would be expected to keep the till saturated and would minimize weathering and reduce the permeability of fractures from the swelling of smectitic clays. This interpretation is confirmed from the groundwater well logs (Table 2) where only one well that was drilled though the outwash deposits encountered oxidized till, with the remainder of wells hitting lake clays or unoxidized and transitional tills beneath the outwash deposits. Therefore the till beneath the saturated outwash deposits likely has similar hydrological properties as the unweathered till in the PPR. The large reduction in Na and Cl ions (~85%) reported by Murphy et al. (1998) from the weathered to unweathered till beneath reserve pits in North Dakota illustrates that although co-produced water contamination can migrate downward in the unweathered till, the majority of contamination remains above this contact. Without groundwater chemistry from the underlying till, the magnitude of contamination cannot be assessed; however it would seem highly probable that similar reductions in ion concentrations would exist across the boundary between the saturated outwash deposits and the underlying till. Therefore, the large difference between the vertical and lateral contaminant transport rates and likely minimal contaminant storage in the underlying till emphasizes the potential for co-produced water contamination to aquatic resources far down-gradient of oil field sites in glacial outwash deposits in the PPR.

CONCLUSIONS AND FUTURE WORK

Hydrogeologic Investigations

Results from hydrogeological investigations in 1989 and 2009 were used to evaluate the first hypothesis that saline groundwater plumes have continued to migrate down gradient towards West Goose Lake, increasing the lateral extent of co-produced water contamination. Despite the expected increase in the extent of contamination, it was hypothesized that the natural attenuation of contaminated surface and groundwater resources will have resulted in the reduction in CI values in sample locations identified as contaminated in 1989.

Hydraulic Conductivity

Rising head slug tests were performed on six groundwater wells (124 H, 124 J, 126 C, 264 B, 264 R, and 264 S) in 2009 to measure K values within the surficial sand and gravel outwash deposits within the Goose Lake detailed study area. Results from the rising head slug tests (range: 0.12 – 13.50 ft/d and a geometric mean of 2.06 ft/d, Table 4 and Appendix A) were drastically lower than a 200-minute aquifer test (100 ft/day) performed at oil field site 124 in 1989 (Reiten and Tischmak, 1993). The discordance in K values from 1989 and 2009 are likely due to differences in test design and the amount of aquifer tested. The variability of K values from the rising head slug tests showed no relationship with the ratio of outwash to glacial till/lake clay in the screened interval, and therefore may reflect differences in sorting and texture within the outwash deposits.

Additionally, the pump test results illustrate the difficulty in applying a single K value to even a small area of glacial outwash deposits.

Hydraulic Gradient

Groundwater levels in the wells listed above were then monitored for eleven months to determine hydraulic gradient, with data from two wells recording incomplete records. Groundwater elevation changes were small (< 2.5 ft) and the relative elevations between wells remained consistent, signifying no seasonal changes in the local groundwater flow directions (Fig. 16-A). The monitoring of only six wells precluded the ability to investigate local scale changes in groundwater flow directions from surface and groundwater interactions near wetlands; however, these fluctuations would have little effect on the general flow regime. Groundwater flow directions determined in 2009 were equivalent to results from 1989, and defined two major flow paths that converge near West Goose Lake.

Despite the consistency of groundwater elevations between wells, individual wells showed marked differences in hydraulic response to an intense precipitation event and seasonal recharge. The water table is configured differently in the two flow paths, with the depth to the water table increasing down-gradient in the southern flow path and decreasing down-gradient in the northern flow path. Recharge from the infiltration of meteoric water during an August rain storm was only observed in the wells in or down-gradient of areas with a relatively shallow water table (124 H, 264 A, and 264 S, Fig. 16-B). Additionally, water level rises from seasonal recharge were less in wells with deeper water tables. Recharge events can likely remobilize salts that have evaporated in the capillary fringe and may produce pulses of enhanced contamination. The hydrograph results demonstrate the potential for meteoric recharge and subsequent salt remobilization from intense precipitation events and spring recharge increases as the depth to the water table decreases. At the Goose Lake detailed study area, this translates into more frequent and greater fluctuations in the water table depth near the Hammer M tank battery than oil field sites 124 and 126.

Surface and Groundwater Chemistry

Eleven surface and seventeen groundwater samples were collected and analyzed for major ion concentrations in 2009 with twenty six of the locations, ten surface and sixteen groundwater wells, having been sampled in 1989. Additionally, twelve of the groundwater wells were sampled in 2005/2006. Water samples from groundwater wells 126 C and 264 T in 1989 and wetland BGWL in 2009 had very low, and comparable chloride concentrations and CI values, with these values considered representative of uncontaminated, or reference waters, in the area. In 1989, all surface and fourteen groundwater samples had elevated chloride concentrations and specific conductance values that yielded CI values documenting widespread co-produced water contamination. The two other wells were located up-gradient of saline contamination in the northern flow path (126 C), and down-gradient of saline contamination in the southern flow path (264 T), and were not sampled in 2005/2006. All twenty six water samples from sites sampled in 1989, and two additional sites, had CI values indicating co-produced water contamination in 2009, including wells 126 C and 264 T. Therefore, the continued presence of saline contamination at all previously contaminated sites, and CI values in

2009 indicating contamination at sites considered uncontaminated in 1989, confirms the hypothesis that the lateral extent of co-produced water contamination has increased at the Goose Lake detailed study area (Fig. 20).

Increases in the chloride concentration and CI value at well 264 T were expected as this well is located a short distance down-gradient from the furthest well contaminated by the plume emanating from the Hammer M tank battery 1989 (well 264 S). The increased chloride concentration and CI value at well 264 T validates the first hypothesis that the increased lateral extent of contamination is due to the down-gradient transport of contaminants. Down-gradient transport is further illustrated by similar increases in the most down-gradient wells in the northern and southern flow paths, wells 124 O and 264 R, respectively. Although down-gradient increases in chloride concentrations and CI values were observed in groundwater samples, the chloride concentration and CI value decreased at the hydraulically lowest sample location, West Goose Lake. The water chemistry of West Goose Lake is influenced by large lakes to the east and evaporated salts can be removed through aeolian deflation; consequently, the reduction in chloride concentration and CI value are not considered as evidence against the hypothesis of down-gradient transport of contaminants. Due to the hydraulic position of well 126 C, up-gradient of a known saline contaminated groundwater plume documented in 1989, the increase in chloride concentration and CI were unexpected. However, the increases are almost certainly caused by a transport line break a short distance hydraulically up-gradient of the well (Fig. 31). Therefore, co-produced water contamination at well 126 C also documents down-gradient contaminant transport and highly emphasizes the potential for co-produced water contamination to aquatic resources within the broader PPR from routine oil field operations. Furthermore, with the predicted expansion of oil and gas development in the Bakken Formation, the potential for contamination from transport line breaks will likely increase as more wells become operational, requiring a larger transport line network.

Although extensive saline contamination was observed in both the southern and northern groundwater flow paths, distinct differences were observed between the CI profiles from the two flow paths. In the southern flow path, the chloride concentrations and CI values in groundwater samples steadily decreased down-gradient in 1989, 2005, and 2009 from respective maximum values measured at well 264 A located immediately below the main contaminant source (the Hammer M tank battery, Fig. 27-A). The CI values from surface water samples were more variable in 1989 and 2009 and often markedly different than nearby groundwater wells; however the general trend of decreasing CI value with increased distance from the tank battery is still observed (Fig. 27-B). In contrast, CI values from contaminated groundwater samples in the northern flow path were very similar in 1989, 2006, and 2009, showed no decrease with distance from contaminant source, and lower CI values did not necessarily correspond to lower chloride concentrations. In fact, the greatest chloride concentrations in 1989, 2006, and 2009, (wells 124 H, 124 P, and 124 H, respectively, Fig. 18) and CI values in 2006 and 2009 (124 P and 124 H, Fig. 30) were reported in wells several hundred meters from the oil field sites, near the lower end of the flow path. Finally, surface water samples in the northern flow path had very similar CI values and were comparable to nearby groundwater wells.

The discordance in CI profiles between the two flow paths are likely due to the differences in topography, water table configuration, and the hydrologic position of sampled wetlands. The topography near the Hammer M tank battery channels runoff from the spring thaw and precipitation events into the slough at the top of the southern flow path where the water table is very shallow. In contrast, oil field sites 124 and 126 in the northern flow path, lack steep topography capable of focusing runoff water near the main contaminant source and the water table is much deeper. This likely promotes more frequent and greater rises in the water table near the contaminant source in the southern flow path producing pulses of enhanced saline contamination. Runoff water channeled into the slough further down-gradient in the southern flow path can easily infiltrate into the shallow groundwater system or flow overland to lower wetlands, diluting the saline contamination as it moves down-gradient. The deeper water table in the northern flow path near oil field sites 124 and 126 likely minimizes the number of potential recharge events and the magnitude of groundwater elevation changes, resulting in a more constant release of saline contamination. Additionally, the lack of topography to focus potential recharging meteoric waters keeps the groundwater from being significantly diluted, resulting in the relatively consistent chloride concentrations and CI values. Finally, the sampled wetlands in the northern flow path are discharge wetlands and receive a strong hydrological input from the shallow groundwater system, explaining the similarity in CI values of the wetlands and nearby groundwater wells in 1989 and 2009. In the southern flow path, sampled wetlands are mainly recharge and flow-though wetlands, with water levels more dependent on precipitation and runoff. The weaker hydrological influence of the shallow groundwater system on the wetland chemistry in the southern flow path and highly variable nature of precipitation in the PPR explains the discrepancy of CI values between the sampled wetlands and nearby groundwater wells in 1989 and 2009.

Despite the widespread contamination still present at the Goose Lake detailed study area in 2009, CI values decreased in sixteen of the twenty one sample locations used to determine if a significant reduction in the CI value at contaminated sites had occurred since 1989. The observed median twenty year reduction in CI values was significant using the Wilcoxon signed rank test, confirming the hypothesis that the natural attenuation of surface and groundwater resources has resulted in a decrease in CI values for sites considered contaminated in 1989. Well 264 A located in the southern flow path, and well 124 H located in the northern flow path, had the two largest chloride concentrations and CI values measured in 2009. Assuming reductions in CI values proceed at the same linear rate, contamination would still be evident at well 264 A for 138 years, and at well 124 H for 125 years. The preceding estimates due not take into account the fate of contaminants and the decreases in CI values in the most contaminated sites would likely be offset by increased CI values in minimally or uncontaminated areas down-gradient. The majority of co-produced water at the Goose Lake detailed study area was introduced between 1967 and 1974, suggesting that saline contamination to aquatic resources in the PPR from oil and gas development in glacial outwash deposits could persist for 180 years after initial contamination.

Geophysical Surveys

Apparent conductivity measurements from geophysical surveys in 2009 using a Geonics EM-34 were used to test the second hypothesis: Contaminated groundwater plumes have migrated laterally; mainly confined within the permeable outwash sediments mantling the underlying glacial till. It was expected that the glacial till is acting as an aquitard, allowing little penetration and downward transport of saline groundwater relative to horizontal flow directions. Additionally, geophysical surveys were conducted in 2004 and 2009 with a Geonics EM-31 to determine the lateral extent of saline groundwater plumes, which allowed for a comparison between the apparent conductivity profiles.

EM-31 Surveys

Elevated apparent conductivities were measured with the EM-31 near the vast majority of contaminated groundwater wells. The strong correlation between elevated conductivities and co-produced water contamination validates the use of the EM-31 to map the lateral extent of saline groundwater plumes; however, the increase in conductivities near wetlands is likely the result of increased clay content and soil saturation. The EM-31 survey results in the southern groundwater flow path from 2004 and 2009 were similar but exhibited slight differences in conductivity profiles (Figs. 22 and 23). In 2004, several areas of extremely elevated conductivities were recorded in the slough below the Hammer M tank battery compared to one larger area in 2009, directly above the culvert beneath the county road. The greater total precipitation and shorter period of below average precipitation before 2009 (Fig. 29) probably produced more frequent recharge events capable of impounding contaminated surface water behind the culvert preceding 2009 relative to 2004. Once impounded, surface waters behind the culvert infiltrated into the shallow groundwater system, explaining the single area of extremely elevated conductivity measurements in 2009. The decrease in magnitude and expansion of the area of elevated conductivities near well RAB 2 likely illustrates the lower level of saline contamination seen in groundwater samples from RAB 2 in 2005 and 2009, resulting from the down-gradient transport of contaminants, respectively. Differences in conductivity measurements near oil field site 128 are primarily due to different survey lines, station locations, and total number of measurements between 2004 and 2009. The influence of topography and depth to the water table is evidenced by the thin strip of elevated measurements in the slough leading to well 264 R, with elevated measurements only obtained when the water table was within the exploration depth of the EM-31.

EM-31 surveys were conducted in the northern groundwater flow path in 2009 and defined saline groundwater plumes emanating from oil field sites 124 and 126. The majority of groundwater contamination in the northern flow path appears to have originated from oil field site 124, where elevated conductivities defined a contaminated groundwater plume that expands east and southward as the groundwater flows down hydraulic gradient towards West Goose Lake. The EM-31 surveys near oil field site 126 provide convincing evidence that the increase in chloride concentrations and CI values seen in wells 126 A and 126 C between 1989 and 2009 are the result of the transport line break in 2006. An area of elevated conductivity, not seen in 1989, defined a new contaminated groundwater plume expanding south from the approximate location of the transport line break past wells 126 A and 126 C. The increased contamination documented at oil field site 126 emphasizes the potential for continued contamination to aquatic resources in the PPR from routine oil field operations. Furthermore, the potential for co-produced water contamination from transport line breaks will likely increase as the number of transport lines increase with additional oil development in the Bakken Formation.

EM-34 Surveys

Elevated apparent conductivities were measured with the EM-34 near the vast majority of contaminated groundwater wells (Fig. 32). The strong correlation between elevated conductivities and co-produced water contamination validates the use of the EM-34 to characterize the three dimensional geometry of saline groundwater plumes (Figs. 24, 25, 27, 30 and 32). As in the EM-31 survey, elevated conductivities were measured in and near wetlands; however the effects of clay content and soil saturation in the near surface (i.e. wetlands 264 Y and 264 P) were only evident in the shallower surveys (10 m vertical and 10 m and 20 m horizontal dipole orientations). In contrast, the presence of buried lake clays (i.e. beneath wells 264 S and 264 T) resulted in elevated conductivities that persisted in the deeper surveys (20 m and 40 m vertical dipole orientation). The varying response from the different intercoil spacings and dipole orientations to changes in subsurface geology demonstrates the necessity of a solid stratigraphic framework for the interpretation of the EM-34 data. Using the refined stratigraphic framework for the Goose Lake detailed study area from the groundwater well logs, the EM-34 results documented minimal penetration of saline groundwater into the underlying till. Additionally, the EM-34 results illustrated the subsurface topography of the underlying till also provides a control on the lateral transport of contaminants.

Conductivities were generally much larger in the horizontal dipole orientation (Fig. 24) than the vertical dipole orientation (Fig. 25) throughout the entire study area. The horizontal dipole orientation receives the majority of the cumulative response from the saturated outwash deposits at the near surface, whereas the majority of the cumulative response is obtained from ~ 0.4 intercoil spacings in the vertical dipole orientation. In the 20 m and 40 m vertical dipole surveys, the majority of the signal is obtained from the underlying till as 0.4 intercoil spacings is deeper than the outwash deposits over much of the study site. Maximum conductivities in all horizontal dipole surveys were approximately 2.5 times greater than the maximum measurements from all vertical dipole surveys, due to the high level of saline contamination in the outwash deposits. The lower and nearly identical average conductivity and standard deviation in the 20 m and 40 m vertical dipole surveys attests to the minimal penetration of saline contamination into the underlying till. Therefore, the EM-34 survey results confirm the second hypothesis, illustrating that the underlying till is acting as an aquitard which prevents significant vertical migration of saline contaminated groundwater.

Furthermore, the controls on contaminant transport imposed from the subsurface topography of the underlying till were observed in the northern groundwater flow path (Fig.32). Here, well 124 P is located near a subsurface glacial till high, with the till surface elevation decreasing near wells 124 H to the north and well 124 O to the south.

Density gradients commonly develop in saline contaminated outwash deposits and the glacial till high likely results in the most contaminated groundwater being channeled through preferential flow paths in the deeper outwash deposits near wells 124 H and 124 O. The existence of two preferential groundwater flow paths around well 124 P are supported by elevated conductivities in all horizontal dipole surveys and the 10 m vertical dipole survey near wells 124 H and 124 O. Elevated conductivities were similar in magnitude, implying relatively consistent water chemistry, throughout the northern preferential flow path leading to West Goose Lake. In contrast, conductivities in the southern preferential flow path show a precipitous drop across the county road, east of well 124 O. The chloride concentration and CI value increased drastically at well 124 O between 2006 and 2009, and the elevated conductivities seen at and west of well 124 O are likely documenting the later arrival of extremely contaminated groundwater. The lower values across the county road imply that this highly contaminated groundwater has yet to migrate much past well 124 O. These results illustrate that while groundwater contamination is transported laterally in the saturated outwash deposits along hydraulic gradient, the subsurface topography of the underlying till can easily influence the nature and direction of down-gradient groundwater flow.

Normalized conductivity profiles of the EM-31 and EM-34 10 m horizontal dipole surveys were comparable, with most of the observed differences due to a greater number of measurements taken with the EM-31 compared to the EM-34 (Fig. 26). The background conductivity used for the EM-34 is considered a low estimate as measurements were taken over unsaturated outwash deposits. The potentially low background conductivity estimate is likely the cause of higher maximum normalized conductivities in the EM-34 relative to the EM-31; however maximum conductivities were similar 15.8 and 17.5, respectively. Results from the normalized conductivity profiles, demonstrate the effectiveness of this technique to compare readings from different geophysical equipment with similar exploration depths. Additionally, when the data are combined, the normalized conductivity profiles illustrate widespread saline groundwater contamination throughout the majority of the saturated outwash deposits within the Goose Lake detailed study area.

Finally, estimates of the maximum depth of penetration were similar for the southern and northern groundwater flow paths, with both values less than 30 m in 42 years, for a maximum vertical transport rate of ~ 0.7 m/yr. In contrast, saline contamination was observed in West Goose Lake in 1989 and at well 264 T in 2009, implying minimum lateral transport rates of ~ 35 and ~ 39 m/yr, respectively. The high levels of contamination still present in 2009, and the preferred lateral transport of contaminants in the overlying outwash deposits illustrates the longevity of saline contamination and the potential for co-produced water contamination to surface and groundwater resources far down-gradient of oil and gas developments in glacial outwash deposits in the PPR, respectively.

Implications for the Prairie Pothole Region

The results presented here have significant implications for the broader PPR due to the sheer magnitude of oil and gas development in the Williston Basin. Although geological maps are currently not at a fine enough resolution to accurately state the number of oil wells drilled in surficial glacial outwash deposits, as of January 2011 over 10,300 oil and gas wells have been drilled in the United States portion of the PPR (Data from Montana Department of Natural Resources and Conservation (http://bogc.dnrc.mt.gov/onlinedata.asp); North Dakota Industrial Commission, Department of Minneral Resources, Oil and Gas Divisions (https://www.dmr.nd.gov/oilgas/); and South Dakota Department of Environmental and Natural Resources, Minerals and Mining Program, Oil and Gas Section (http://denr.sd.gov/des/og/welldata.aspx), obtained 2/22/11). Furthermore, roughly 4,000 of these wells were installed prior to enactment of environmental laws pertaining to drilling fluids and reserve pits in 1974, and likely used unlined reserve pits to store drilling fluids and co-produced water. Subsequently, leachates from unlined reserve pits that can produce high levels of saline contamination are likely being generated from at least 40% of the oil field sites in the PPR. Finally, the majority of oil wells installed after 1974 probably lined the reserve pit and reclaimed the site by burial, resulting in approximately 6,000 additional buried reserve pits with the potential to contaminate aquatic resources if the liner fails.

Additionally, oil field sites within the Goose Lake detailed study area all produced from the Ratcliffe interval, whereas oil development has occurred in numerous geologic formations throughout the PPR. Eleven co-produced water samples from the Ratcliffe interval had TDS values that averaged approximately a third less than the values reported by Iampen and Rostron (2000) from fifty active oil wells producing from pre-Mississippian formations in North Dakota and Saskatchewan. The reason for the higher

TDS in co-produced waters from oil wells further east is likely due to stagnant groundwater flow conditions in the basin center, where brines are dominated by Na, Ca, and Cl, whereas the Goose Lake field is likely positioned above the mixing zone of the brackish, Ca and SO_4 dominated, brines on the west side of the basin (Iampen and Rostron 2000). Therefore, it would seem highly probable that saline contamination produced from oil field sites further east, in the stagnant groundwater flow zone, would have higher chloride and TDS concentrations than observed in this study. However, annual precipitation increases from west to east across the PPR, whereas evaporation increases from the northeast to the southwest, resulting in surface and groundwater chemistries that tend to be more naturally saline in the western portion of the PPR (Rosenberry, 2003). Therefore, the CI value used to determine the presence of co-produced water contamination would likely have to be refined to accommodate the natural changes in water chemistry before being applied to the broader PPR.

Finally, results from this thesis have strong implications for land owners and managers in the PPR. The PPR provides critical grassland and wetland habitats to migratory birds and waterfowl, with the region annually producing on average 50 – 80 % of North American waterfowl (Batt, 1989). In order to preserve the biological productivity of PPR wetlands, several groups including the USFWS, local and state governments, ducks unlimited, and pheasants forever, own and manage numerous tracts of land, similar to the USFWS management of the Rabenberg WPA in the southern half section of T36N, R58E, S27. Furthermore, agriculture and cattle framing are the dominate land uses in the PPR. Both wildlife conservation efforts and agriculture operations can be impaired due to the increased salinity from co-produced water contamination to surface and groundwater resources.

In 1988, the EPA released the national aquatic life criteria standards, which set the chronic exposure limit of chlorides at 230 mg/L. In the Goose Lake detailed study area, only two contaminated wetlands had chloride concentrations below 230 mg/L (264 P – 98 mg/L and 264 Y – 120 mg/L); however, these concentrations would likely increase as evaporation increases through the spring and summer. Thus, the high chloride concentrations observed in 2009 in the majority of wetlands in the Goose Lake detailed study area illustrates the longevity of co-produced water contamination and the challenges faced by land owners and managers in selecting and maintaining parcels of land for habitat conservation, where contamination can still be present from oil field activities conducted over fifty years ago. Additionally, the presence of saline groundwater contamination approximately 1,600 m down-gradient of the Hammer M tank battery in the southern groundwater flow path attests to the long distances contaminants can travel in the shallow groundwater system. Therefore, these results can aid land owners and managers in the PPR in assessing the likelihood of saline contamination to aquatic resources from oil and gas field sites installed in surficial glacial outwash deposits on future land purchases and to allocate limited funding resources to monitor existing co-produced water contamination.

Future Work

The research presented here leaves two fundamentally important questions unanswered. First, what is the nature of the reduction in chloride concentrations and CI values from natural attenuation in the PPR? For example, are decreases linear, exponential, or do changes in recharge patterns and differences in the water table configuration control the release of contaminants creating varying responses for different oil field sites in the same geologic materials? This leads to the second question; what is the influence of climate, namely the amount and timing of yearly precipitation, on the release of contaminants?

The first question could likely be answered by additional water sampling within the Goose Lake detailed study area. Taking water samples from the same locations in another ten and twenty years would be relatively inexpensive and should provide the additional data points needed to determine the nature of reductions in the CI values in glacial outwash deposits. Additionally, the two flow paths had different recharge patterns, water table configurations, and CI profiles. Therefore, continued monitoring here could also evaluate the potential of differences in the nature of natural attenuation within similar glacial sediments. Understanding the nature of natural attenuation and whether the potential exists for different responses in similar geologic materials would be extremely useful for land owners and managers in determining the longevity and future magnitude of saline contamination in aquatic resources down-gradient of oil field sites.

The second question could be evaluated with additional geophysical surveys in another twenty years. This question is important as the PPR contains numerous buried glacial outwash aquifers that are used as sources of drinking water. The continued vertical migration of saline groundwater could potentially lead to the contamination of municipal water supplies in areas where buried aquifers are beneath oil field sites. If the contamination remains at roughly the same depth, this may allow an estimate of the maximum depth of penetration expected beneath oil field sites in similar geologic settings. In contrast, if the contamination continues to penetrate vertically, albeit slowly, this could imply a substantial risk of co-produced water contamination to socially important buried glacial outwash aquifers in the PPR.

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APPENDICES

APPENDIX A:

SLUG TEST DATA

124H

Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points
17/07/2009 19:21:58	1213	
17/07/2009 19:22:03	1213	
17/07/2009 19:22:08	1213	
17/07/2009 19:22:13	1202	
17/07/2009 19:22:18	1204	
17/07/2009 19:22:23	1203	
17/07/2009 19:22:28	1205	
17/07/2009 19:22:33	1205	
17/07/2009 19:25:38	684	y(o)
17/07/2009 19:25:43	851	
17/07/2009 19:25:48	961	
17/07/2009 19:25:53	1041	
17/07/2009 19:25:58	1102	
17/07/2009 19:26:03	1133	
17/07/2009 19:26:08	1155	
17/07/2009 19:26:13	1174	
17/07/2009 19:26:18	1176	
17/07/2009 19:26:23	1198	y(t)
17/07/2009 19:26:28	1201	
17/07/2009 19:26:33	1203	
17/07/2009 19:26:38	1204	
17/07/2009 19:26:43	1204	
17/07/2009 19:26:48	1204	
17/07/2009 19:26:53	1205	
17/07/2009 19:26:58	1205	
17/07/2009 19:27:03	1205	
17/07/2009 19:27:08	1205	

124J

Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points
17/07/2009 18:44:17	1233	
17/07/2009 18:44:27	1232	
17/07/2009 18:44:37	1230	
17/07/2009 18:44:47	1229	
17/07/2009 18:44:57	1228	
17/07/2009 18:48:57	550	
17/07/2009 18:49:07	533	y(o)
17/07/2009 18:49:17	790	
17/07/2009 18:49:27	985	
17/07/2009 18:49:37	1082	
17/07/2009 18:49:47	1137	y(t)
17/07/2009 18:49:57	1163	
17/07/2009 18:50:07	1177	
17/07/2009 18:50:17	1184	
17/07/2009 18:50:27	1188	
17/07/2009 18:50:37	1190	
17/07/2009 18:50:47	1190	
17/07/2009 18:50:57	1191	
17/07/2009 18:51:07	1191	
17/07/2009 18:51:17	1192	
17/07/2009 18:51:27	1191	
17/07/2009 18:51:37	1190	
17/07/2009 18:51:47	1190	
17/07/2009 18:51:57	1190	

264B

Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points
17/07/2009 16:52:02	1243	
17/07/2009 16:52:12	1243	
17/07/2009 16:52:22	1243	
17/07/2009 16:52:32	1243	
17/07/2009 16:52:42	1243	
17/07/2009 16:52:52	1243	
17/07/2009 16:53:02	1243	
17/07/2009 16:55:22	245	y(o)
17/07/2009 16:55:32	521	
17/07/2009 16:55:42	737	
17/07/2009 16:55:52	819	
17/07/2009 16:56:02	914	
17/07/2009 16:56:12	978	
17/07/2009 16:56:22	1022	
17/07/2009 16:56:32	1057	
17/07/2009 16:56:42	1096	
17/07/2009 16:56:52	1137	
17/07/2009 16:57:02	1161	y(t)
17/07/2009 16:57:12	1176	
17/07/2009 16:57:22	1187	
17/07/2009 16:57:32	1193	
17/07/2009 16:57:42	1199	
17/07/2009 16:57:52	1203	
17/07/2009 16:58:02	1206	
17/07/2009 16:58:12	1209	
17/07/2009 16:58:22	1211	
17/07/2009 16:58:32	1212	
17/07/2009 16:58:42	1214	
17/07/2009 16:58:52	1213	
17/07/2009 16:59:02	1215	
17/07/2009 16:59:12	1216	
17/07/2009 16:59:22	1217	
17/07/2009 16:59:32	1218	
17/07/2009 16:59:42	1219	
17/07/2009 16:59:52	1219	
17/07/2009 17:00:02	1220	
17/07/2009 17:00:12	1221	
17/07/2009 17:00:22	1221	
17/07/2009 17:00:32	1222	
17/07/2009 17:00:42	1222	

264R

Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points
16/07/2009 19:34:09	1234	
16/07/2009 19:34:29	1234	
16/07/2009 19:34:49	1234	
16/07/2009 19:35:09	1231	
16/07/2009 19:35:29	1230	
16/07/2009 19:35:49	1230	
16/07/2009 19:36:09	1230	
16/07/2009 19:36:29	1230	
16/07/2009 19:40:09	157	
16/07/2009 19:40:29	346	y(o)
16/07/2009 19:40:49	479	
16/07/2009 19:41:09	592	
16/07/2009 19:41:29	688	
16/07/2009 19:41:49	766	
16/07/2009 19:42:09	835	
16/07/2009 19:42:29	893	
16/07/2009 19:42:49	944	
16/07/2009 19:43:09	987	
16/07/2009 19:43:29	1020	
16/07/2009 19:43:49	1041	
16/07/2009 19:44:09	1066	
16/07/2009 19:44:29	1089	
16/07/2009 19:44:49	1108	
16/07/2009 19:45:09	1124	
16/07/2009 19:45:29	1139	
16/07/2009 19:45:49	1153	
16/07/2009 19:46:09	1164	
16/07/2009 19:46:29	1174	
16/07/2009 19:46:49	1183	
16/07/2009 19:47:09	1191	
16/07/2009 19:47:29	1198	
16/07/2009 19:47:49	1203	
16/07/2009 19:48:09	1207	y(t)
16/07/2009 19:48:29	1211	
16/07/2009 19:48:49	1216	
16/07/2009 19:49:09	1219	
16/07/2009 19:49:29	1221	
16/07/2009 19:49:49	1224	
16/07/2009 19:50:09	1226	
16/07/2009 19:50:29	1228	
16/07/2009 19:50:49	1229	
16/07/2009 19:51:09	1231	
16/07/2009 19:51:29	1231	
16/07/2009 19:51:49	1232	

126C Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points	126C continued Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points	126C continued Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points
17/07/2009 17:02:42	1226		17/07/2009 18:13:53	939		17/07/2009 18:23:33	1090	
17/07/2009 17:02:52	1226		17/07/2009 18:14:03	943		17/07/2009 18:23:43	1092	
17/07/2009 17:03:02	1226		17/07/2009 18:14:13	946		17/07/2009 18:23:53	1094	
17/07/2009 17:03:12	1226		17/07/2009 18:14:23	950		17/07/2009 18:24:03	1095	
17/07/2009 18:01:53	1262		17/07/2009 18:14:33	953		17/07/2009 18:24:13	1097	
17/07/2009 18:02:03	1261		17/07/2009 18:14:43	956		17/07/2009 18:24:23	1099	
17/07/2009 18:02:13	1277		17/07/2009 18:14:53	960		17/07/2009 18:24:33	1101	
17/07/2009 18:02:23	1274		17/07/2009 18:15:03	963		17/07/2009 18:24:43	1103	
17/07/2009 18:02:33	1273		17/07/2009 18:15:13	966		17/07/2009 18:24:53	1104	
17/07/2009 18:02:43	1272		17/07/2009 18:15:23	969		17/07/2009 18:25:03	1106	
17/07/2009 18:02:53	1271		17/07/2009 18:15:33	973		17/07/2009 18:25:13	1108	
17/07/2009 18:06:03	670		17/07/2009 18:15:43	976		17/07/2009 18:25:23	1110	
17/07/2009 18:06:13	681		17/07/2009 18:15:53	979		17/07/2009 18:25:33	1116	
17/07/2009 18:06:23	688		17/07/2009 18:16:03	982		17/07/2009 18:25:43	1118	
17/07/2009 18:06:33	694	y(o)	17/07/2009 18:16:13	985		17/07/2009 18:25:53	1119	
17/07/2009 18:06:43	700		17/07/2009 18:16:23	989		17/07/2009 18:26:03	1121	y(t)
17/07/2009 18:06:53	706		17/07/2009 18:16:33	992		17/07/2009 18:26:13	1123	
17/07/2009 18:07:03	713		17/07/2009 18:16:43	995		17/07/2009 18:26:23	1124	
17/07/2009 18:07:13	720		17/07/2009 18:16:53	998		17/07/2009 18:26:33	1125	
17/07/2009 18:07:23	728		17/07/2009 18:17:03	1001		17/07/2009 18:26:43	1127	
17/07/2009 18:07:33	736		17/07/2009 18:17:13	1004				
17/07/2009 18:07:43	748		17/07/2009 18:17:23	1007				
17/07/2009 18:07:53	756		17/07/2009 18:17:33	1010				
17/07/2009 18:08:03	764		17/07/2009 18:17:43	1013				
17/07/2009 18:08:13	771		17/07/2009 18:17:53	1015				
17/07/2009 18:08:23	779		17/07/2009 18:18:03	1018				
17/07/2009 18:08:33	786		17/07/2009 18:18:13	1021				
17/07/2009 18:08:43	793		17/07/2009 18:18:23	1023				
17/07/2009 18:08:53	800		17/07/2009 18:18:33	1025				
17/07/2009 18:09:03	806		17/07/2009 18:18:43	1028				
17/07/2009 18:09:13	813		17/07/2009 18:18:53	1030				
17/07/2009 18:09:23	818		17/07/2009 18:19:03	1033				
17/07/2009 18:09:33	823		17/07/2009 18:19:13	1036				
17/07/2009 18:09:43	828		17/07/2009 18:19:23	1038				
17/07/2009 18:09:53	834		17/07/2009 18:19:33	1040				
17/07/2009 18:10:03	839		17/07/2009 18:19:43	1043				
17/07/2009 18:10:13	844		17/07/2009 18:19:53	1045				
17/07/2009 18:10:23	849		17/07/2009 18:20:03	1048				
17/07/2009 18:10:33	854		17/07/2009 18:20:13	1050				
17/07/2009 18:10:43	858		17/07/2009 18:20:23	1053				
17/07/2009 18:10:53	863		17/07/2009 18:20:33	1056				
17/07/2009 18:11:03	867		17/07/2009 18:20:43	1059				
17/07/2009 18:11:13	872		17/07/2009 18:20:53	1061				
17/07/2009 18:11:23	877		17/07/2009 18:21:03	1063				
17/07/2009 18:11:33	881		17/07/2009 18:21:13	1065				
17/07/2009 18:11:43	886		17/07/2009 18:21:23	1068				
17/07/2009 18:11:53	891		17/07/2009 18:21:33	1069				
17/07/2009 18:12:03	895		17/07/2009 18:21:43	1072				
17/07/2009 18:12:13	899		17/07/2009 18:21:53	1074				
17/07/2009 18:12:23	903		17/07/2009 18:22:03	1075				
17/07/2009 18:12:33	907		17/07/2009 18:22:13	1077				
17/07/2009 18:12:43	911		17/07/2009 18:22:23	1079				
17/07/2009 18:12:53	916		17/07/2009 18:22:33	1080				
17/07/2009 18:13:03	920		17/07/2009 18:22:43	1082				
17/07/2009 18:13:13	924		17/07/2009 18:22:53	1084				
17/07/2009 18:13:23	928		17/07/2009 18:23:03	1086				
17/07/2009 18:13:33	932		17/07/2009 18:23:13	1088				
17/07/2009 18:13:43	936		17/07/2009 18:23:23	1089				

264S

Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points
17/07/2009 16:10:41	1264	
17/07/2009 16:10:56	1264	
17/07/2009 16:11:11	1264	
17/07/2009 16:11:26	1264	
17/07/2009 16:11:41	1264	
17/07/2009 16:11:56	1264	
17/07/2009 16:12:11	1264	
17/07/2009 16:12:26	1264	
17/07/2009 16:12:41	1264	
17/07/2009 16:12:56	1264	
17/07/2009 16:13:11	1264	
17/07/2009 16:13:26	1264	
17/07/2009 16:13:41	1264	
17/07/2009 16:13:56	1264	
17/07/2009 16:14:11	1263	
17/07/2009 16:14:26	1262	
17/07/2009 16:14:41	1262	
17/07/2009 16:14:56	1262	
17/07/2009 16:15:11	1261	
17/07/2009 16:15:26	1261	
17/07/2009 16:15:41	1261	
17/07/2009 16:15:56	1261	
17/07/2009 16:17:26	85	
17/07/2009 16:17:41	90	
17/07/2009 16:17:56	101	
17/07/2009 16:18:11	110	
17/07/2009 16:18:26	300	y(o)
17/07/2009 16:18:41	420	
17/07/2009 16:18:56	527	
17/07/2009 16:19:11	612	
17/07/2009 16:19:26	690	
17/07/2009 16:19:41	759	
17/07/2009 16:19:56	821	
17/07/2009 16:20:11	876	
17/07/2009 16:20:26	923	
17/07/2009 16:20:41	964	
17/07/2009 16:20:56	1001	
17/07/2009 16:21:11	1030	
17/07/2009 16:21:26	1050	
17/07/2009 16:21:41	1074	
17/07/2009 16:21:56	1092	
17/07/2009 16:22:11	1104	
17/07/2009 16:22:26	1120	
17/07/2009 16:22:41	1129	
17/07/2009 16:22:56	1144	
17/07/2009 16:23:11	1156	
17/07/2009 16:23:26	1167	
17/07/2009 16:23:41	1178	
17/07/2009 16:23:56	1187	
17/07/2009 16:24:11	1196	
17/07/2009 16:24:26	1205	
17/07/2009 16:24:41	1209	
17/07/2009 16:24:56	1214	
17/07/2009 16:25:11	1219	
17/07/2009 16:25:26	1224	
17/07/2009 16:25:41	1230	
17/07/2009 16:25:56	1234	
17/07/2009 16:26:11	1237	

264S continued

Date Time d/m/y 24hr	Lineraly Adj. Water Height mm	Interval Points
17/07/2009 16:26:26	1239	y(t)
17/07/2009 16:26:41	1241	
17/07/2009 16:26:56	1244	
17/07/2009 16:27:11	1245	
17/07/2009 16:27:26	1247	
17/07/2009 16:27:41	1248	
17/07/2009 16:27:56	1250	

Slug-Test Worksheet

Enter values/notes in bordered cells. Spreadsheet will do the rest.

Use "Save As" command to save results.

See Bouwer and Rice (1976) for further details.

Spreadsheet by Don Rosenberry, v. 2, modified to calc. A, B, and C

Explaination of Variables

D = Vertical distance between water table and bottom of aquifer (ft)

H = Vertical distance between water table and bottom of well (ft)

rc = Inside radius of casing plus any porosity in permeable envelope (dependent on water level) (ft)

rw = Radial distance between the undisturbed aquifer and the well center (ft)

L = Vertical distance of perforated section of well (ft)

L/rw = Value used in determining variables A, B, and C

A = Dimensionless coefficient that is a function of L/rw. (obtained from Fig. 3, Bouwer and Rice, 1976)

B = Dimensionless coefficient that is a function of L/rw. (obtained from Fig. 3, Bouwer and Rice, 1976)

C = Dimensionless coefficient that is a function of L/rw. (obtained from Fig. 3, Bouwer and Rice, 1976)

y(o) = Vertical distance between water level in well and equilibrium water table in aquifer at t = 0. (ft)

y(t) = Vertical distance between water level in well and equilibrium water table in aquifer at t = t. (ft)

t = Time (decimal minutes)

ln[(D-H)/rw] = Effective upper limit is 6. If this value is greater than six spreadsheet will output 6.

ln(Re/rw) = Effective radius over which y is dissipated. This uses coefficients A and B in calculation, unless D = H, in which case coefficient C is used.

Site =	124 H	**Well ID =**	890935
Date =	7/17/2009	**Inj./Withd. =**	WITHDRAWL

If yes, and test is injection, results may be invalid.

Variables	**Enter data here**		
D	14.37	Is static water level	
H	14.37	in screened interval of well?	no
rc	0.10	0.10	Note: Assumes ø = 0.3
rw	0.29		
L	4.00		
L/rw	13.71		
A	1.91	Note: A, B, and C are calculated based	
B	0.28	on Yang and Yeh, 2004	
C	1.22	Note: Use C if D=H, otherwise leave blank.	
y(o)	1.74		
y(t)	0.05		
t	1.25		
ln[(D-H)/rw]	#NUM!		
ln(Re/rw)	2.70		
K (ft/d) =	**1.35E+01**		
K (cm/s) =	**4.74E-03**		
K (m/d) =	**4.10E+00**		

Notes:

Assumptions:
D = H
L = Only the section in the outwash

Site =	124 J	**Well ID =**	890928	
Date =	7/17/2009	**Inj./Withd. =**	WITHDRAWL	

If yes, and test is injection, results may be invalid.

Variables	**Enter data here**			
D	12.66	Is static water level		
H	12.66	in screened interval of well?		no
rc	0.10	0.10	Note: Assumes ø = 0.3	
rw	0.29			
L	7.00			
L/rw	24.00			
A	2.29	Note: A, B, and C are calculated based		
B	0.30	on Yang and Yeh, 2004		
C	1.62	Note: Use C if D=H, otherwise leave blank.		
y(o)	2.30			
y(t)	0.31			
t	1.11			
ln[(D-H)/rw]	#NUM!			
ln(Re/rw)	2.78			
K (ft/d) =	**5.01E+00**			
K (cm/s) =	**1.77E-03**			
K (m/d) =	**1.53E+00**			

Notes:

Assumptions:
D = H
L = Only the section in the outwash

Site =	126 C	**Well ID =**	890942
Date =	7/17/2009	**Inj./Withd. =**	WITHDRAWL

If yes, and test is injection, results may be invalid.

Variables	**Enter data here**		
D	8.44	Is static water level	
H	8.44	in screened interval of well?	no
rc	0.10	0.10	Note: Assumes ø = 0.3
rw	0.29		
L	4.00		
L/rw	13.71		
A	1.91	Note: A, B, and C are calculated based	
B	0.28	on Yang and Yeh, 2004	
C	1.22	Note: Use C if D=H, otherwise leave blank.	
y(o)	1.86		
y(t)	0.46		
t	32.50		
ln[(D-H)/rw]	#NUM!		
ln(Re/rw)	2.41		
K (ft/d) =	**1.81E-01**		
K (cm/s) =	**6.40E-05**		
K (m/d) =	**5.53E-02**		

Notes:

Assumptions:
D = H
L = Only the section in the outwash

Site =	264 B	**Well ID =**	890445
Date =	7/17/2009	**Inj./Withd. =**	WITHDRAWL

If yes, and test is injection, results may be invalid.

Variables	**Enter data here**		
D	7.00	Is static water level	
H	6.96	in screened interval of well?	no
rc	0.10	0.10	Note: Assumes ø = 0.3
rw	0.29		
L	3.00		
L/rw	10.29		
A	1.81	Note: A, B, and C are calculated based	
B	0.29	on Yang and Yeh, 2004	
C	blank	Note: Use C if D=H, otherwise leave blank.	
y(o)	3.27		
y(t)	0.27		
t	2.78		
ln[(D-H)/rw]	-1.88		
ln(Re/rw)	2.13		
K (ft/d) =	**4.50E+00**		
K (cm/s) =	**1.59E-03**		
K (m/d) =	**1.37E+00**		

Notes:

Assumptions:
D = H
L = Only the section in the outwash

Site =	264 R	**Well ID =**	890936
Date =	7/17/2009	**Inj./Withd. =**	WITHDRAWL

If yes, and test is injection, results may be invalid.

Variables	**Enter data here**		
D	15.34	Is static water level	
H	15.34	in screened interval of well?	no
rc	0.10	0.10	Note: Assumes ø = 0.3
rw	0.31		
L	4.00		
L/rw	12.80		
A	1.88	Note: A, B, and C are calculated based	
B	0.28	on Yang and Yeh, 2004	
C	1.19	Note: Use C if D=H, otherwise leave blank.	
y(o)	2.91		
y(t)	0.09		
t	12.78		
ln[(D-H)/rw]	#NUM!		
ln(Re/rw)	2.66		
K (ft/d) =	**1.29E+00**		
K (cm/s) =	**4.54E-04**		
K (m/d) =	**3.92E-01**		

Notes:

Assumptions:
D = H
L = Only the section in the outwash

Site =	264 S	**Well ID =**	890938
Date =	7/16/2009	**Inj./Withd. =**	WITHDRAWL

If yes, and test is injection, results may be invalid.

Variables	**Enter data here**		
D	13.83	Is static water level	
H	13.83	in screened interval of well?	no
rc	0.10	0.10	Note: Assumes ø = 0.3
rw	0.29		
L	5.00		
L/rw	17.14		
A	2.03	Note: A, B, and C are calculated based	
B	0.28	on Yang and Yeh, 2004	
C	1.34	Note: Use C if D=H, otherwise leave blank.	
y(o)	3.16		
y(t)	0.08		
t	13.33		
ln[(D-H)/rw]	#NUM!		
ln(Re/rw)	2.75		
K (ft/d) =	**1.07E+00**		
K (cm/s) =	**3.77E-04**		
K (m/d) =	**3.26E-01**		

Notes:

Assumptions:
D = H
L = Only the section in the outwash

APPENDIX B:

WATER CHEMISTRY DATA

Appendix 1

Field Parameters and Analytical Chemical Analysis of Water Samples from the Goose Lake Detailed Study Site

Site information, field parematers, and strontium isotope data (1/2)

Site name	Sample date	Sample time	MBMG GWIC ID	Latitude	Longitude	Projection	Site type	Well depth (bls)	Below mp	mp height	Below land surface	Temp (degrees Celsius)	pH (standard units)	Specific conductance (uS/cm)	Chloride (mg/L)	Oxygen Reduction Potential	87Sr/86Sr stable-isotope ratio (per mil)	d87Sr
BGWL	5/15/2009	1230	--	485110.2	1040719.0	NAD27	Wetland	--	--	--	--	9.09	8.43	378			--	--
BGWL-r	5/15/2009	1235	--	485110.2	1040719.0	NAD27	QA/QC	--	--	--	--	9.09	8.43	378			--	--
BGWL	5/15/2009	1235										9.09	8.43	378			0.70898	-0.3
PW1	5/15/2009	1100	--	485105	1040718.0	NAD27	Oil tank	--	--	--	--	--	--	291,400			--	--
PW1	5/15/2009	1100										--	--	291,400			0.70802	-1.67
117F	4/21/1989													2,080	450			
117F	9/17/2009	930		485122	1040641.0	NAD27	Wetland	--	--	--	--	--	9.07	10,870			--	--
117F	9/17/2009	935										--	9.07	10,870			0.71	1.21
117F-r	9/17/2009	935		485122	1040641.0	NAD27	Wetland	--	--	--	--	--	9.07	10,870			--	--
117F-r	9/17/2009	935										--	9.07	10,870			0.71004	1.18
117J	10/13/1989	915	890941	485110.8	1040621.1	NAD27	Well	15				10.3	--	--			--	--
117J	4/11/2006	1445										4.43	7.02	11,884			--	--
117J	5/15/2009	945							5.37	3.6	1.77	3.39	7.31	5,862			--	--
124C	4/21/1989	--	890429	485105	1040618.0	NAD27	Well	8				6.9	7.26	27,570		-129.5	--	--
124C	4/11/2006	1251										3.41	7.33	6,066			--	--
124C	5/14/2009	1445							5.6	2.6	3	5.95	7.66	7,220			--	--
124D	4/21/1989													11,090	5060			
124D	5/14/2009	1400	220933	485104	1040615.0	NAD27	Wetland	--	--	--	--	12	8.43	5,711			--	--
124D	5/14/2009	1400										12	8.43	5,711			0.70858	-0.88
124E	5/2/1989													34,700	16,870			
124E	5/14/2009	1530	220934	485058	1040608.0	NAD27	Wetland	--	--	--	--	13.54	8.93	20,760			--	--
124E	5/14/2009	1530										13.54	8.93	20,760			0.70861	-0.83
124F	5/2/1989													27,030	12,790			
124F	5/14/2009	1430	--	485057.7	1045057.7	NAD27	Wetland	--	--	--	--	13.9	8.65	13,727			--	--
124F	5/14/2009	1430										13.9	8.65	13,727			0.70884	-0.51
124H	10/12/1989	1545	890935	485103	1040557.2	NAD27	Well	17				12.9	6.68	95,690			--	--
124H	4/12/2006	1154										5.82	6.93	39,576			--	--
124H	5/15/2009	1045							3.77	1.9	1.87	5.09	6.83	49,456			--	--
124J	10/12/1989	1220	890928	485105	1040620.0	NAD27	Well	18				14.2	6.95	46,500			--	--
124J	4/10/2006	1430										4.43	7.22	13,419			--	--
124J	5/14/2009	1345							6.57	2.1	4.47	4.54	7.37	17,005	5,600	-157.4	--	--
124J	5/14/2009	1345										4.54	7.37	17,005			0.70853	-0.94
124N	10/17/1989	1130	890929	485105.8	1040627.1	NAD27	Well	20.5				15.9	7.19	31,740			--	--
124N	4/4/2006	1350										6.6	7.15	14,052			--	--
124N	5/14/2009	1215							11.13	1.3	9.83	6.97	7.57	9,807		-135.1	--	--
124O	10/12/1989	1630	890932	485048	1040557.0	NAD27	Well	23				14	6.79	21,160			--	--
124O	4/12/2006	1556										6.72	7.03	9,339			--	--
124O	5/16/2009	1215							9.22	1.5	7.72	11.4	6.76	40,168		-139.9	--	--
124O-b	5/16/2009	1230	--	--	--	--	QA/QC		--	--	--	15	9.27	52			--	--
124P	10/12/1989	1715	890933	485054.9	1040557.7	NAD27	Well	17				14.3	7.3	41,850	16,228		--	--
124P	4/12/2006	1417										10.18	7.34	56,076			--	--
124P	5/16/2009	1115							10.79	1.5	9.29	7.47	8.17	12,318		-100.2	--	--

Appendix 1 (Cont.)

Field Parameters and Analytical Chemical Analysis of Water Samples from the Goose Lake Detailed Study Site

Site information, field parematers, and strontium isotope data (2/2)

Site name	Sample date	Sample time	MBMG GWIC ID	Latitude	Longitude	Projection	Site type	Well depth (bls)	Below mp	mp height	Below land surface	Temp (degrees Celsius)	pH (standard units)	Specific conductance (uS/cm)	Chloride (mg/L)	Oxygen Reduction Potential	87Sr/86Sr stable-isotope ratio (per mil)	d87Sr
126A	4/20/1989	1415	890432	485109	1040640.0	NAD27	Well	22				8.2	7.25	6,360			–	–
126A	5/14/2009	945							12.73	3.2	9.53	6.02	7.22	17,101		-133.3	–	–
126B	4/20/1989	1500	890433	485107	1040637.0	NAD27	Well	23				8.5	6.86	32,600			–	–
126B	4/5/2006	1135										10.76	7.27	41,712			–	–
126B	5/14/2009	1730							15.57	1.4	14.17	11.9	7.28	39,620			–	–
126C	10/12/1989	1040	890942	485107	1040643.0	NAD27	Well	18.9				13.9	7.35	1,018			–	–
126C	5/15/2009	1630							12.73	3.2	9.53	7.56	7.23	3,476			–	–
126C	5/15/2009	1630										7.56	7.23	3,476			0.70935	0.21
264A	10/13/1989	1230	890940	485102	1040719.0	NAD27	Well	23				8.8	–	–			–	–
264A	9/15/2005	1426										13.7	6.8	78,143			–	–
264A	5/13/2009	1230							17.39	1.85	15.54	4.23	6.92	90,270			–	–
264A-r	5/13/2009	1235		485102	1040719.0	NAD27	QA/QC					4.23	6.92	90,270			–	–
264A	5/13/2009	1235										4.23	6.92	90,270			0.70828	-1.3
264B	4/20/1989	1545	890445	485059.7	1040718.8	NAD27	Well	8				6	–	–			–	–
264B	9/15/2005	1355										18.1	6.64	58,884			–	–
264B	5/15/2009	1430							2.98	3.1	-0.12	8.46	7.45	3,270		-141.5	–	–
264D	4/20/1989	1645	890446	485054.5	1040713.1	NAD27	Well	13				5	–	–			–	–
264D	9/15/2005	1506										16.4	7.01	21,625			–	–
264D	5/15/2009	1645							2.38	2.2	0.18	5.97	6.92	16,283		-173.3	–	–
264J	5/1/1989													1,740	350			
264J	5/12/2009	1530	220987	485042	1040701.3	NAD27	Wetland	–	–	–	–	18.1	7.68	2,720			–	–
264K	5/1/1989													6,800	2,190			
264K	5/12/2009	1545	–	485033.4	1040705.2	NAD27	Wetland	–	–	–	–	18	8.02	4,080	930		–	–
264M	5/1/1989													890	140			
264M	9/17/2009	1100	–	485032	1040700.0	NAD27	Wetland	–	–	–	–	–	8.49	4,520			–	–
264M	9/17/2009	1100										–	8.49	4,520			0.70991	1
264P	5/2/1989													840	80			
264P	5/12/2009	1645	220993	485034.5	1040614.3	NAD27	Wetland	–	–	–	–	17.7	8.88	2,517			–	–
264Q	10/13/1989	1100	890939	485051.6	1040708.1	NAD27	Well	23				13.7	–	–			–	–
264Q	5/15/2009	1530							4.56	2	2.56	9.97	7.16	11,816		-152.7	–	–
264R	10/13/1989	915	890936	485040.3	1040557.2	NAD27	Well	22				10.7	–	–			–	–
264R	5/15/2009	1800							7.37	1.3	6.07	6.13	7.21	3,342		-155.7	–	–
264S	10/13/1989	1030	890938	485031.1	1040655.9	NAD27	Well	18				13.4	–	–			–	–
264S	5/16/2009	1330							5.02	2.9	2.12	8.02	7.53	5,693			–	–
264T	10/13/1989	945	890937	485033	1040638.8	NAD27	Well	27				11.7	–	–			–	–
264T	5/16/2009	1430							5.22	2.85	2.37	5.73	7.66	4,227		-166	–	–
264Y	5/12/2009	1620	–	485031.5	1040640.5	NAD27	Wetland	–	–	–	–	17.78	7.89	1,196			–	–
264Y	5/12/2009	1620										17.78	7.89	1,196			0.71042	1.72
307B	4/21/1989													41,680	4480			
307B	5/15/2009	1030	221011	485102	1040544.2	NAD27	Wetland	–	–	–	–	9.54	8.83	63,916			–	–
307B	5/15/2009	1030										9.54	8.83	63,916			0.70922	0.02
RAB-02	9/15/2005	1226	221723	485037.4	1040706.8	NAD27	Well	13				13.9	7.24	5,620			–	–
RAB-02	5/12/2009	1145							9.62	1.85	7.77	9.01	7.2	4,397	1,446		–	–

Appendix 1 (cont.)

Field Parameters and Analytical Chemical Analysis of Water Samples from the Goose Lake Detailed Study Site

Analytical Chemical Data (1/2)

Site name	Sample date	pH (standard units)	Specific conductance (uS/cm)	Hardness (mg/L as CaCO3)	Calcium (mg/L as Ca)	Magnesium (mg/L as Mg)	Potassium (mg/L as K)	Sodium Absorption Ratio	Sodium (mg/L as Na)	Alkalinity (mg/L as CaCO3)	Iodide (mg/L as I)	Bromide (mg/L as Br)	Chloride (mg/L as Cl)	Flouride (mg/L as F)	Silica (mg/L as SiO2)	Sulfate (mg/L as SO4)	Boron (ug/L as B)	Iron (ug/L as Fe)	CI (Cl/S.C.)	Dissolved solids, computed (mg/L)
BGWL	5/15/2009	7.6	394	140	43	8.04	16.7	0.8	22	190	0.015	< 0.02	9	E 0.05	54.6	1	258	88	0.022	269
BGWL-r	5/15/2009	7.8	387	140	43	7.96	16.2	0.8	22	190	0.016	< 0.02	9	E 0.05	44.2	0	259		0.023	256
BGWL	5/15/2009	--	--	--	E 37	--	--	--	--	--	--	--	--	--	E 64	--	233	--		--
PW1	5/15/2009	6.1	E 223,000	17,000	5,030	961.00	2040	240	70,500	142	29.5	240	121,000	2.65	16.7	1,230	230,000	3,650	0.543	201,365
PW1	5/15/2009	--	--	--	E 4,950	--	--	--	--	--	--	--	--	--	< 61	--	E 176,000	--		--
117F	4/21/1989																		0.216	
117F	9/17/2009	8.92	10,750	4,370	992	459.00	71.7	6.9	1,044	84	0.034	4.05	2,941	0.13	11.7	1,775	1,103	E 13	0.274	7,350
117F	9/17/2009	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--		--
117F-r	9/17/2009	8.94	10,690	4,270	967	452.00	69.4	6.7	1,010	84	0.034		2,957	0.14	12.2	1,787	1,037	E 14	0.277	7,310
117F-r	9/17/2009	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--		--
117J	10/13/1989	7.09	9,211	3,052	469	457.00	52.6	5.868	745	255	--	2.1	2,560	0.2	31.4	859	310	9,465	0.278	5,497
117J	4/11/2006	7.24	11,000	4,048	586	628.00	58.5	7.365	1,077	367	--	< 2.5	3,434	< 2.5	29.1	1,010	< 300	10,800	0.312	7,284
117J	5/15/2009	7.2	5,980	2,100	302	316.00	41.8	5.7	593	267	0.009	1.57	1,550	0.16	24.8	634	203	2,930	0.259	3,624
124C	4/21/1989	7.3	29,096	2,045	501	193.00	110	59.7	6,180	249	--	< 0.1	10,600	< 1	15.4	234	17,500	15	0.364	18,149
124C	4/11/2006	7.5	5,810	903	159	123.00	24.4	13.117	906	307.56	--	2.59	1,678	< 2.5	17.8	< 125	2,810	< 500	0.289	3,283
124C	5/14/2009	7.4	7,540	870	160	114.00	29.5	17	1,130	283	0.19	3.78	2,330	0.26	19.9	113	4,370	< 16	0.309	4,075
124D	4/21/1989																		0.456	
124D	5/14/2009	7.6	E 10,600	1,400	284	171.00	61	18	1,520	160	0.2	4.78	3,280	0.17	1	605	7,150	75	0.309	6,030
124D	5/14/2009	--	--	--	E 338	--	--	--	--	--	--	--	--	--	< 4.3	--	E 6,100	--		--
124E	5/2/1989																		0.486	
124E	5/14/2009	8.3	E 22,000	2,000	306	298.00	97.9	38	3,910	< 800	0.449	12.5	7,020	E 0.09	2.3	729	14,200	83	0.319	12,870
124E	5/14/2009	--	--	--	E 337	--	--	--	--	--	--	--	--	--	< 11	--	E 12,100	--		--
124F	5/2/1989																		0.473	
124F	5/14/2009	7.9	E 14,700	1,200	261	134.00	54.5	27	2,150	256	0.107	6.98	4,870	0.3	6.2	299	7,810	E 32	0.331	7,943
124F	5/14/2009	--	--	--	E 329	--	--	--	--	--	--	--	--	--	7.3	--	E 7,400	--		--
124H	10/12/1989	7.47	78,617	10,260	1,900	1,340.00	340	80.761	18,800	348.57	--	12.3	36,500		20.8	268	34,200	< 4	0.464	59,608
124H	4/12/2006	7.31	31,000	4,093	716	560.00	118	51.463	7,565	437.15	--	< 25	13,263	< 25	20.1	< 1,250	17,927	< 250	0.428	22,775
124H	5/15/2009	6.9	E 50,000	5,700	1,030	764.00	146	53	9,170	< 800	0.62	31.3	19,500	E 0.06	20.4	648	21,900	< 100	0.390	31,812
124J	10/12/1989	7.44	44,480	4,659	1,190	410.00	202	60.053	9,420	266.56	--	65.4	18,000	0.06	20	268	28,500	1,279	0.405	29,869
124J	4/10/2006	6.96	11,950	1,480	347	149.00	56.9	26.255	2,321	214.06	--	8.2	4,129	< 5	19.1	< 250	5,461	380	0.346	7,284
124J	5/14/2009	7.2	E 17,900	1,900	463	180.00	64.8	27	2,740	257	0.368	10.7	6,150	0.41	19.8	137	10,100	289	0.344	9,930
124J	5/14/2009	--	--	--	E 517	--	--	--	--	--	--	--	--	--	21	--	E 8,600	--		--
124N	10/17/1989	7.49	22,557	2,960	790	240.00	154	51.983	6,500	221.45	--	< 0.1	12,000	0.9	22.8	177	17,800	< 4	0.532	20,156
124N	4/4/2006	7.2	12,640	1,386	344	128.00	57	28.603	2,447	268.2	--	8.56	4,217	< 5	20.3	< 250	6,679	< 50	0.334	7,549
124N	5/14/2009	7.5	E 10,300	970	236	92.10	45.9	26	1,850	208	0.202	5.52	3,340	0.82	20.8	93	5,840	< 24	0.324	5,815
124O	10/12/1989	7.51	21,375	9,529	1,970	1,120.00	44.51	6.285	1,410	257.53	--	15.9	7,840	1	22.5	1,220	265	509	0.367	13,948
124O	4/12/2006	7.21	8,510	2,531	514	303.00	16.7	8.901	1,029	337.09	--	< 5	2,574	< 5	19.6	555	< 300	2,060	0.302	5,439
124O	5/16/2009	6.8	E 44,500	11,000	2,100	1,370.00	53.7	24	5,660	< 800	0.123	28.4	17,200	--	17.3	1,330	638	1,400	0.387	28,240
124O-b	5/16/2009	8.5	E 6	M	0	0.04	< 0.06	E 0.1	E 0	< 8	--	< 0.02	0	< 0.08	< 0.2	E 0	< 2	< 4	0.043	6
124P	10/12/1989	7.7	40,603	8,122	945	1,400.00	123	36.21	7,500	245.23	--	< 0.1	16,300	0.2	22.2	1,450	1,840	< 4	0.401	28,041
124P	4/12/2006	7.51	41,600	7,979	1,041	1,307.00	149	53.402	10,963	205.86	--	< 25	20,126	< 25	18.8	1,518	9,769	< 250	0.484	35,374
124P	5/16/2009	7.9	E 13,600	1,200	156	189.00	34.3	28	2,180	290	0.079	7.19	4,370	0.52	19.3	439	4,960	< 32	0.321	7,574

Appendix I (cont.)

Field Parameters and Analytical Chemical Analysis of Water Samples from the Goose Lake Detailed Study Site

Analytical Chemical Data (2/2)

Site name	Sample date	pH (standard units)	Specific conductance (uS/cm)	Hardness (mg/L as CaCO3)	Calcium (mg/L as Ca)	Magnesium (mg/L as Mg)	Potassium (mg/L as K)	Sodium Absorption Ratio	Sodium (mg/L as Na)	Alkalinity (mg/L as CaCO3)	Iodide (mg/L as I)	Bromide (mg/L as Br)	Chloride (mg/L as Cl)	Flouride (mg/L as F)	Silica (mg/L as SiO2)	Sulfate (mg/L as SO4)	Boron (ug/L as B)	Iron (ug/L as Fe)	CI (Cl/S.C.)	Dissolved solids, computed (mg/L)
126A	4/20/1989	7.45	6,233	298	67	31.70	12	18.138	719	266.56		< 0.1	1,060	0.3	19.4	111	1,590	0	0.170	2,348
126A	5/14/2009	6.5	E 17,900	1,700	421	153.00	37.1	32	3,040	E 198	0.619	10.9	6,100	0.14	19.8	154	4,590	187	0.341	10,060
126B	4/20/1989	7.08	59,538	9,904	2,120	1,120.00	162	48.53	11,100	212.42	–	< 0.1	23,600	< 2	16.3	116	15,700	9	0.396	38,529
126B	4/5/2006	6.24	32,900	5,196	1,156	561.00	106	48.228	7,989	232.93	–	30.3	15,382	< 25	17.2	< 1,250	14,543	< 250	0.468	25,497
126B	5/14/2009	6.9	E 41,300	4,700	1,020	521.00	101	41	6,460	< 800	0.636	30	15,800	–	20	230	18,000	38	0.383	24,681
126C	10/12/1989	7.51	1,203	524	116	57.05	9.06	0.513	27	307.6	–	< 0.1	7	0.3	27.2	257	157	980	0.005	877
126C	5/15/2009	7.1	3,630	1,700	372	183.00	13.5	0.5	47	242	0.039	1.87	964	0.15	20.9	220	64	7,310	0.266	1,968
126C	5/15/2009	–	–	–	E 415	–	–	–	–	–	–	–	–	–	24	–	< 199	–		–
264A	10/13/1989	7.18	80,729	10,452	2,770	859.00	860	165.993	39,000	262.45	–	2.2	66,900	0.3	10.1	669	89,700	962	0.829	111,435
264A	9/15/2005	7.15	47,000	4,175	1,001	407.00	384	118.125	17,540	430.59	–	–	30,841	–	–	702	–	–	0.656	51,400
264A	5/13/2009	7	E 91,700	5,600	1,360	539.00	407	110	18,200	386	3.71	55.3	39,300	0.29	12.9	509	73,200	< 200	0.429	60,692
264A-r	5/13/2009	7	E 91,300	5,700	1,370	543.00	402	110	18,600	381	3.66	57.1	36,300	0.27	12.8	512	71,400	< 200	0.398	58,101
264A	5/13/2009	–	–	–	E 1,630	–	–	–	–	–	–	–	–	–	< 44	–	< 59,200	–		–
264B	4/20/1989	6.95	78,626	11,887	2,420	1,420.00	272	63.455	15,900	262.45	–	< 0.1	32,800	< 5	9.2	250	20,600	3	0.417	53,421
264B	9/15/2005	7.05	39,500	5,657	1,227	630.00	223	67.914	11,739	400.24	–	–	22,638	–	–	537	–	–	0.573	37,482
264B	5/15/2009	7.5	3,400	750	141	95.60	19.6	6.2	386	388	0.09	1.04	762	0.31	31.6	215	1,570	E 7	0.224	1,887
264D	4/20/1989	6.7	48,105	7,256	1,500	853.00	110	48.986	9,590	448.63	–	< 0.1	18,600	< 1	11.9	1,210	12,200	73	0.387	32,439
264D	9/15/2005	7.31	16,220	2,810	560	343.00	61.8	31.331	3,817	510.97	–	–	7,302	–	–	838	–	–	0.450	13,545
264D	5/15/2009	6.9	E 16,800	2,200	460	265.00	44.3	25	2,680	456	0.084	7.97	5,330	0.45	16	804	8,480	E 23	0.317	9,890
264J	5/1/1989																		0.201	
264J	5/12/2009	7.6	2,680	750	173	78.30	22.1	3.4	213	162	0.013	0.85	661	E 0.1	30.6	181	322	65	0.247	1,457
264K	5/1/1989																		0.322	
264K	5/12/2009	7.6	2,030	640	143	67.90	15.8	2.6	149	186	0.013	0.23	396	E 0.1	25	252	249	114	0.195	1,161
264M	5/1/1989																		0.157	
264M	9/17/2009	8.1	4,500	800	110	128.00	30.2	8.5	551	108	0.007	1.19	1,270	E 0.06	16.3	242	768	E 12	0.282	2,410
264M	9/17/2009	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–		–
264P	5/2/1989																		0.095	
264P	5/12/2009	8.6	2,160	1,200	280	117.00	43.6	0.7	59	334	0.021	0.16	98	E 0.07	30.8	842	355	28	0.045	1,671
264Q	10/13/1989	7.03	28,810	4,975	1,010	596.00	56.9	31.215	5,060	223.91	–	7.1	10,300	0.1	15.7	1,060	6,200	< 4	0.358	18,380
264Q	5/15/2009	7.3	E 12,500	2,100	419	260.00	27.1	17	1,790	318	0.079	5.74	3,940	0.1	16.5	496	3,990	6	0.315	7,149
264R	10/13/1989	7.67	2,182	947	178	122.00	19.15	1.414	100	350.21	–	0.2	195	0.1	22.9	538	156	< 4	0.089	1,603
264R	5/15/2009	7.2	3,460	1,200	225	165.00	23.5	2.5	203	364	0.016	0.71	732	E 0.09	21.3	412	139	510	0.212	2,001
264S	10/13/1989	7.5	8,800	2,323	406	318.00	30.93	8.631	956	287.88	–	2.8	2,810	0.1	30.6	182	1,210	7,840	0.319	4,910
264S	5/16/2009	7.3	6,290	1,500	232	220.00	23.4	7.7	680	325			1,910	E 0.1	25.5	241	619	8,590	0.304	3,528
264T	10/13/1989	7.7	761	364	81	39.44	5.86	0.456	20	263.27	–	< 0.1	12	0.35	23	130	114	1,354	0.016	634
264T	5/16/2009	7.3	4,650	1,800	167	339.00	20	3.5	344	258	0.017	0.55	1,030	0.19	21.6	860	437	6,720	0.222	2,938
264Y	5/12/2009	7.5	1,080	430	85	52.10	19.9	0.9	45	190	0.015	0.06	120	E 0.05	39	181	199	28	0.111	655
264Y	5/12/2009	–	–	–	E 99	–	–	–	–	–	–	–	–	–	51	–	190	–		–
307B	4/21/1989																		0.107	
307B	5/15/2009	8.6	E 66,600	4,500	489	785.00	263	150	23,000	< 800	0.291	30.6	4,050	E 0.05		45,300	5,380	< 140	0.061	74,403
307B	5/15/2009	–	–	–	521	–	–	–	–	–	–	–	–	–	< 42	–	5,040	–		–
RAB-02	9/15/2005	7.41	5,050	1,411	255	188.00	15.4	7.345	634	442.89	–	–	1,578	–	–	182	–	–	0.312	3,426
RAB-02	5/12/2009	7.2	3,970	1,200	213	151.00	11	4.2	326	416	0.027	1.55	975	0.11	22.8	161	236	E 8	0.246	2,111