Identifying Potential Land Use-derived Solute Sources to Stream Baseflow Using Ground Water Models and GIS

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Abstract

This paper presents an approach to examine potential relationships between land use-derived solutes and baseflow surface water quality using regional ground water and solute transport models linked to GIS. We demonstrate this approach by estimating the chloride fluxes from road salt through ground water into surface water in a regional coastal watershed in Michigan. The geologically parameterized model for this case study provides a good fit to measured hydraulic heads in the watershed and allows us to estimate spatially and temporally variable solute fluxes via ground water to streams and directly to the Great Lakes. The results demonstrate that there is a considerable legacy of land use influencing surface water quality at the study site. The simulated chloride concentrations are similar to measured surface water chloride concentrations throughout most of the watershed, except in regions where other sources for chloride (e.g., septic systems, locations of oil brine fields and high-density human populations) likely exist. Impacts of other land use related solutes on baseflow surface water quality could also be explored using this approach. As a result, watershed managers could be provided with better information about the potential impacts of developments and associated surface-applied solutes on future surface water quality.

Introduction

Ground water chemistry is reflective of time-weighted averages of anthropogenic inputs originating from spatial and temporal patterns of land use (Modica et al., 1997; 1998). These spatially and temporally varying impacts of land use on surface water quality are complex. Thus, ground water flow and solute transport models, linked with spatial data management and analysis tools such as geographic information systems (GIS), may be the best approach to evaluate these impacts. The principal objective of this paper is to describe an approach that links land use-derived solutes to spatial variability in surface water quality through process-based ground water flow and solute transport models and GIS. This approach can be used to assess the impact of different baseflow solute contributions to surface water chemistry.

Numerous field studies have explored how changes in land use affect ground water dynamics and surface water quality. Pucci and Pope (1995) identified significant differences in ground water systems between developed and undeveloped areas in the northern Coastal Plain of New Jersey. Forest clearing has also been shown to have a substantial effect on ground water recharge (e.g., Thorburn et al., 1991). Researchers have also found a correlation between land use patterns and stream chemistry. For example, Flintrop et al. (1996) showed that land use patterns affect the major ion and nutrient geochemistry in selected tributaries of the Rhine River. Many groups have studied the influence of road salt on ground water quality (e.g., D'Itri, 1992; Jones and Sroka, 1997).

Numerical models have also been developed to examine ground water and streamflow response to a variety of land use changes (e.g., Hookey, 1987; Arnold and Allen 1996; Salama et al., 1999). However, only a few studies have coupled ground water flow and solute transport models to quantify potential influences of altered landscapes on solute fluxes at a regional scale (e.g., Dunn and Mackay, 1995; Dunn et al., 1996). Ground water models are commonly used to describe regional flow patterns (Anderson and Woessner, 1992), while solute transport models are less commonly used at this scale (e.g., Martin and Frind, 1998).

In this paper, we quantitatively explore the influences of land use on water quality using regional (3300 km^2) high-resolution ground water flow and solute transport models. Land use is integrated into our models by relating a specific land use (roads) from GIS databases to an associated chemical input to ground water (chloride from road salt). The approach we present here allows us to explore the long-term spatial dynamics between land use and surface water quality, and the implications for watershed management.

Study Area

Michigan's Grand Traverse Bay Watershed (GTBW) is an ideal location to explore the potential impacts of land use on water quality of the Great Lakes. The GTBW is located in the northwestern portion of Michigan’s Lower Peninsula (Figure 1). Its shoreline is one of the longest in the Great Lakes, thus its potential water quality impacts are significant. Land use in the GTBW is predominantly forest (49%) and agriculture (20%). Much of the forested portions of this watershed are managed within the Pere Marquette State Forest, which encompasses most of the upper reaches of the Boardman River tributaries. The other main land uses are shrub/grasslands (15%), water (9%), urban (6%), and wetlands (1%).

Traverse City is the largest city in the watershed with a resident population of approximately 100,000. From 1980 to 1998, the watershed increased its resident population by over 100%, making it one of the fastest growing areas in the Midwest

United States (Vesterby and Heimlich, 1991). In addition, temporary seasonal and transient recreational population in the summer often exceeds 500,000. The work discussed in this paper is part of a larger study that examines how land use changes affect water quality. The results should provide better information for managing the region's superior water quality.

The Boardman River is the main tributary draining the GTBW. This watershed contains over 100 lakes, including Elk Lake (Figure 1), a large water body whose sediment record shows the influence of land use change (Simpson et al., 2000). The region receives an average annual rainfall of approximately 105 cm, of which approximately 40 cm is recharged to ground water (Holtschlag, 1997).

The regional geology results from Quaternary glacial advances and retreats, in which glaciers carved deep valleys into the shale and limestone bedrock and deposited sediment accumulations as thick as 365 meters. Sediment characteristics vary widely across the watershed, in some areas changing from a thick lacustrine clay to a coarse grained moraine within 100 meters. The regional geology has been described in a variety of reports and papers (e.g., Leverett and Taylor, 1915; Martin, 1957; Eschman et al., 1973). Martin (1957) stated that most of the glacial sediments in the southern portion of the watershed are coarse-grained moraine and outwash deposits. The Port Huron moraine system is stratified and has characteristics of an outwash setting (Blewett and Winters, 1995), despite its earlier classification as a moraine. Devonian limestone and shale outcrop along the east shore of Grand Traverse Bay (Figure 1), and Mississippian shale outcrops northeast of Torch Lake (Kesling, 1974).

The impacts of land use on water quality have been previously documented in the GTBW. Rajagopal (1978) examined land use impacts on ground water quality near Old Mission Peninsula and Traverse City (Figure 1) and found high levels of nitrate in wells surrounded by cherry orchards. A 1984 to 1986 USGS study in Grand Traverse County (Cummings et al., 1990) of 34 water wells and 24 surface water sites showed a possible correlation between nitrogen input (fertilizer, septic, etc.) and ground water nitrate concentrations. We studied the application of road salt and its transport through groundwater and to streams in the watershed. Chloride is an excellent solute to examine with our groundwater flow and solute transport models because: 1) there is no natural source of this thus it is a surrogate for human activity; and, 2) based on our analysis of stream water chemistry (Woodhams et al., 1998) in the area, chloride from road salt appears to be a significant source of chloride.

Modeling Approach

Data Collection and Analysis

One of the challenges to regional scale ground water flow modeling is compiling the necessary data for an effectively parameterized model. The approach that we take in this paper is to parameterize our ground water models using a small number of zones based on the shallow aquifer geology. We use GIS, specifically ARC/INFO (ESRI, 1998a) and ArcView (ESRI, 1998b), to manage, manipulate, and analyze the spatial data for our ground water flow and solute transport models.

A large geologic database was compiled for the GTBW region from oil/gas and water-well drilling logs and regional geologic maps. A digitized one-kilometer surficial geologic map for the region (Farrand and Bell, 1984) was used to delineate the unconfined aquifer into six zones of roughly equivalent flow properties (Figure 2). The elevation and lithology of the bedrock surface was derived from: a) 6,700 oil and gas well logs from the Michigan Department of Environmental Quality, b) a subset of the approximately 5,000 residential water-well logs from the Michigan Department of Natural Resources that were deep enough to reach bedrock, and c) the few bedrock outcrops in the northeastern portion of the watershed. Any well logs that had recorded elevations that differed from the DEM at the recorded location by more than one meter or had inadequate or unreasonable geologic characteristics were removed from the database. The bedrock elevation values were interpolated to the ground water model grid using ordinary kriging with an exponential variogram, a variance of 1300, and an isotropic range of 3500 meters. The interpolated bedrock elevations were then used to define the bottom of the glacial aquifer for the ground water model. The glacial sediment thickness was calculated by subtracting the bedrock elevation grid from the 30-meter DEM (Figure 3). Notable features in the maps depicted in Figure 3 include bedrock valleys in the southeastern portion of the watershed and the deeply incised valley of the Boardman River in the lower central portion of Figure 3b.

A database of transportation and hydrologic features for the GTBW was obtained from the Michigan Department of Natural Resources (Michigan Resource Information System) MiRIS Office. MiRIS is a statewide land use database developed from 1978 1:24,000 aerial photography. MiRIS line files of the transportation network and locations of rivers, lakes, and the bay boundary were integrated into our GIS database to locate model boundaries and potential solute sources. Human population densities at the block level for the GTBW were obtained from the 1990 Topologically Integrated Geographic Encoding and Referencing system (TIGER) database developed by the U.S. Census Bureau.

Development of ground water models

Our conceptual model for the GTBW was developed using available hydrogeologic information. Based on the geologic maps and well log data, we chose to develop a two-layer model that accounts for flow and transport through the glacial sediments. We did not have the computational resources necessary to construct a model with more than two layers at the time of development. The bedrock underlying the glacial sediments is mostly low-conductivity shale, which limits vertical movement of water in or out of the shallow aquifer, although a small amount of higher conductivity limestone exists in the northern areas of the watershed (Kelly, 1968). While the limestone could have been simulated as a conductive layer, the extent of this region was small and thus its influence on the model was assumed to be negligible.

We developed a three-dimensional ground water flow model of the GTBW with 100-meter by 100-meter finite difference grid cells using MODFLOW (Macdonald and Harbaugh, 1988). This grid cell resolution was chosen to adequately describe the elevations and geometry of the rivers and lakes (more than 34,000 river/lake cells), and to allow for accurate solute transport simulations.   The Groundwater Modeling System (GMS) preprocessor (BYU, 1994) allowed us to convert GIS-based data into model input files. Incorporating GIS databases into the watershed-scale model facilitated our ability to examine influences of land use on ground water flow and solute transport at this resolution.

The hydraulic conductivity values for the region are parameterized into zones that are concordant with the mapped glacial units (Figure 2). Both layers have equal hydraulic conductivity values except in areas where the high conductivity lacustrine sand and gravel units overlie low conductivity clays. Hydraulic conductivity values for the geologically parameterized zones (Figure 2) are between 10% and 30% of pump test values conducted in the lacustrine sand and gravel, the glacial outwash, and the end moraines (Cummings et al., 1990). For zones with no pump test data, hydraulic conductivity values were selected relative to the materials with estimates. For example, conductivity values in fine-grained till are likely to be lower than coarse-grained till, which in turn are likely to have lower values than the sand and gravel outwash deposits. This is a relatively simple method to incorporate regional geology into hydrologic models, although other parameterization methods may be more appropriate if higher resolution hydraulic conductivity data were available. Unfortunately, the well logs for the region contain limited information about aquifer properties and thus were not used to parameterize hydraulic conductivity for the model.

The hydrologic boundaries used in our model are shown in Figure 4. Constant head boundaries are used along the perimeter of Grand Traverse Bay, Lake Michigan, and Lake Charlevoix (Figure 1), with the head values set to known lake elevations. The remaining boundaries are significant surface water flow divides, which are represented as no-flow boundaries. The northeast and northwest boundaries of the model were extended beyond the surface water divides because of the uncertainty in the location of these divides, as discussed later in the "Delineating Ground water Flow Divides" section.

Effective ground water recharge for the model region is 40 cm/yr (Holtschlag, 1997) (see Study Area Section). Annual precipitation varies by only a few centimeters across the watershed, but soil type and land use types likely result in spatially variable recharge. For example, in areas where the soils are predominantly sand, recharge would be higher than in areas with clay rich soils (e.g., till). Since the necessary soil data were not digitized for this region at the time of this model development, we used a single recharge value of 40 cm/yr for the entire region.

Locations of river cells were developed for the model based on topography because inconsistencies existed between the regional MiRIS river coverage and the DEM. These inconsistencies included incorrectly coded lines, such as roads, and aberrant river gradients obtained by overlaying the river coverage with the DEM. For example, in several cases river elevations increased, and then decreased downstream, which is clearly unrealistic. This oscillatory effect was likely caused by the spatial averaging of the DEM and slight errors in the locations of the digitized MiRIS lines. To address these problems, ARC/INFO GRID was used to delineate drainage networks based on a DEM with 30 m by 30 m cells. First, the FLOWDIRECTION function in ARC/INFO GRID was used to calculate the direction water would flow on the land surface based on this high resolution DEM, and then the FLOWACCUMULATION function in ARC/INFO GRID was used to calculate the number of cells flowing into each grid cell. Cells with a high-calculated flow accumulation are areas of concentrated flow and can thus be used to identify likely stream channel locations. A threshold of 400 cells was placed on this process to define river reaches that approximated the extent of the MiRIS river coverage. In addition, we removed river cells that extended significantly beyond the ends of this coverage, because drainage patterns delineated using this technique can generate river cells that exist only during extreme high-flow events. The modified drainage networks were then resampled using the GIS to a 100 m by 100 m model grid using the lowest 30-m elevation within each 100-m cell as the river elevation.

Our solute transport model uses simulated hydraulic head values from our ground water flow model, and thus uses the same flow parameters. The solute transport model, based on MT3D (Zheng, 1992), was developed by assigning a constant chloride concentration of 70 mg/L to the recharge applied to each cell containing a major road. We discuss the choice of this concentration later in the *Simulating a Land Use Water Quality Scenario* section. In this 50-year simulation, we used chloride as an analog for road salt, which is primarily sodium chloride but can include small percentages of magnesium chloride or potassium chloride. Since chloride is conservative, no retardation factors or reactions were simulated. Sources for chloride to the environment are from: a) the dissolution of salt applied to roads in the winter months, b) brine application to dirt roads in some areas for dust suppression in summer months, c) septic fields, and d) brines from oil and gas exploration. Local county road commissions provided the locations of county and state roads with applied salt. Road salt is only applied in the winter months, but we approximate chloride concentrations by simulating an average annual input. If transient salt application and recharge data were available for a region, this could be incorporated into this approach to more accurately simulate transient chloride fluxes into surface water.

We used our model to see if simulated concentrations based on a single chloride source (road salt) to the region's baseflow water quality could describe observed chloride concentrations, and if outliers to these data were located in regions with alternative sources. In addition, this approach allowed us to quantify the predictive ability of our model, by comparing simulated chloride concentrations to measured stream water concentrations. Stream sampling was performed over a three-day period during low flow, when the primary input to streams is baseflow from ground water. These samples were collected from 80 stream sites (Figure 1) throughout the watershed to characterize spatial variability of geochemical parameters such as chloride.

Measurements of stream solute concentrations were compared to estimates from our flow and solute transport models using flow-weighted averages of simulated inputs to river cells upstream of each sample site. In the simulations, chloride concentrations were saved every five days for cells along the shoreline of Grand Traverse Bay and river cells, while chloride concentrations were saved every ten years for the entire watershed. At each stream node the flux (L/day) of water discharged from the aquifer and the concentration of chloride (mg/L) were used to determine a total mass flux (mg/day) of chloride at that stream node. Both the mass flux of chloride and the flux of water were summed along stream paths to our sampling locations (Figure 1). Simulated chloride concentrations (mg/L) at each stream sample site were then calculated by dividing the total mass flux of chloride (mg/day) into river cells upgradient of a site by the total streamflow (L/day) calculated at that site. Since we are comparing our simulated concentrations to observed concentrations taken at low-flow hydrologic conditions, we have made the reasonable assumption that ground water flow is the primary input.

Results and Discussion

Model Calibration

Simulated hydraulic heads in the GTBW vary from 177 meters at the Grand Traverse Bay boundary to over 350 meters in the eastern high topography areas (Figure 4). Significant ground water flow divides exist on both the northeastern and northwestern boundaries of Grand Traverse Bay. The Boardman River southeast of Traverse City (Figure 1) is a dominant ground water discharge area due to the ground water gradients in this region (Figure 4).

The hydraulic conductivity values for the six geologically parameterized zones in Figure 2 were adjusted by 10-30% of pump test values from Cummings et al. (1990) to minimize the squared residuals between observed and simulated heads (Figure 5). The effective recharge value for our simulation was obtained from (Holtschlag, 1997), and this value was not adjusted during model calibration because recharge estimates have much lower uncertainty than conductivity estimates and both should not be simultaneously adjusted. Several head datasets are available for the region, with monthly readings between January 1985 and January 1986, and another set in August 1998. The steady state calibration provides a good match between simulated and observed heads (Figure 5) for both 1985 and 1998. The R-squared linear correlation coefficient was of 0.99 for 1998 (n=19), and 0.98 for September 1985 (n=10). The simulated heads also match the trend in a large regional database of heads measured during well installations with a R-squared of 0.94 (n=3604). The slope of the linear regression was 0.95 for this large database and 1.0 for the USGS wells.

We did not consider annual and long-term temporal variability in recharge because it is beyond the scope of this paper. The values that were used, however, are representative long-term averages for effective ground water recharge in the region (Holtschlag, 1997). We plan to explore the spatial and temporal variability in recharge and their influence on water table elevations and solute concentrations in a future paper.

Direct Ground water Flux Estimates

Ground water fluxes were estimated into Grand Traverse Bay by summing the simulated fluxes into the constant head boundary cells that represent the border of the bay (Figure 4), using an approach similar to that of Krabbenhoft et al. (1990). This approach provided an estimated direct ground water discharge of 0.23 m^3/s. Our model also indicates that roughly 7% of the discharge to the bay is directly from ground water while the remaining 93% is from stream discharge, which is also derived from ground water discharge during low flow. Therefore, ground water processes have a major role in controlling overall hydrology within this regional watershed.

Delineating Ground water Flow Divides

If significant topography exists and the bottom of the aquifer is relatively flat, ground water divides can be estimated from surface water divides and used as a proxy for no-flow boundaries in ground water models. However, when topography is flat or the aquifer bottom is sloping, such estimates may not be valid, as is the case for the northwestern and northeastern area of the watershed. An alternative method of defining these divides, which could be used in any topographic setting, is to predict their locations based on flow models that use true hydrologic boundaries. In our model, the northeast and northwest boundaries were extended beyond the surface water divides because the aquifer bottom was found to have a significant slope (Figure 3b). Since there is little topography in these areas and the bottom of the aquifer has a significant slope, we allowed the model to delineate these ground water divides.

The simplest technique to develop ground water divides using the above approach is to plot simulated ground water flow vectors and digitize the boundary between regions that flow into the watershed and regions that flow out of the watershed. An alternative, which was not used in this study, is to import the simulated hydraulic heads into ARC/INFO and calculate flow directions and flow accumulations, with the approach we used to delineate our river network. This method is similar to delineating watershed boundaries for surface water catchments (Hornberger et al., 1998). We found that in the northwestern portion of the watershed, the surface water divide deviates significantly from the ground water divide.

Simulating a Land Use Water Quality Impact Scenario

Our coupled ground water flow and solute transport model was developed to infer the potential impacts of land use-related solutes on surface water quality. The scenario we chose to examine illustrates the potential influence of chloride from road salting practices. For example, additional sources of chloride in the GTBW region include septic systems and the improper disposal of subsurface brines brought to the surface by oil and gas wells.

We assessed chloride contributions from road salt to baseflow surface water by simulating chloride transport through ground water to streams and by comparing simulated versus observed values, as discussed in the methods section. Figure 6a shows two primary clusters in the simulated versus observed chloride concentrations in rivers. The lower cluster in Figure 6a (blue triangles) represents calculated chloride concentrations that are similar to measured concentrations, while the upper cluster (red circles) represents calculated concentrations that are underestimates of the measured chloride concentrations in streams.

A spatial analysis of residuals between simulated and observed chloride concentrations (Figure 6b) shows revealing trends associated with potential solute sources. The tan striped horizontal lines indicate an area where there is risk for ground water contamination by the improper disposal or leakage of subsurface brines and the green triangles indicate sites with documented ground water contamination by brines (Fryer, 1982; Skillings, 1982). The magenta region refers to the Traverse City region on municipal sewer service. Human population densities greater than 30 people per square kilometer outside this municipal sewer service area are mapped in yellow. The sites that showed similar calculated and observed concentrations in Figure 6a (blue triangles) are generally in regions with relatively low population densities. This potentially implies that road salt application is a significant chloride source in these areas. The sites that are *underestimated* in Figure 6a (red circles), using roads as the only chloride source, are all either in or downstream from the noted oil/gas trend region (Figure 6b, horizontal tan stripes), or the portion of the watershed that is not on municipal sewer service and has high population densities. This implies that other sources, such as brine or septic systems, may be contributors along with road salt to chloride concentrations in baseflow in these regions.

Our initial applied chloride concentration for the model was chosen to be 100 mg/L, which is a value that appears to be reasonable for chloride inputs from road salt in Michigan (MDOT 1993). We then adjusted this value to 70 mg/L in order to calibrate the transport model to the lower trend in Figure 6a. The range of values from 70 to 100 mg/L are reasonable for ground water chloride concentrations in sources areas near where road salt has been applied, based on published values for Michigan (Gales and VanderMeulen, 1992). To adjust the initial value, we calculated the ratio of simulated low trend values to measured values for the 100 mg/L simulation, then multiplied this ratio by the initial value to provide our final input recharge concentration of 70 mg/L. This provided a reasonable approximation of the lower trend. We could have calibrated the model to the upper trend, but there was no logical reason to do this since we could not explain the resulting spatial variability of overestimated values.

Figure 7 shows maps of predicted chloride concentrations for 10 and 50 years after road salt application begins. Areas with high simulated chloride concentrations are mostly within urban regions, such as Traverse City, due to the high density of salted roads in the simulation. The change from the 10-year to the 50-year concentration profile shows the expected increase in chloride concentrations across the watershed through time (Figure 7). Areas with the highest simulated concentrations are either in or downgradient of regions with high density of roads with applied road salt. In contrast, regions with the lowest simulated concentrations are in areas where no roads are present, such as the Pere Marquette State Forest (Figure 1). Our model also shows that transport distances can exceed 10 kilometers in a 50-year period (Figure 7).

The temporally varying chloride flux into Grand Traverse Bay from direct discharge and due to ground water discharge to streams is illustrated in Figure 8. This plot shows the anticipated time lag between road salt application and the expected asymptotic chloride flux of roughly 3640 kg/day to the bay due to a constant chloride simulated input of 70 mg/L. This asymptotic value is equal to the steady state input flux applied to the model, which is calculated by summing the recharge rate to road cells multiplied by the chloride concentration applied to these cells (70 mg/L). Note that this simulation begins to approach the asymptotic value after 50 years, which implies that we would expect nearly steady state chloride flux to the bay after 50 years of road salt application. This assumes that road salting operations remained constant and the amount of roads under application did not increase. It is interesting to note, however, that this region has undergone road salt application for nearly 50 years, thus the model suggests that we may now be approaching a steady state concentration due to past road salting practices. The results also illustrate the time it takes for land use to impact surface water quality. This effect, which we call a *land use legacy*, is an important concept to consider, as watershed management should consider not only the location of sources but also the temporal lag between the introduction of solutes into a watershed and the potential impacts to the ecosystem.

Simpler approaches exist for estimating total solute fluxes from a watershed, such as the export coefficient model (Johnes, 1996; Mattikalli and Richards, 1996). This model is capable of predicting nutrient loading as a function of the spatial distribution of nutrient sources within a catchment draining to a measurement site. Although this is a reasonable approach to estimate short-term loadings from overland flow, it does not describe the processes that transport nutrients through ground water. As a result, this approach does not incorporate the time lag for such transport that is associated with ground water inputs to stream baseflow, which may account for a significant percent of solute loadings throughout an annual period.

Conclusions

Ground water flow and transport processes need to be described to effectively interpret the influence of land use on surface water quality. We have demonstrated that regional scale ground water flow and solute transport models can help identify the impact of land use related solute sources in a watershed. The approach that we used is to: 1) select a potential solute source to assess if its contribution to a region's baseflow is significant (e.g., chloride from road salt), 2) develop a regional ground water model to represent fluid flux rates through the region of interest, 3) simulate steady state input of the chosen solute and the resulting transport through ground water to discharge points such as rivers, 4) calculate the flux weighted average concentrations into stream reaches upstream of sample sites, and 5) compare these estimates to values obtained during base flow surface water measurements. We used this approach to estimate the influence of chloride from road salt in one watershed of the Great Lakes.

The presented case study explored the transport of chloride from roads through ground water into streams using measured chemistry from low-flow stream samples as an analog for ground water inputs. This approach is very powerful for three reasons. First, it provides measurements of ground water inputs across an entire watershed with a small number of easily collected samples. Second, we were able to use our measured stream chloride concentrations to calibrate the regional transport model. Third, deviations from expected concentrations provide valuable information about land use influences on water quality. In our case study, chloride from salt applied to roads does not appear to be the sole contributor of chloride to streams, although it appears to be significant source. Chloride from anthropogenic sources (such as septic systems) and subsurface brines from oil and gas exploration likely contribute to the chloride concentrations in streams of the GTBW.

The model was also used to constrain watershed boundaries, which has implications for the magnitudes of estimated solute flux to the GTBW. Surface water divides and ground water divides often mimic each other in areas of high topography, but in the GTBW, there are areas with relatively low topography where these divides differ. If solute flux predictions use the incorrect divides, the estimates would be biased by the amount of solute applied to the region between the divides. Therefore, proper development of ground water divides ultimately leads to more accurate water and solute budgets for the watershed. This approach can also be used to decipher ground water source areas for smaller regions of a watershed.

The use of GIS was beneficial for model development and analysis of model output, because large amounts of spatial data were required to execute the model. Data from a variety of sources were in different geographic projections and GIS facilitated conversion of databases into a standard projection. We also used GIS to examine spatial data for potential errors prior to model input and to determine stream locations from digital elevation model data. GIS also aided in the interpretation of model results. For example, overlays of model output with road networks, human population densities, locations of stream sample sites, and noted locations of ground water contamination from brines allowed us to interpret potential sources of chloride.

The approach developed in this paper has significant implications for using ground water models in watershed management. Our ground water flow and solute transport models appear to be able to predict the impact of road salt on resulting surface water quality. In our case study, we appear to correctly identify regions of a watershed that have been influenced by road salt using a recharge source of chloride to cells with identified roads that are deiced. In addition, our approach is able to estimate the time it takes for a land use to influence water quality, which we call a land use legacy. Our simulation also showed that chloride flux into the bay reached an asymptotic concentration after about 50 years, at which time the spatial pattern of chloride across the watershed remained relatively stable. As a result, this information can be used to assess the spatial pattern of environmental degradation once this asymptote is reached. In our case, the 50-year estimate likely provides a reasonable representation for the current situation, because we are approaching the asymptotic value and road salting began in the Grand Traverse region about 50 years before present.

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Figure 1. The Grand Traverse Bay Watershed, the study area, is located in the northeastern portion of the Lower Peninsula of Michigan. The area that is modeled is shown in gray. Old Mission Peninsula divides Grand Traverse Bay into an east and west bay.

Figure 2. Map of the region's surficial geology digitized to a one-kilometer grid from the map by Farrand and Bell (1982), and associated hydraulic conductivity values used in our ground water flow model.

Figure 3. The surface elevation (a) compiled from over forty USGS 7.5 minute DEMs at 30 m resolution was combined with bedrock elevation data, (b) to estimate the glacial sediment thickness map, (c) for the watershed.

Figure 4. Simulated hydraulic heads range from 382 masl to 177 masl, which is the elevation of the Grand Traverse Bay. Boundaries for the flow model are also shown. The regions labeled as "no data" are where the unconfined aquifer is not present or very thin due to shallow bedrock.

Figure 5. Plot of simulated versus observed hydraulic heads for the USGS wells located in Grand Traverse County.

Figure 6. (a) Spatial plot of surface water sampling locations classified into two categories, 1) locations that matched the measured chloride trend in (b), and 2) locations that did not match this measured trend. (b) Simulated vs. observed concentrations at stream sampling locations after 50 years of salt application to roads.

Figure 7. Spatial maps of simulated chloride concentrations after (a) 10 years of chloride application, and (b) after 50-years of chloride application to roads.

Figure 8. The simulated cumulative flux of chloride to Grand Traverse Bay over the period of the 50-year simulation demonstrating the potential legacy of road salting on water quality.