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 geographic information systems (GIS). We demonstrate this approach by estimating chloride concentrations in surface water due to road salt transport through ground water in a large coastal watershed in Michigan. The geologically parameterized model for this study provides a good fit to measured hydraulic heads in the watershed and offers a method to estimate spatially and temporally variable solute fluxes via ground water to streams and 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 produced with salted roads as the only chloride source are similar to measured surface water chloride concentrations throughout most of the watershed, except in regions where other sources for chloride (e.g., high-density septic systems, locations of oil brine fields) 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 quantitative 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), provide a powerful 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 (Thorburn et al. 1991). Researchers have also found 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 demonstrated influences of road salt on ground water quality (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 (Hooke 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 concentrations and fluxes at a regional scale (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 (Martin and Frid 1998).

In this paper, we quantitatively explore the influences of land use on water quality using regional (3300 km²) 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 potential 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 40,000. From 1980 to 1998, the watershed increased its resident population by more than 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 GTBW contains more than 100 lakes and an extensive river system (Figure 1). The Boardman River is the main tributary draining the GTBW. The region receives an average annual rainfall of
Figure 2. Map of the region's surficial geology digitized to a 1 km grid from the map by Farrand and Bell (1982), and associated hydraulic conductivity values used in our ground water flow model. Salted roads are shown in red.

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, during which glaciers carved deep valleys into the shale and limestone bedrock and deposited sediment accumulations as thick as 365 m. Sediment characteristics vary widely across the watershed, in some areas changing from a thick lacustrine clay to a coarse-grained moraine within 100 m. The regional geology has been described in a variety of reports and papers (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 et al. 1974).

The impacts of land use on water quality have been previously documented in the GBW. 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 U.S. Geological Survey (USGS) study of 34 water wells and 24 surface water sites in Grand Traverse County (Cummings et al. 1990) 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 ground water and to streams in the watershed. Chloride is an excellent solute to examine with our flow and transport models because (1) there is little or no natural source of this solute in this watershed, thus it is a surrogate for human activity, and (2) based on our analysis of stream water chemistry in the area (Woodhams et al.
Figure 3. The surface elevation (a) compiled from more than 40 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.

1998), road salt appears to be the dominant source of chloride across the watershed with other inputs from septic systems and oil/gas well brines.

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 ARCTINFO (ESRI 1998a) and ArcView (ESRI 1998b), to manage, manipulate, and analyze the spatial data for our flow and 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 1 km surficial geologic map for the region (Farrand and Bell 1982) 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 (1) roughly 6700 oil and gas well logs from the Michigan Department of Environmental Quality, (2) a subset of the approximately 5000 residential water well logs from the Michigan Department of Natural Resources that were deep enough to reach bedrock, and (3) the few bedrock outcrops in the northeastern portion of the watershed. Any well logs that had recorded elevations that differed from the Digital Elevation Model (DEM) at the recorded location by more than 1 m 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 m. 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 m 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 3a.

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 shal-
low 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 m by 100 m 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 chosen based on the relative nature of the materials (Freeze and Cherry 1979). 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 (Holltschlag 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 river 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 FLOWACUMULATION 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 road with recorded salt application. Local county road commissions provided the locations of county and state roads with applied salt. We discuss the choice of this concentration later in the “Simulating a Land Use Water Quality Impact Scenario” section. In this 50-year simulation, we used chloride as an analog for road salt, which is primarily sodium chloride but can include percentages of magnesium chloride, calcium chloride, and/or potassium chloride. Since chloride is conservative, no retardation factors or reactions were simulated. Sources for chloride to the environment include (1) the dissolution of salt applied to roads in the winter months, (2) brine application to dirt roads in some areas for dust suppression in summer months, (3) septic fields, and (4) brines from oil and gas exploration. 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 are 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 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 chloride 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 river cells and cells along the shoreline of Grand Traverse Bay, while chloride concentrations were saved every 10 years for model cells throughout the 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 approximate 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.
Figure 6. (a) Simulated vs. observed concentrations at stream sampling locations after 50 years of road salt application. (b) Spatial plot of surface water sampling locations classified into two categories: (1) locations that matched the measured chloride trend in (a), and (2) locations that did not match this measured trend.
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 m at the Grand Traverse Bay boundary to more than 350 m in the eastern high topography areas (Figure 4). Significant ground water flow divides exist in both the northeastern and northwestern portions of the GTBW. 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% to 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 our recharge estimates have lower uncertainty than our conductivity estimates and both should not be simultaneously adjusted. This was true in this case because a regional value of recharge had previously been estimated (Holtschlag 1997) but only limited hydraulic conductivity data were available for the GTBW, and hydraulic conductivity varies by several orders of magnitude across the region. Several head data sets 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 1985 and 1998. The R-squared linear correlation coefficient was 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 an 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).

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 m³/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.

Delineating Ground Water Flow Divides
If significant topography exists and the bottom of an unconfined aquifer is relatively flat, ground water divides may be roughly 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
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 (Figure 6b). 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 source 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 in this simulation. 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 downstream 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 km in a 50-year period (Figure 7).

The simulated temporally varying chloride flux into Grand Traverse Bay from both direct discharge to the bay and ground water discharge to streams is illustrated in Figure 8. The plot illustrates the time it would take for the watershed to approach an asymptotic chloride flux of roughly 3640 kg/day to the bay, making a simplistic assumption that constant chloride input of 70 mg/L occurred during the entire the simulation period. 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). Our sim-

**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 explore illustrates the possible influence of chloride from road salting practices, given an assumed constant chloride loading over time. 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, which could also be explored using this approach.

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 "Developing Ground Water Models" 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 simulated concentrations that are underestimates of the measured chloride concentrations in streams.

A spatial analysis of residuals between simulated and observed
ulation begins to approach the asymptotic value after 50 years, which implies that we would expect nearly steady-state chloride flux to the bay if there had been 50 years of road salt application at the same rate. However, road salting operations have not remained constant and the amount of roads under application have increased over this time period. If chloride loadings increased throughout the simulation period, the resulting concentrations in streams would also logically increase, and thus would not reach a steady state.

Although this region has undergone road salt application for nearly 50 years, the rate and area of application has increased over this time period due to rapid development. As a result, the model with steady-state inputs provides only a preliminary estimate of chloride concentrations in streams and is not intended to predict solute loadings. Temporal chloride application data would be needed to provide such estimates. In this case, our steady-state simulation simply provides a preliminary estimate of the delay for applied road salt to fully impact surface water quality from the existing salted road network. This effect, which we call a land use legacy, is an important concept to consider, as watershed management should take into account not only the location of sources but also the temporal lag between the introduction of solutes into a watershed and the potential future long-term impacts to the ecosystem.

Simpler approaches exist for estimating total solute fluxes from a watershed, such as the export coefficient model (Johnes 1996; Mattikall and Richards 1996). This approach is capable of predicting solute loading as a function of the spatial distribution of solute 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 solutes through ground water. As a result, this approach does not incorporate the time lag associated with the transport of 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 solute 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 flow 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 recharge points such as rivers; (4) calculate the flux-weighted average of simulated concentrations into stream reaches upstream of sample sites; and (5) compare these estimates to values obtained during baseflow surface water measurements. We used this approach to estimate the influence of chloride from road salt on stream concentrations in a Great Lakes watershed.

The present 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 is a powerful approach 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 study, chloride from salt applied to roads does not appear to be the sole contributor of chloride to streams, although it appears to be a 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 simulated 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 surface water and ground water 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 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 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 delineated. 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.

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