Modeling the Effect of Landfill Leachate on Regional Groundwater Chemistry, Easthampton MA

Nicholas Newcomb
Hampshire College
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Chair:
Christina Cianfrani

Committee Members:
Steve Roof
Robert M. Newton

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Abstract:

This study analyzed the effect of landfill leachate on regional concentrations of dissolved oxygen a potential metal mobility. Three municipal landfills exist over the primary recharge area of the Barnes Aquifer which provides several towns in Hampshire and Hampden counties, Massachusetts with drinking water. Existing water chemistry data from wells in the vicinity of the Northampton landfill provide preliminary evidence suggesting that landfill leachate may produce reducing conditions capable of mobilizing iron, manganese, and arsenic constituents from aquifer sediments. The purpose of this study was to construct a groundwater flow model using MODFLOW in order to quantify the extent and concentration of dissolved organic carbon (DOC) present in leachate plumes generated from three landfills and assess the potential effects on municipal water sources. The study relied on leachate production results yielded from an EPA Hydrologic Evaluation of Landfill Performance (HELP) model coupled with a 3-dimensional reactive transport package (RT3DV2.5). The model was able to accurately predict groundwater head under steady state conditions. Contaminant transport results indicate that contaminant plume containing high biological oxygen demand (BOD) produces a plume of depleted dissolved oxygen (DO) which is transported to a high yield municipal well. Low background DO concentrations at this site suggest that small changes in geochemistry could have large impacts on iron, arsenic, and manganese concentrations.

Introduction

The Northampton Landfill is located in the primary recharge area of the Barnes Aquifer (Figure 1-1). Groundwater resources from the aquifer provide several towns in Hampshire and Hampden counties, Massachusetts with drinking water. Recent concerns regarding the expansion of the landfill have raised significant questions regarding water quality issues associated with leachate contamination of the aquifer (Kraft, 2007). Particular emphasis has been placed on the potential for contaminants in leachate to impact the Maloney Well that serves municipal water needs for the town of Easthampton. The Northampton Landfill is constructed in the wellhead protection, or Zone II, of the Maloney Well. Though significant study has been targeted towards assessing contaminant risks attributed to landfill expansion (Dufresne-Henry, 2005), there has been little effort to address the effect leachate from the current landfill operation has on groundwater chemistry particularly in regard to iron and arsenic reduction and mobilization. In addition, almost no information exists concerning the cumulative effects of leachate from both the Northampton Landfill and Easthampton Landfill on the large scale groundwater quality of the Barnes Aquifer.

Existing water chemistry data from wells in the vicinity of the Northampton Landfill provide preliminary evidence suggesting that landfill leachate may produce reducing conditions capable of mobilizing iron, manganese, and arsenic from aquifer sediments. Monitoring wells located both up and down gradient of the landfill have shown a marked increase in dissolved iron and arsenic concentrations in recent years despite an effort to reduce leachate production by capping an unlined portion of the landfill in 1996 (Figure 1-2). At Hannum Brook, located downgradient of the landfill, oxidized iron is accumulating from the ground surface to a depth of approximately 0.25 meters. Recent chemistry results from a domestic well survey conducted by Fuss & O'Neill (2007) indicate that arsenic concentrations in a domestic well located at 981 Park Hill Road has exceeded the Maximum Contaminant Level (MCL). Results from an October 2007 semi-annual water quality monitoring test conducted by Fuss & O'Neill indicate that several monitoring wells in the vicinity of the landfill exceed Secondary MCL's for iron, manganese, pH, and volatile organic compounds (Fuss & O'Neill, 2007).

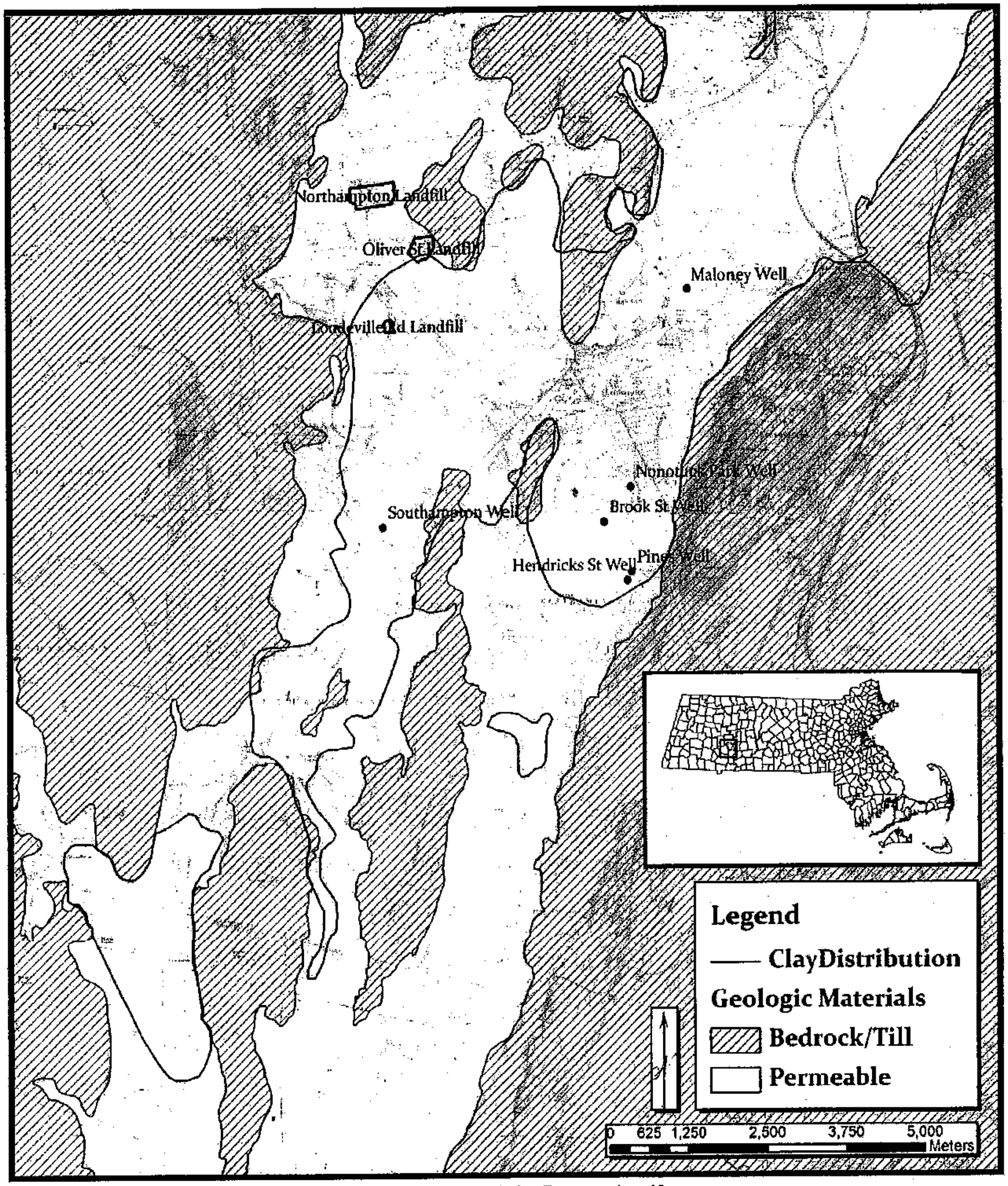

Figure 1-1. Site map of the Barnes Aquifer

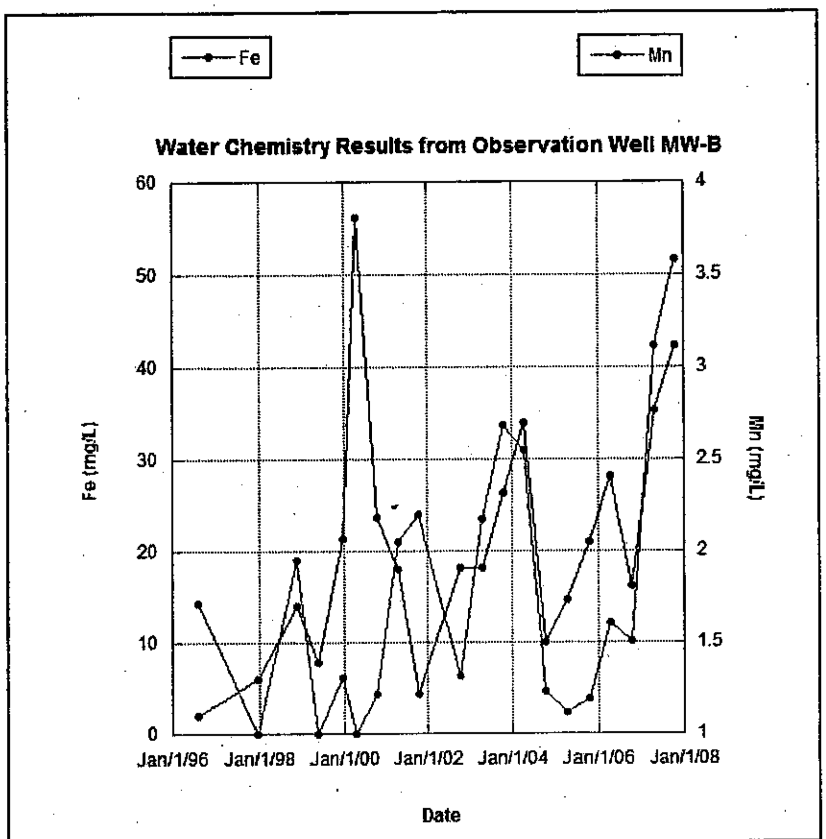

Figure 1-2. Manganese and iron concentrations at observation well MW-B (Stantec, 2007)

Though groundwater chemistry results indicate that landfill leachate is interacting with aquifer sediments in a localized scale, parameters regarding leachate generation, transport, and impacts on the large groundwater chemistry have not been sufficiently addressed. Previous studies have failed to adequately assess leachate produced from the unlined sites at the Northampton and Easthampton Landfills and calculate the extent of the resulting contaminant plumes. In addition, quantitative analysis in relation to the reaction processes between contaminated groundwater and the aquifer sediments have been predominantly ignored. Previous studies have focused on quantifying the effect of chemical species contained within landfill leachate as opposed to assessing the effects of landfill leachate on the regional groundwater chemistry particularly in relation to the mobility of naturally occurring metals. As a result, the impact of the leachate produced by landfill operations on domestic and municipal water supplies cannot be effectively analyzed. Increasing information on these processes can provide insight on how to proceed with future study, municipal landfill planning and site selection, and methods of contaminant remediation.

This study synthesizes existing data from previous studies, water monitoring reports, and original documentation to develop a groundwater model using MODFLOW 2000 capable of providing a framework to evaluate the impact of the landfill leachate on the Barnes Aquifer. It incorporates data from geologic maps, boring logs, and information from the 2005 Dufresne Henry model to develop a finite difference grid capable of effectively calculating groundwater flow parameters with sufficient detail. Additional data from pump tests and well logs are used to refine layer and hydraulic parameters to accurately assess groundwater flow. Model calibration is conducted using 18 monitoring wells throughout the area of interest. Quantification of leachate generation is achieved using the EPA Hydrologic Evaluation of Landfill Performance Model (HELP) (Shroeder, 1997). HELP calibration was achieved using weather data and knowledge of hydraulic layer constraints within the landfill. Reactive chemical transport of leachate species generated from the landfill are represented as Biological Oxygen Demand (BOD) and evaluated using RT3D (Clement, 1997). BOD is an indirect representation of the landfill leachates propensity to consume dissolved oxygen (DO) present in the groundwater. DO concentrations can be a good preliminary indicator of the oxidation reduction potential in the groundwater. Because of the complexity in coupling results from the reactive transport model with models capable of evaluating the mobility of metals, this model is constrained to only analyze the effects of landfill leachate on aquifer BOD and dissolved oxygen concentrations.

This study begins placing parameters on previously unknown processes pertaining to the mobility of species due to landfill leachate from municipal landfills in Easthampton and Northampton. Though it does not provide specific results pertaining to speciation of metals, it produces preliminary results pertaining to the regional geochemical effects of leachate production which have been primarily overlooked in previous studies. Results can be used to guide future work in analyzing the relationship between landfill leachate and species mobility in the Barnes Aquifer and provide insight into the effect of the existing landfill on groundwater chemistry of the region.

Background

Geologic Setting:

Simulated groundwater flow is controlled by aquifer porosity and hydraulic conductivity. Porosity is a dimensionless measure of void spaces within a geologic unit given by:

$$n = \frac{Vv}{Vt} ...where$$

n = porosity
Vv = volume of voids
Vt = total volume

Hydraulic conductivity is a proportionality constant derived from Darcy's Law used to estimate the amount of water a cross sectional area of a geologic unit can transmit under a unit gradient expressed in terms of length over time (cm/s). These flow parameters are primarily influenced by the geologic material, structural features and depositional environment. As a result, the geologic history of the region has a significant impact on extrapolating groundwater flow parameters.

Bedrock geology within the modeling area is composed of Newark Series sedimentary and igneous rocks formed during the upper Triassic. The majority of the area is underlain by arkosic sandstones and siltstones that comprise the Sugarloaf Arkose. Exposed areas show evidence of groundwater flow through joints, faults and bedding planes. Carboniferous Williamsburg Granodiorite, exposed at the base of Pomeroy Mountain, underlies the western portion of the modeled area. The contact between granodiorite and sedimentary rocks is marked by an eastward dipping fault or unconformity. Bedrock in the Eastern portion of the modeled area is dominated by the Holyoke Basalt which is principally composed of tholeiitic basalts and small amounts of volcanic bombs, breccia, and tuff. The Holyoke Basalt forms a continuous ridge of resistant material forming a cuesta within the Hartford Basin. Structural features including columnar joints and faults occur throughout the unit.

Suficial geology within the model area is largely the result of Wisconsinan glacial environments during the late Pleistocene epoch. These unconsolidated deposits are primarily composed of glacial till, meltwater sands and gravels, and lake sediments. Most of the sediment was deposited as glaciers retreated at the end of the last ice age. During this period the retreat of an ice lobe through the Connecticut Valley produced a number of depositional environments. Depositional processes were significantly influenced by a series of proglacial lakes which developed due to discharge from meltwater streams impounded between the ice front and a terminal moraine located in Long Island, NY. Surficial geology can be broadly characterized by relatively impermeable clays deposited as lake sediments, low permeability till deposited subglacially, and high permeability sands and gravels deposited by high energy glacial meltwaters.

Glacial till is composed of poorly sorted, unstratified clay to boulder sized sediments deposited either subglacially or melted out of glacial ice in the zone of ablation. Hydraulic conductivity values of glacial till typically range from 8×10^{-10} to 2×10^{-4} cm/s (Weight and Sonderegger, 2001). Till is exposed throughout the modeling area, particularly in the north (Stone et al., 1978). Because of the low permeability of till, it is not considered aquifer material and is used to delineate horizontal hydraulic no flow boundaries.

Glaciolacustrine sediments deposited in proglacial lakes are distributed throughout the lower elevation stream valleys forming a confining layer over the majority of the Northern portion of the modeled region (Figure 1-1). Lake sediments range from silt and fine sand to clay deposited in alternating bands known as varve clays. The estimated hydraulic conductivity for clays is 10^{-11} to 4.7×10^{-7} cm/s (Weight and Sonderegger, 2001). Due to relative impermeability, areas overlain by clay prohibit aquifer recharge. According to geologic maps, clays reach a maximum thickness of approximately 60m (Langer, 1979). Field reconnaissance and bore hole data shows that clay layer thickness is highly variable indicating that it may be leaky in many places.

Permeable materials in the modeled portion of the aquifer are primarily comprised of glaciofluvial sediments deposited by glacial meltwater streams. Deglaciation during the late Wisconsinan is defined by the gradual retreat of an ice lobe which occupied the Connecticut Valley. Pauses in retreat allowed large kame deltas to be built above the lake surface. These exposed areas of sand and gravel became the recharge area of the Barnes Aquifer. Analysis of morphologic structures and sequences show that meltwater streams deposited sediments in outwash plains, deltas, kame terraces, and eskers (Figure 2-1).

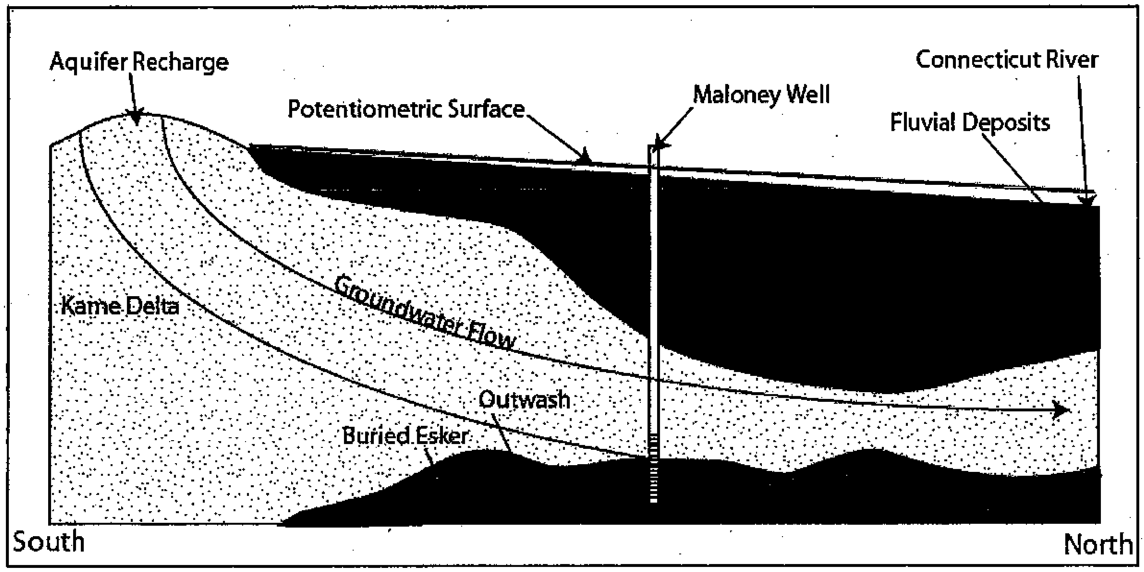

Figure 2-1. Conceptual depositional model of Barnes Aquifer sediments

During pauses in retreat glacial meltwaters deposited stratified sediments along the glacial margin. In the absence of glacial lakes these sediments were deposited as stratified outwash plains composed of silts, sands, and gravels and range in hydraulic conductivity from 10^{-4} to 1 cm/s (Hiscock, 2005). During periods when glacial meltwaters deposited directly into glacial lakes, stratified material was deposited on prodelta slopes glaciolacustrine deltas or subaqueous fans composed of sands and gravels. Deltaic topset sedimentary stuctures are typically seen when fluviolacustrine deposits exceed the lake level. Other high conductivity sediments were deposited near the terminal zone of glaciers due to subglacial or periglacial meltwater streams. Typically these features appear as long ridges of high conductivity sands and gravels running parallel to the direction of ice flow. The two features mapped by Larsen (1972) are kame terraces and eskers. Eskers are deposited as sinuous ridges of stratified drift deposited by subglacial streams. Kame terraces are comprised of stratified sediments deposited by superglacial or periglacial streams. During glacial melting, terraces may slump towards the center of the glacial valley.

In the Northern portion of the modeling extent, where the elevation of fluviolacustrine sediments did not exceed lake levels, permeable sediments were covered by fine grained lake sediments effectively confining the aquifer. Due to the complex depositional environment during glacial retreat, many sedimentary structures that provide insight into aquifer permeability are buried. As a result, the distribution of high permeability sediments were interpreted based on the development of a facies model that reflects the geomorphic sequence of the region.

Arsenic Mobilization from Landfill Leachate:

Arsenic is a major threat to water resources throughout the world. Many documented instances of arsenic contamination have been linked to mobilization of naturally occurring arsenic bound to soil and aquifer sediments. This process has been attributed to groundwater contamination in Bangladesh and has affected approximately 57 million people (Polizotto et al., 2005).

Redox potential and pH of groundwater have been identified as primary controls on arsenic mobilization (Smedely and Kinniburgh, 2002). Arsenic is thought to be mobilized through either oxidation of arsenic rich iron sulfide or the reduction of iron sulfide. In many cases, arsenic is mobilized in natural waters as it reduced from As(V) to As(III). Evidence from a study in Bangladesh has shown that these processes occur in geochemical environments characterized by low oxidation-reduction potential and high chemical or biological oxygen demand often attributed to human disturbances (Zheng et al., 2004). Studies have shown that this process can be caused or enhanced by dissolved organic carbon present in groundwater (Bauer and Blodau, 2005). Elevated arsenic concentrations in the United States have been linked to microbial activity stimulated by organic carbon in glacial sediments with low arsenic (Erikson and Barnes, 2005).

Landfill leachate is generated through microbial and chemical reactions as rainwater percolates through layers of waste (Christensen and Kjeldsen, 1989). Leachate migration can often be affected by local mounding of the water table beneath landfills (Kjeldsen et al., 1998). Strongly reducing leachate plumes have been identified extending down-gradient of landfills as a result of the development of a methonogenic zone near the landfill. As a result, reduction of iron, manganese, sulfate and nitrate has been observed within 300 m of landfills in Denmark (Bjerg et al., 1996). Additionally, bacteria cultures have been identified as key components in driving redox processes observed in anearobic leachate plumes (Ludvigsen et al., 1998, Albrechtsen and Christensen, 1994). Leachate transport has been characterized by low values of dispersion and generally controlled by the aquifer material (Jensen et al., 1993). Natural attenuation of the dissolved organic carbon responsible for reduction has been linked primarily to degradation and sorption processes (Christiansen et al., 2001). Keimowitz et al. (2005) identified a case in Saco Maine where a strongly reducing leachate plume is responsible for the mobilization of naturally occurring arsenic from glacial sediments. In this case elevated arsenic concentrations were seen as high as 300 $\mu g\ L^{-1}$. This study also reported deposits of arsenic rich iron floc, similar to the one observed near the Northampton Landfill, down-gradient of the landfill (Keimowitz et al., 2005). Similar observations near the Northampton Landfill and comparable depositional environments in the vicinity of the Northampton, Loudeville and Oliver St. Landfills suggest that elevated arsenic concentrations could be attributed to similar reducing environments generated from leachate plumes.

Landfill Background:

Due to a change in regulation prohibiting the burning of waste in 1968, the City of Northampton began actively seeking a new system of waste disposal (Northampton Board of Health, 1969). Based on studies conducted by White (1969), the 52acre Omasta Gravel Pit property located on 170 Glendale Rd. was selected as the optimum site to dispose of solid waste despite knowing of the "probability of the ground water pollution as being high". After much controversy, the Northampton Board of Health granted site assignment to the Omasta Site and the Northampton Landfill went into operation in July of 1969. The landfill occupied the gravel pit that was excavated during gravel operations to an elevation of approximately 262 ft. According to the proposal submitted by Alex Huntley, a consultant for the town, the 22 acre site could be developed to a elevation of 300 ft (Huntley, 1968). Over the next two decades the site accepted an average of 700,000-900,000 cubic yards of Northampton's municipal and light industrial waste (Dufresne-Henry, 1986).

In 1982, Northeast Consultants developed a closure plan for the Northampton Landfill that allowed vertical expansion to continue to an elevation between 320 and 328 ft (Dufresne-Henry, 1986). During this time, concerns regarding water quality of Hannum Brook led to the development of a groundwater-monitoring program implemented by the Northampton Board of Health (Dufresne-Henry, 1986). In 1982, the Northampton Water Department began taking water samples near the landfill and Park

Hill Rd. Sites were monitored for major dissolved ions and other indicators of contamination and degraded quality including Volatile Organic Carbon.

As the landfill neared capacity in the 1980's, the town of Northampton began exploring a 16 acre expansion option that would be broken into three separate phases. In 1985 a contract with Dufresne-Henry led to an environmental study of the landfill (CT Male Associates, 1992). The study assessed the impact of constructing a 16 acre lined site capable of holding up to 1.3 million cubic yards of municipal and industrial waste (Dufresne-Henry, 1986). Detailed focus was placed on interpreting the geology of the region to assess the hydrologic effects of expansion. The study conducted seismic refraction and measured electromagnetic terrain conductivity in order to determine the bedrock depth and delineate the leachate plume. Borings and water chemistry were also analyzed to determine flow characteristics and current leachate contamination to Hannum Brook (Wagner and Associates, 1986; Dufresne-Henry, 1986). A final Environmental Impact Report submitted in January of 1987 addressed concerns regarding impact of water quality due to the 16acre expansion that emphasized the limited impact on Hannum Brook fisheries (Dufresne-Henry, 1987).

In 1989 the construction of the first 5 acre phase of the new lined landfill was completed and it began operation in 1990. During this time, the unlined portion of the landfill was closed (CT Male Associates, 1992). The phase 2 lined section of the landfill, covering a 6 acre site, was completed in 1993 (Dufresne-Henry, 2005). As the phase 2 expanded section of the landfill neared capacity in 1995, the Board of Health gained authorization to construct phase 3 expansion of the landfill (Schleeweis, 1995). According to a proposal prepared by Dufresne-Henry, plans called for closure of the 6 acre phase II expansion and the construction of a new 7 acre landfill lined with 24 in of clay (Schleewies, 1995). By the fall of 1995 construction of the new landfill was complete and closure plans allowed for vertical expansion to an elevation of 334.5 ft (Schleewies, 1996). At the time of this construction the unlined portion of the landfill was capped (Dufresne-Henry, 2005). Phase 4 of the landfill expansion was created in the pit left by digging fill for phases 1-3 and thus constituted a vertical expansion on the existing footprint. Authorization to construct was granted in 2000 and the landfill began accepting waste in that year (Dufresne-Henry, 2005).

Currently, the City of Northampton is in the process of approving a phase 5/5B expansion on the current landfill site. Dufresne-Henry prepared a DEIR that addressed environmental impacts associated with a 29.2 acre expansion. Of the proposed expansion, 9.5 acres would be a vertical expansion on existing sections to an elevation of 410 ft thus is exempt from site assessment. The proposed site would allow for the disposal of an estimated 1.8 million cubic yards extending the life of the landfill by 21.3 years and maintaining refuse contracts with 39 communities in Western Massachusetts. Included in the plan are designs for a 3.5 ft thick clay liner coupled with a sand liner and leachate treatment method. Closure plans include capping smaller cells of the landfill as they fill in order to reduce leachate discharge. Massachusetts Site Assignment Regulation 310CMR16.40 prohibits the construction of landfills within the Zone II of a municipal supply well. In order to gain a waiver allowing the construction of the landfill within the Zone II of the Maloney Well, Dufresne-Henry completed a contaminant transport model simulating the effects of a contaminant release from the proposed landfill. Results suggest that the site would not negatively impact water quality at the well. The DEIR provides methods in order to successfully mitigate impacts on groundwater, air, and habitat quality associated with the construction (Dufresne-Henry, 2006).

Previous Modeling:

The Massachusetts Department of Environmental Protection (DEP) established the Conceptual Zone II for the Maloney Well during the initial stages of the Phase 5 permitting process. In 2004, Dufresne-Henry constructed a groundwater model that redelineated the Maloney Well Zone II to an extent that the proposed Phase 5 Expansion was outside the prohibited area. A DEP review of the material forced Dufresne-Henry to modify model parameters showing that the site occupied the Zone II for the Maloney Well. In order to obtain a waiver allowing for the Phase 5 Expansion Dufresne-Henry completed a contaminant transport model using the US Geological Survey Modular Three-Dimensional Finite-Difference Groundwater Flow Model (MODFLOW) computer program in 2005 (Dufresne-Henry, 2005). The model illustrates the characteristics of groundwater flow in the Barnes Aquifer and simulates the effect of landfill leachate and contaminant releases under various scenarios. Summarized below are the parameters and assumptions used in the model and results yielded.

Dufresne-Henry relied on an abundance of data available from various USGS mapping projects, independent geologic research and investigations, and pump tests in order to determine aquifer boundaries used in their contaminant transport model. These sources include bedrock maps created by Lonquist (1973, 1975) and Larsen (1972), clay data provided by Langer (1979), and USGS topographic maps (1979) in order to determine stream and pond elevations. Hydraulic conductivity and storativity were based on results from pump tests conducted on municipal wells in Easthampton and Southampton. Test analysis yielded four distinct hydraulic conductivities of 1.15×10^{-2} cm/s, 5.78×10^{-2} cm/s, 1.15×10^{-1} cm/s, and 2.31×10^{-1} cm/s within the aquifer. Storage was estimated as 6.5×10^{-3} in the confined portion of the aquifer and .05 in the unconfined. Recharge in the unconfined portion of the aquifer was calculated as 19 inches per year.

The model was calibrated to head measurements from 9 wells throughout the aquifer using average pump rates under a steady state simulation. The model was validated through a 5 day transient simulation comparing predicted drawdown to observed drawdown of the Maloney and Southampton Wells during pump tests which resulted in a 3 meter residual mean difference. Results from conducting a 180-day no recharge transient simulation with maximum approved pump rates determined that the Maloney Well capture zone had negligible increase compared to steady state conditions. This simulation showed that the steady-state capture zone could adequately simulate Zone II conditions used in the contaminant transport model. Model sensitivity was analyzed by increasing or decreasing hydraulic conductivity values by 100% and 50% respectively.

The contaminant transport model was run for manganese, zinc, methylene chloride, biological oxygen demand, and a non-reactive theoretical contaminant. Contaminant transport was modeled using the Mass Transport in 3 Dimensions Model (MT3D) developed by the EPA (Zheng, 1990). The MT3D model has the capability to model transport by accounting for the potential for contaminant advection, dispersion and retardation. The Reactive Transport in 3 Dimensions Model (RT3D) developed by the EPA was used to model the effects of biological oxygen demand and changes in dissolved oxygen due to leachate transport in the aquifer. Knowledge of dissolved oxygen concentration provides a more complete understanding of groundwater geochemistry and is important in understanding contaminant mobility.

Estimated values for aquifer porosity, conductivity, groundwater velocity, gradient, dispersivity and contaminant degradation were applied to the MT3D model in developing an accurate representation of transport. With these parameters, the MT3D is capable of determining how specified contaminant concentrations are transported and naturally attenuated through binding on the aquifer skeleton. Contaminant concentrations were determined by compiling concentration data for given contaminants from landfills in Northampton, Granby, Chicopee, and South Hadley. BOD, manganese, zinc, and methylene chloride were estimated at 5000mg/L, 20mg/L, 10mg/L, and 600μg/L respectively.

The contaminant transport model was run from three leachate release scenarios. Scenarios model the effect on aquifer contamination based on liner leakage from the entire site, a catastrophic two-day fixed release from the expanded section, and the effect of both occurring simultaneously. Conservative estimates of leachate volume released from the existing and proposed lined landfills were based on empirical equations for liner leakage reported in the literature (Qian et al., 2002). The equation was used conservatively to evaluate flow based on estimates for hole distribution and size and liner construction. An estimated 287.69 L/day was used in the model simulation. Leachate release from the unlined portion was assumed as 61,588 L/year using the EPA Hydrologic Evaluation of Landfill Performance model. Based on per acre peak leachate generation monitored from the leachate collection system from the existing landfill, the two-day fixed release from the proposed landfill was established at 582,953 L/day.

Results from the three release scenarios suggested that the landfill expansion would not present a risk of contamination to the Maloney Well. In all scenarios the BOD plume did not significantly extend beyond the landfill footprint thus immobilizing manganese. Zinc and methylene chloride were naturally attenuated in the aquifer material and did not develop a plume. Results from simulating a theoretical contaminant with no capacity to attenuate showed that even a 1000mg/L initial concentration would be diluted to a maximum of 1.5μg/L at the Maloney Well under the most conservative assumptions.

The existing model suggests that expanding the Northampton Landfill would have a limited impact on the regional groundwater chemistry. It predicts that the potential impact on the Maloney Well is negligible. However, the model fails to adequately assess the impact on metal mobility despite the knowledge of low dissolved oxygen and high manganese in the confined portion of the aquifer.

Groundwater Flow Program:

Groundwater flow in the Barnes Aquifer is modeled using Visual MODFLOW Pro. Visual MODFLOW Pro was developed as a graphical interface for the U.S. Geological Survey MODFLOW modeling software and runs using an updated version of MODFLOW 2000 released in 2004 (Waterloo Hydrogeologic, 2004). MODFLOW models groundwater flow in an aquifer in three dimensions and evaluates how it changes with time using a finite difference grid solution to the groundwater flow equation. It allows for easy manipulation of input parameters without manipulating the over all model structure. Visual MODFLOW is used in the development of the Barnes Aquifer leachate contamination model because it provides a user-friendly graphical interface that allows for various options in visualizing model inputs and results.

MODFLOW is capable of determining the head distribution and components of three-dimensional groundwater flow in an aquifer using a finite difference method to solve for the partial differential groundwater flow equation. By incorporating the hydraulic conductivity, hydrologic boundaries, and changes in storage estimated from field measurements MODFLOW uses the finite difference method to calculate flows based on empirically derived equations (McDonald and Harbaugh, 1983).

MODFLOW calculates groundwater flow by dividing the horizontal and vertical modeling area into a grid of nodes that represent three-dimensional cells with uniform hydraulic properties for the x, y, and z flow directions. The principle assumptions in the model are: 1) a constant density of water and 2) the flow out of a cell will equal flow into a cell +/- changes in storage where:

$$Vol_{in} = Vol_{out} + \Delta Storage$$

Groundwater flow is determined by replacing the partial differential groundwater flow equation with algebraic equations which govern flow between cells based on head differences. This is achieved by calculating the flow from each face of cells using a separate flow equation. MODFLOW uses an iterative method to estimate head values in cells that provide a solution that fulfills the set of groundwater flow equations (McDonald and Harbaugh, 1983).

The MODFLOW software groups smaller components of the program code into modules that complete specific tasks. Modules are grouped into "packages," such as the recharge or river package, that simulate the effects of a particular aspect of hydrologic inputs and stresses (McDonald and Harbaugh, 1983). Two packages are required to run the model. The Basic Package controls how different components of the model operate including model output and initial conditions. The Block-centered Flow Package controls the operation of the finite difference equations that govern flow. Additional stress packages such as the Well Package, River Package, and Recharge Package allow for the modeler to simulate actual conditions affecting groundwater flow in an aquifer. Solver Packages such as the Strongly Implicit Procedure Package refine techniques used to solve equations allowing for more rapid solutions (McDonald and Harbaugh, 1988). Subsequent releases of MODFLOW in 1988, 1996, 2000 have included additional packages that allow for the simulation of more complex modeling parameters (Harbaugh et al., 2000).

Leachate Production Program:

Leachate generated from landfills included in the contaminant transport model was quantified using the EPA Hydrologic Evaluation of Landfill Performance (HELP) program developed by the U.S. Army Engineer Experiment Waterways Station for the EPA in 1984 (Shroeder, 1997). HELP is a DOS based quasi-two-dimensional model used to simulate flow into, through, and out of landfills used to optimize landfill design. HELP considers a number of climatic, design, vegetation, and hydrologic variables in order to calculate the water budget for specific landfill designs. Design data from the Northampton, Loudeville Rd., and Oliver St. landfills was coupled with climatic data from Northampton and Worcester, MA in order to quantify annual leachate production from landfill cells in the region.

Climatic variables used in HELP are grouped into precipitation, evapotranspiration, temperature, and solar radiation. Climatic variables are coupled with vegetation data, to estimate surficial hydrologic parameters. Required climatic inputs are daily precipitation, daily temperature, daily solar radiation, quarterly relative humidity, latitude, annual average windspeed, and growing period. Daily climate data can be entered manually, imported as ASCII files or directly from National Oceanic and Atmospheric Administration (NOAA) tape, or synthetically generated using existing data from nearby cities (Ruffner, 1985). Vegetation inputs include evaporative zone depth, leaf index, and stand quality (Schroeder, 1997).

HELP includes a variety of landfill design parameters in order to assess hydrologic flow in a variety of settings. Soil layers are divided into four numbered groups: 1) vertical percolation layer, 2) lateral drainage layer, 3) barrier soil liner, and 4) geomembrane liner. Soils and materials are subdivided into 42 textures that include default values for porosity, field capacity, wilting point, and saturated hydraulic conductivity (Schroeder, 1997). HELP requires the user to input landfill size, slope, cover, and layer placement and thickness. In addition, specific information regarding geomembrane type, placement, and pinhole density is required for accurate evaluation of barrier leakage (Schroeder, 1997).

HELP utilizes a number of equations and methods in order to calculate hydrologic parameters used in model simulations (Schroeder, 1997). Complex variables such as frozen soil and snow accumulation and melting employ models such as Chemicals, Runoff and Erosion from Agricultural Management Systems (Knisel, 1980) and the NWSRFS SNOW-17 model (Anderson, 1973) in order to accurately estimate model responses. Other hydrologic parameters are estimated using a variety of flow evapotranspiration, runoff, and infiltration equations. The program relies heavily on the use of the Penman Method (1963), and Darcian flow equations (Schroeder, 1997).

Solute Program:

Contaminant transport of landfill leachate is modeled using Reactive Transport in 3-Dimensions (RT3DV2.5). RT3D code provides solutions to partial differential equations used to predict the fate and transport of multiple mobile and immobile species in groundwater environments. RT3D relies on MODFLOW code to provide head distribution within the area of interest. RT3D is an adaptation of MT3D code (Zheng, 1990) and relies on the same chemical advection and dispersion solvers (Clement, 1997). Flexibility in RT3D code allows for manipulations of species parameters permitting simulation of a variety of chemical species (Clement, 1997). Currently RT3D supports seven preprogrammed general reaction packages allowing for the reactive transport simulation of petroleum fuel components (BTEX), tetrachloroethene (PCE), and trichloroethene (TCE). RT3D is particularly useful in simulating chemical species which are naturally attenuated through decay processes (Clement, 1997).

The generalized governing equation for three dimensional solute transport in saturated media used by RT3D is determined through smaller equations governing advection, dispersivity, source/sink mixing, and chemical reaction (Clement, 1997). Advection refers to the generalized concept of fluid transport determined primarily by pore velocity and direction. Dispersivity quantifies the tendency for a pollutant to move laterally from the primary trend of groundwater flow. Dispersivity is represented by a dispersion coefficient which is a scale sensitive product of the groundwater velocity derived from Darcy's law and dispersivity. These equations are grouped to form the overall solute transport equation (Clement, 1997):

$$\frac{\partial C_k}{\partial t} = \frac{\partial}{\partial x_i}\left(D_{ij}\frac{\partial C_k}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(v_i C_k) + \frac{q_s}{\phi} + C_{s_k} + r_c \quad \text{where } k = 1,2,\ldots\, m$$

$$\frac{dC_{im}}{dt} = r_c, \text{ where } im = 1,2\ldots\,(n\text{-}m)$$

n = total number of species
m = total mobile species
C_k = aqueous phase concentration of the k^{th} species
C_{im} = solid phase concentration of the im^{th} species
D_{ij} = hydrodynamic dispersion coefficient
v = pore velocity
ϕ = porosity
q_s = volumetric flux per unit volume of aquifer (sources/sinks)
C_s = concentration of source/sink
r_c = aqueous phase reaction rate
r_c = soil phase reaction rate

RT3DV2.5 relies on solutions determined by MT3DMS code for computing the dispersion and advection problems but runs them repeatedly in solving multiple species problems (Zheng, 1999). The reaction solver is a unique code specific to RT3D (Clement, 1997)

RT3D utilizes unique reaction solver packages based on the reaction mechanisms of a given contaminant. Module 1 which simulates the Instantaneous Aerobic Decay of BTEX was used in simulating the degradation of landfill leachate. Under default settings the Module 1 solver assumes a degradation reaction for a generic hydrocarbon given as:

$$C_6H_6 + 7.5O_2 \rightarrow 6CO_2 + 3H_2O$$

where either BTEX or oxygen are substituted into the generalized solute transport equation to simulate removal. Module 1 assumes two total chemical components both of which are mobile. The degradation reaction of either BTEX or carbon is calculated by an instantaneous algorithm reaction where either BTEX or oxygen is completely reduced in each reaction step (Clement, 1997). Two separate degradation reactions are used depending on the limiting reactant (Rifai et al, 1988):

$$H(t+1) = H(t) - O(t)/F \text{ and } O(t+1) = 0, \text{ when } H(t) > O(t)/F$$

$$O(t+1) = O(t) - H(t)\bullet F \text{ and } H(t+1) = 0, \text{ when } O(t) > H(t)\bullet F$$

t = time step
F = stoichiometric ratio (C_6H_6/O_2 : 3.072)

By adjusting the stoichiometric ratio instantaneous reactions between any two species can be simulated.

Hydrologic Flow Model Development:

Groundwater flow in the area of interest (Figure 1-1) was modeled using a fully three dimensional finite grid representation of the aquifer units and clay confining layer. Due to 50 year time parameter used contaminant transport modeling, seasonal fluctuations in recharge, stream leakage, and other dynamic boundaries do not significantly impact the overall model outcome. As a result, the simulation was carried out under steady state conditions. The model was run conservatively under default settings for maximum inner and outer iterations in order to reach a solution that yielded no changes in the total calculated water budget. Development of the model reflects an effort to incorporate all existing geologic and hydrologic data collected from a number of USGS and other engineering reports while yielding a solution that accurately reproduces the distribution of measured heads.

Model Extents

Model extents were chosen by identifying major horizontal flow boundaries within the area of interest. Boundaries exist where groundwater divides, discharge sites, and impermeable materials restrict groundwater flow in the aquifer. Geologic boundaries were selected based on where the sand and gravel glacial stream deposits contact till and bedrock outcrops using USGS surficial geologic maps (Stone et al., 1978). Groundwater divides in the southern extent of the modeling region were conservatively selected based on bedrock divides given a bedrock contour map of the Mount Tom Quadrangle (Londquist, 1973). The lack of a clearly defined bedrock divide in these locations placed model divides in locations where a clear southern slope is apparent. A Northeastern boundary was selected where the aquifer discharges in the Connecticut River. This area was delineated using a topographic map of the Mount Holyoke Quadrangle (USGS[a], 1979). Other extents were selected where the amount of permeable material would not contribute enough groundwater recharge to significantly impact flow parameters without sacrificing overall model detail. Horizontal inactive boundaries in outcrops of bedrock and till are modeled through inactive cell assignment within the rectangular finite difference grid (Figure 3-1).

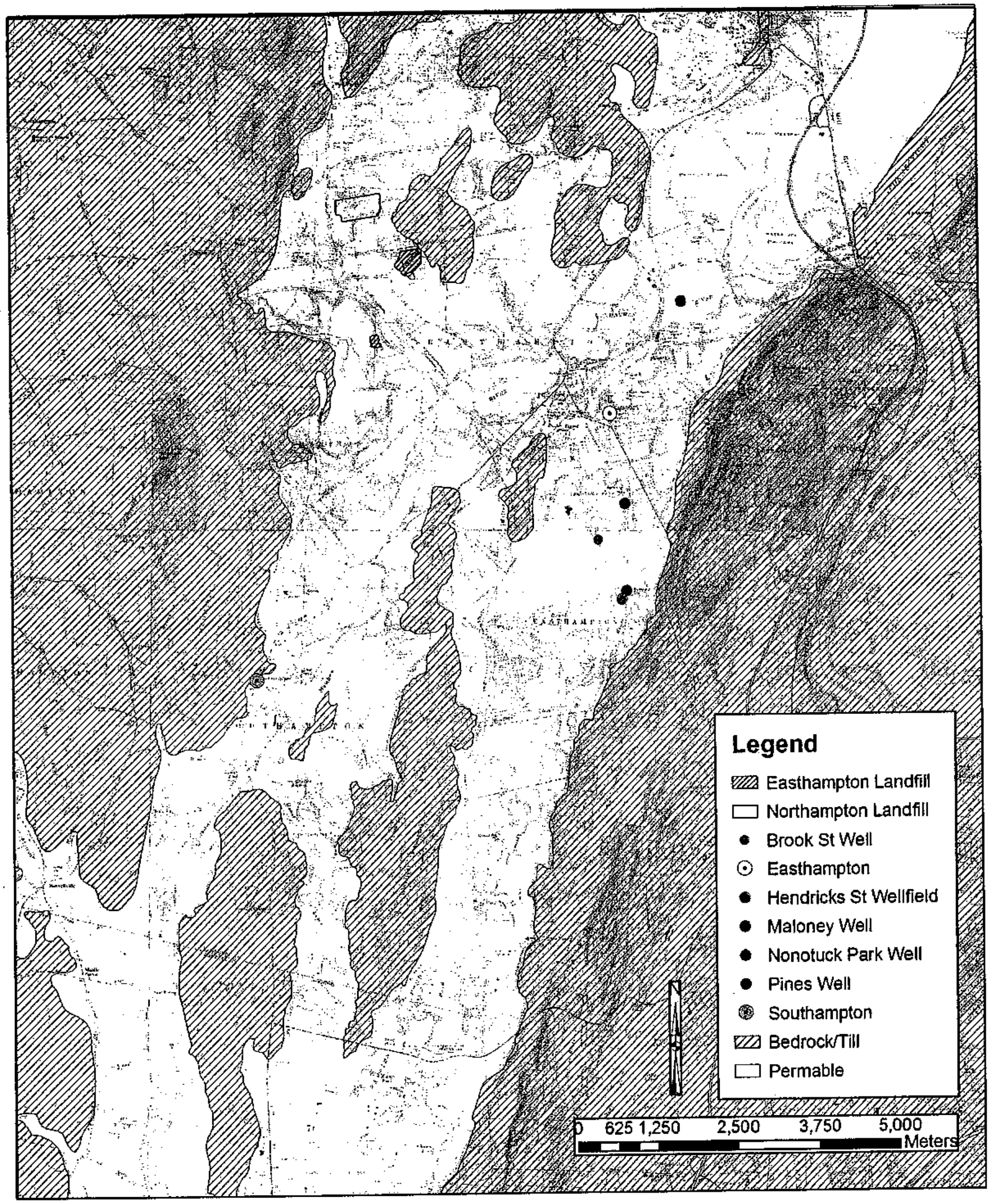

Figure 3-1: Horizontal Boundaries within Modeling Extent

Layer Development:

Vertical flow boundaries between aquifer units were developed using the fully three-dimensional option where aquifer units and confining units are specifically denoted within the layer structure. This option requires explicit representation of flow parameters and geometry of each model layer. The model includes simplified 2-layer reproduction of the aquifer sand and gravel deposits, confining clay layer, and bedrock surface. Though there is evidence of groundwater flow through bedrock fractures, well yields in sites along Park St, Northampton suggest that bedrock hydraulic conductivity is negligible and was not included as a model layer (Massachusetts Department of Conservation and Recreation, 2007). As a result, the bedrock surface is modeled as a lower confining boundary. Simulated layers also exclude topsoil and silty-sand lenses. These layers were modeled by manipulating hydraulic conductivity during model calibration.

Layer surfaces were developed using geologic maps, well logs, and a USGS Digital Elevation Model (DEM) of the ground surface elevation. Three-dimensional subsurface layer boundary surfaces were created using interpolation functions within ArcGIS (ESRI, Redlands CA) and imported into Visual MODFLOW as a grid of points containing spatial and elevation data. Cell elevation data was calculated using nearest neighbor interpolation that averaged elevations of multiple points in cell elevation assignment. Point density was identical in all model layers imported in order to insure matching interpolations within the model software.

Ground surface elevation data for the model area was retrieved from the USGS National Elevation Dataset using the USGS Seamless Server with 30-meter resolution (USGS, 2008). Elevation data was projected in Massachusetts State Plane coordinates and coarsened to 30 meter resolution by converting the elevation raster into a point file and reconverting it into a raster with the desired cell size. This raster was converted into a point file and imported into Visual MODFLOW using the *Create Grid Elevation* feature. Cell top elevation assignment was determined based on the nearest neighbor elevation interpolation method.

Clay surface elevation data was developed through subtracting clay thickness data from surface elevation data from the USGS DEM. Clay thickness was interpreted based on clay isopach maps from the USGS and a Barnes Aquifer study conducted by IEP and well logs collected in the modeling area (Langer, 1979; IEP Inc, 1987). It was digitized by tracing isopach contours and holding spatial and elevation data in feature class using Arc GIS. Due to the increased detail and consistency with collected well logs in the IEP study area, and effort was made to retain map contours as they originally appeared in the map. Changes in clay distribution were made in constructing consistent contour between the boundary IEP study area and the USGS regional map. The USGS isopach maps were used as a base source of data in all other confined portions of the modeling area. Contour lines were added or subtracted in areas where well logs suggested thicker clay deposits (Figure 3-2).

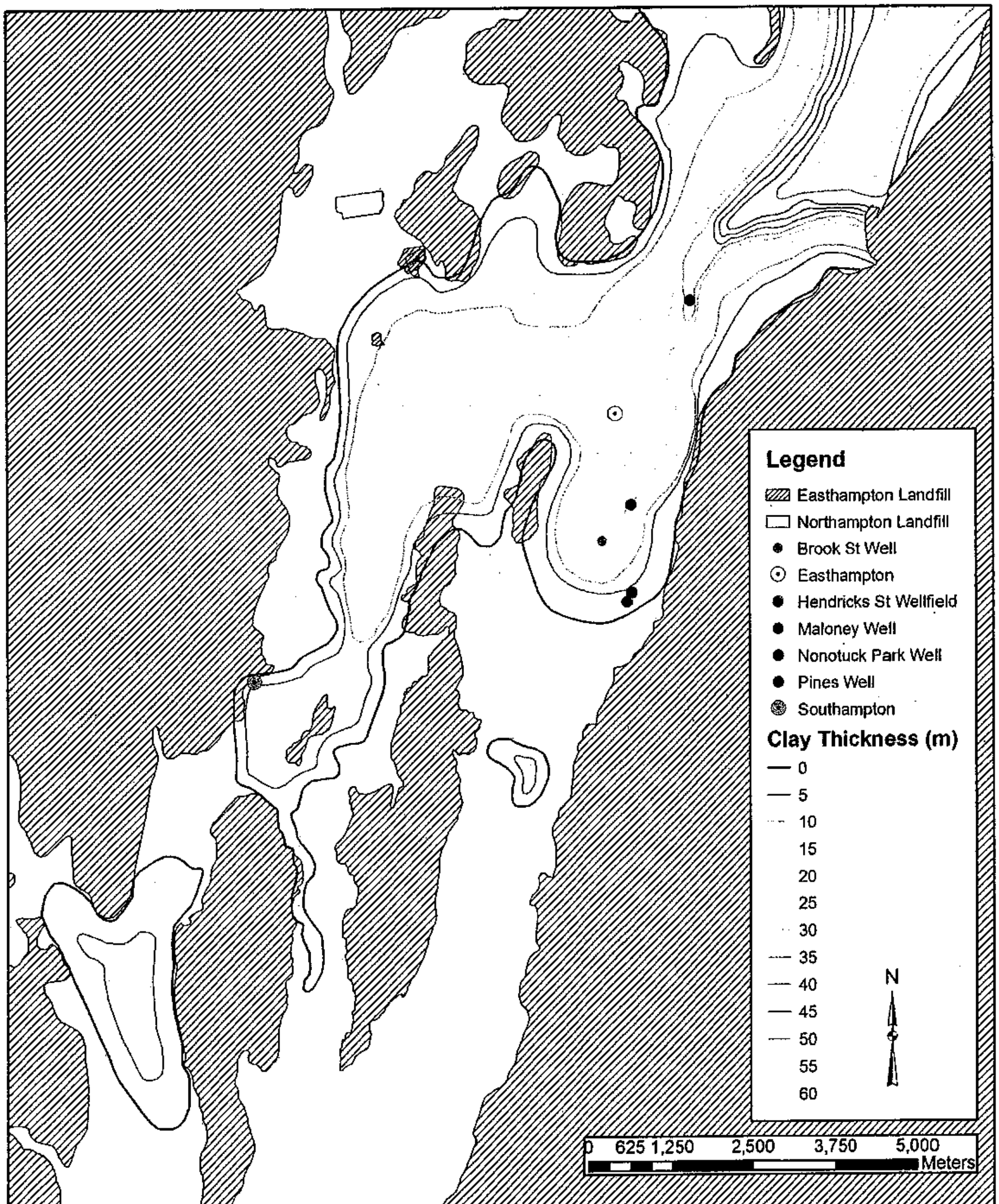

Figure 3-2. Estimated thickness of clay confining layer based on well logs and isopach maps

A three-dimensional surface of the clay thickness was created through conversion of contour data into a TIN surface. TIN interpolation was used because it provides a direct interpolation between contour lines as opposed to other methods that produce visually inaccurate surface. TIN surface elevation data was converted into an integer raster using the *3D Analyst* conversion features. The elevation of the bottom of the clay surface was defined through subtracting the clay thickness raster from the surface elevation raster. In areas without a confining layer the clay surface elevation is identical to the surface elevation. Consistent cell elevation assignment within the modeling software requires that the same number of points are used per cell in the nearest neighbor interpolation method. As a result, the bottom of the clay surface raster was checked to ensure that the 30 meter cell size was maintained before data was converted to a point file. XYZ data was exported to an Excel file and the surface was imported into the modeling software. In areas where the confining clay layer was absent, the upper layer was assigned a default thickness of 1m. Aquifer properties of the upper layer in these areas were consistent with the layer below them in order to maintain isotropic flow between cells.

The lower no-flow boundary of the model was placed where sand and gravel layers contacted low permeability bedrock. In the Mt Tom and Mt Holyoke quadrangles 50 ft interval contour maps were used to develop bedrock elevation data used in the model (Londquist, 1973; Londquist, 1975). A number of maps and reports were synthesized to develop an accurate model of bedrock elevation data in the Easthampton quadrangle. This process employed the use of over 30 well logs from site studies (IEP, 1987), the DCR Well Driller Registration Program (Massachusetts Department of Conservation and Recreation, 2007), maps prepared by Smith College students in 1982 (Newton), and a bedrock map developed by Dufresne-Henry in 2005 (Figure 3-4). The Dufresne-Henry map was used as a base map because it contained bedrock elevation data in the area of the landfill. The conceptual hydrogeologic model of the Barnes Aquifer suggests that groundwater dynamics of the Barnes Aquifer is very dependent on accurately capturing the low point in the bedrock ridge that separates the two distinct aquifer regions. Well logs and locations of bedrock outcrops were used in order to refine the precise location and elevation of the gap in the bedrock ridge. Well log data was also used more comprehensively mapped bedrock contours in the area of the Maloney Well and landfill sites. Synthesized data was used to construct a 10m bedrock contour map of the modeling region using ArcGIS (Figure 3-3).

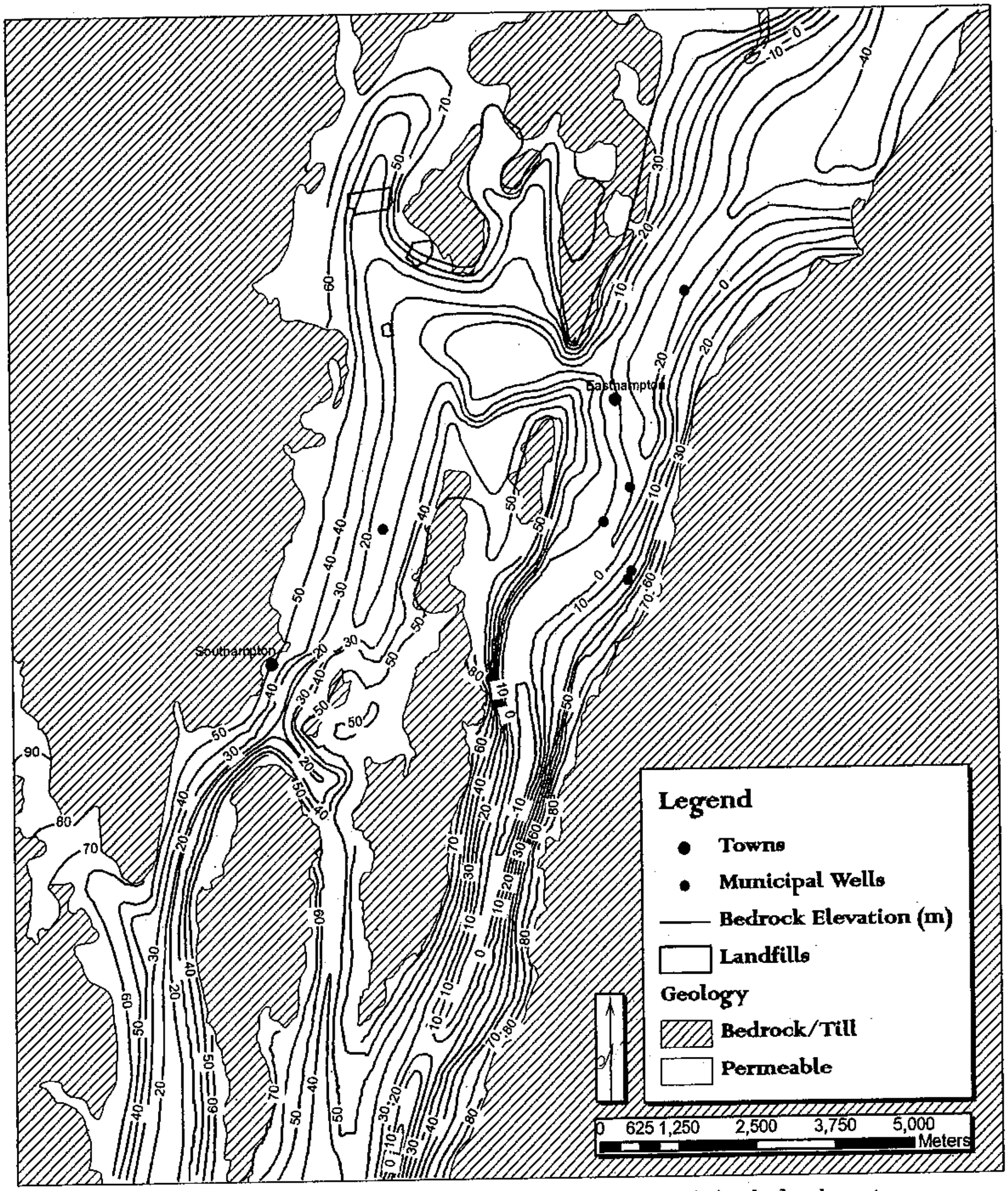

Figure 3-3. Estimated bedrock contours based on boring logs and existing bedrock contour maps

Ten meter contours were used to develop a three dimensional surface model of the bedrock data that could be imported into Visual MODFLOW as a grid of points. Contour lines were converted into a integer based raster through TIN interpolation and conversion. Raster cells were checked to ensure that cell sizes in the bedrock raster matched those of the surface and clay rasters. In order to ensure that the bedrock elevations did not exceed surface elevations, the bedrock raster was subtracted from the surface raster and corrected for negative valued cells. The bedrock raster was converted to a grid of points and XYZ data was exported into a Microsoft Excel file.

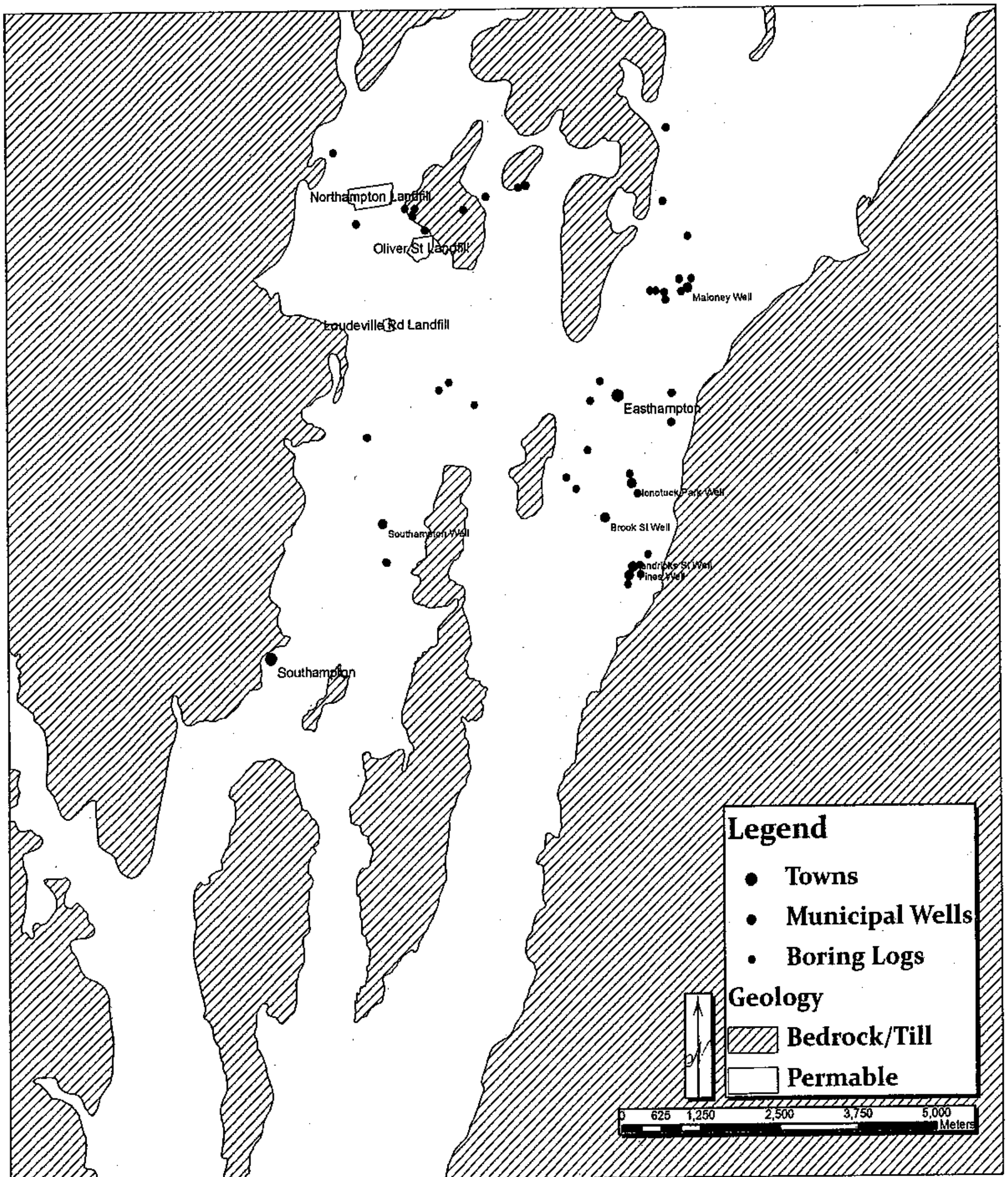

Figure 3-4. Distribution of well logs within modeling area

In order to capture variability in conductivity between lake sediments and more permeable fluvial sediments vertical geometry was modeled through two separate layers. Upper layer thickness was initially identical to clay layer thickness in confined portions of the aquifer and one meter thick in unconfined portions. Conductivity and storage coefficients in the unconfined portion of the aquifer were identical in both layers producing homogeneity between top and bottom layers. Top layer bottom elevations were subsequently dropped during stream cell assignment to ensure that streams were not assigned to the bottom layer.

Groundwater Flow Parameters:

The modeled portion of the Barnes Aquifer discharges primarily in the Connecticut River in the Northeast of the modeling area (USGS[a], 1979). Geologic evidence suggests that it cuts through the impermeable clay layer in various locations permitting groundwater to enter the river channel through permeable sands and gravels. The exact locations of exposed sand and gravel deposits have not been mapped making an explicit representation of hydrologic flow paths impossible. As a result, aquifer discharge was indirectly modeled through constant head cells modeled after the stage of the Connecticut River in the specified location. Constant head cells represent continuous groundwater sources or sinks within the modeling software. As a result, all head values in assigned cells will take on the value of the constant head value. A 27.3 m constant head elevation was assigned based on a 10meter resolution digital elevation model retrieved in February 2008 (USGS, 2008). Constant head were assigned to the permeable second layer and the confining clay layer in all cells overlain by the Connecticut River as shown by USGS topographic maps (USGS[a], 1979).

Groundwater divides represent locations where high elevations of potentiometric head force water to flow in opposite directions forming horizontal hydrologic flow boundaries. Groundwater divides exist in the southern extent of the modeling region where bedrock divides force a portion of the groundwater within model extents to discharge to the south (Figure 3-3). Rather than terminating model extents at inferred groundwater divides, an effort was made to explicitly model locations where groundwater would discharge into other groundwater catchments. Divides in the southern extent of the modeling region were represented through bedrock divides. Course bedrock contour intervals in the bedrock map for the Mt Tom Quadrangle (Londquist, 1973) were refined through the addition of contours in order to precisely simulate the location of bedrock divides. Local mounding of the water table in the southern terminus of the modeling extent was avoided through the assignment of drains. The MODFLOW drain package allows for the subtraction of water in assigned cells from the calculated water budget effectively allowing groundwater flow to escape the model. Drains were assigned as continuous units in the southernmost portion of layer 2. Drain conductance and elevation control the quantity of water exiting the model and the elevation of the groundwater divide. These values were calibrated through successive estimations that matched water table elevations in model outputs to the elevation of lakes and ponds in the southern portion of the model.

The conceptual model of Barnes Aquifer formation suggests that aquifer recharge occurs primarily over unconfined sand and gravel sediments deposited as glacial outwash or glaciolacustrine deltas (Figure 2-1). Aquifer recharge was assigned to unconfined portions of the aquifer primarily in the northwest and southern extent of the modeling region. Accurate estimations of recharge rates are typically calculated indirectly using variations of the Penman Equation to evaluate the water budget. For the purposes of this study, a generalized rate of 405mm of recharge was assumed based on 35% of the average annual regional precipitation determined from NRCS data ranging from 1961-1997.

Small brooks and rivers were represented using the MODFLOW river package. Small streams and rivers are distributed throughout the modeling region. In unconfined portions of the aquifer, surface water and groundwater interactions have a significant impact on groundwater flow, head distribution, and fluxes in the regional water budget. In confined areas, surface interaction was assumed to be insignificant due to the low conductivity clay buffer. The MODFLOW river package quantifies groundwater fluxes through rivers using a generalized conductance formula. This formula considers river bed hydraulic conductivity (K), reach length (L), river width (W), and riverbed thickness (M) to groundwater losses or gains through rivers where:

$$C = \frac{K \times L \times W}{M}$$ (Waterloo Hydrogeologic, 2004)

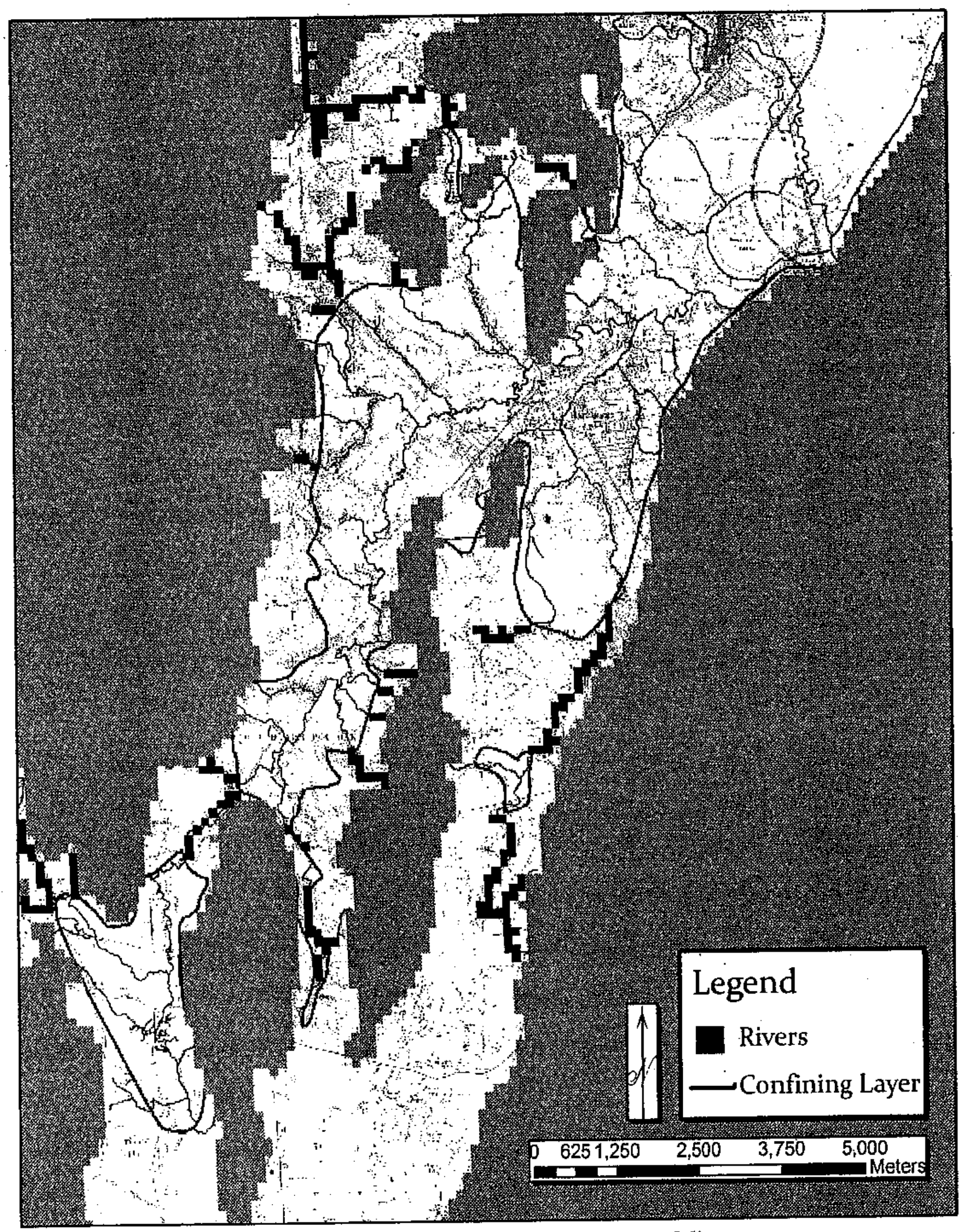

Figure 3-5. River assignment within the modeling area.

Rivers cell assignment was determined using USGS topographic maps developed for the Mount Tom, Easthampton and Mount Holyoke Quadrangles (USGS[abc], 1979). River parameters were estimated based on stream order, slope, and watershed size. River widths ranged from 0.75 m to 1.75 m. Bed thickness ranged from 1 to 2 m. Small rivers in the region are typically underlain by fine sand and were assigned conductivity values of 10^{-3} cm/s although values in the literature were reported as high as 5 x 10^{-2} cm/s (Chen, 2004). Other surface waters such as lakes, ponds and swamps were not assigned river cells due to the lack of an outlet for water. Rivers were assigned to all unconfined portions of the simulated aquifer (Figure 3-5).

Water withdrawals from municipal wells were represented based on 50% the maximum approved pump rate (Table 3-1). Actual pump rates vary seasonally. 50 % was chosen as the simulated value in order to capture a generalized effect of wells in the modeling region.

Well	Max Approved Yield (mLd/mgd)	Simulated Rate (mLd/mgd)
Maloney	5.68/1.50	2.84/0.75
Nonotuck Park	4.31/1.14	2.16/0.57
Brook St.	5.49/1.45	2.76/0.73
Pines	3.82/1.01	1.89/0.50
Hendricks St.	4.54/1.20	2.27/0.60
Southampton	2.99/0.79	1.51/0.40

Table 3-1. Maximum approved well yield and simulated pump rates.

Modeled conductivity values represent an effort to balance conductivity values documented in the literature for materials found in boring logs with values that produced an accurate groundwater head calibration. Hydraulic conductivity was considered the least precisely estimated variable in model development. Absence of data and difficulty in accurately modeling fracture flow in bedrock units prohibited an assessment of groundwater flow outside of unconsolidated materials. As a result, hydraulic conductivity values used to simulate flow in the Barnes Aquifer are overestimated to compensate for loss. Table 3-1 summarizes assumptions used in evaluating hydraulic conductivity values used in model development.

Geologic Unit	Material	Hydraulic Conductivity
Arkose/Granodiorite/Basalt	Bedrock	0 cm/s
Glacial Till	Poorly Sorted Clay-Boulders	0 cm/s
Lacustrine Bottom Sediments	Varved Clay	10^{-6} cm/s
Outwash (fine grained)	Silty Sand	10^{-2} cm/s
Outwash (medium grained)	Sand	5 x 10^{-2} cm/s
Outwash (course grained)	Sand and Gravel	10^{-1} cm/s
Esker/Kame	Well Sorted Sand and Gravel	5 x 10^{-1} cm/s

Table 3-2. Geologic Units, Materials, and Conductivity Assumptions in Model Development

Hydraulic conductivity values were horizontally represented based on inferred depositional environments of aquifer units. Conductivities strongly relate to generalized glacial processes that deposited high conductivity sands and gravels that make up the Barnes Aquifer. Particular emphasis was placed on effectively representing well-sorted sands and gravels deposited as eskers or kame terraces. Evidence from boring logs suggests that these sediments were deposited as North-South trending sinuous ridges that act as major conduits of groundwater flow. Other high permeable materials include outwash plains that were represented by broader areas of high conductivity sediments (Figure 3-6). Major contrasts in the vertical succession of facies due to the deposition of confining varved clays in glacial lakes were represented through low conductivity upper layer in the location of the confining layer. Limited data pertaining to aquifer stratigraphy prevented an explicit vertical representation of the geomorphic sequence within sand and gravel layers. As a result, permeable aquifer units were modeled with vertical homogeneity reflecting generalized aquifer characteristics.

Accurate calibration required the simulation of confining layer leakage in the Western portion of the aquifer. Leakage was simulated as a negative recharge flux in order to ensure that sufficient water was removed from the aquifer in this region. Leakage rates required for calibration were established as 250mm/yr, 300mm/yr, and 380mm/yr. Leakage was simulated in order to reduce the amount of water required to flow over the bedrock gap. Actual processes for groundwater leakage are probably due to river leakage over the confining layer. However, lack of data regarding river leakage prohibited an explicit representation of this process choosing to show a more generalized model for water losses. Leakage was also simulated in the vicinity of the Maloney Well at a rate of 300mm/yr based on site assessment conducted in November, 2006.

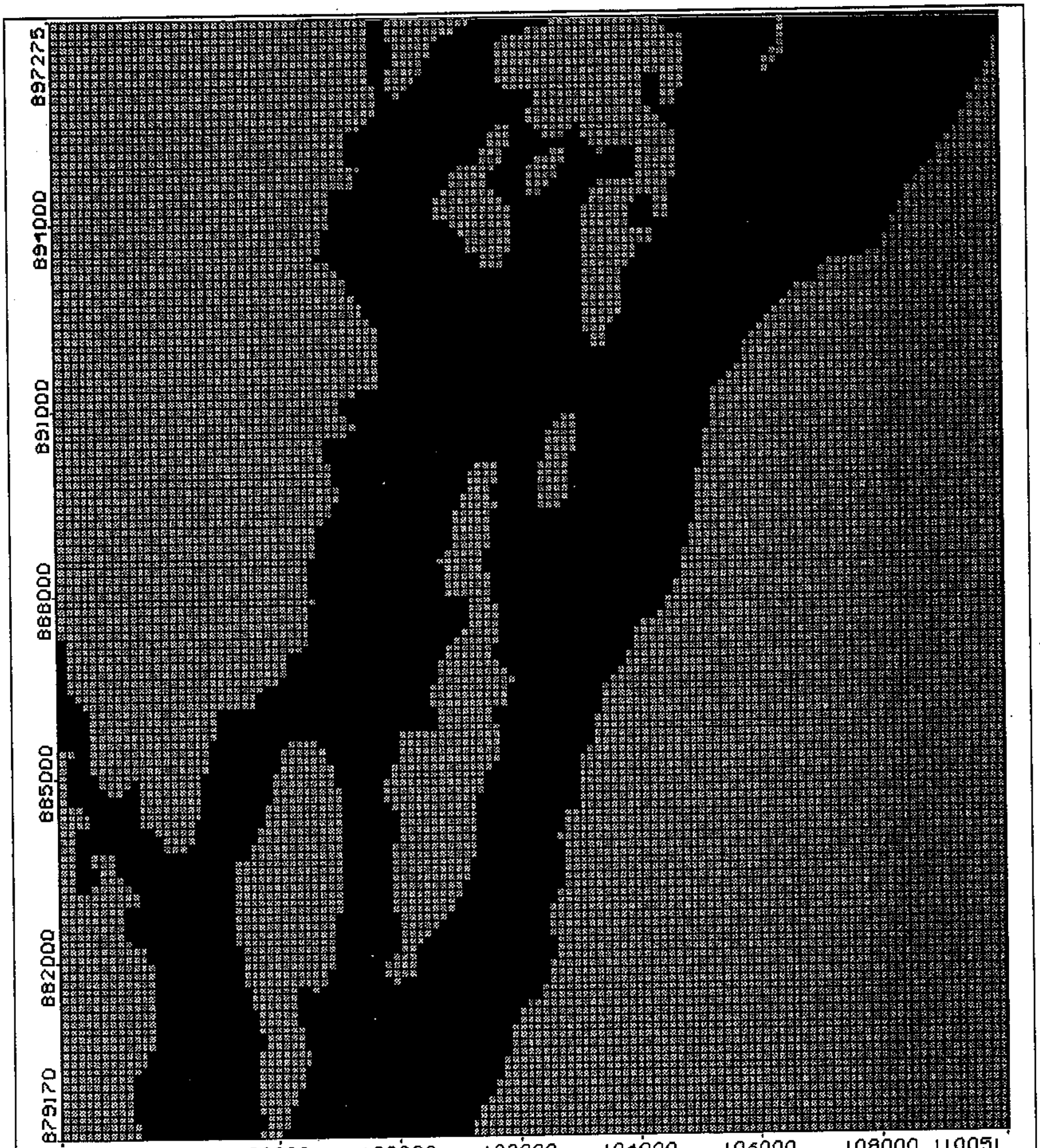

Figure 3-6. Geologically inferred conductivity values assumed in model development (blue: silty sand, green: sand, aqua: sand and gravel, brown: well sorted sand and gravel). Grid represents model cell size.

Solute Transport Model Development:

The contaminant transport model was developed to simulate the regional effects of landfill leachate from existing unlined landfills on groundwater chemistry. The model evaluates the spatial and temporal effects of leachate degradation on dissolved oxygen (DO) concentration and biological oxygen demand (BOD) in order to predict the impact on the mobility of arsenic, manganese, and iron. The model couples groundwater flow parameters calculated in MODFLOW with leachate quantity and transport estimated using EPA HELP and RT3D. Landfill leachate estimates were based on construction parameters presented in the Summary Report on Contaminant Transport Model (Dufesne-Henry, 2005). In this case, dissolved organic matter was represented as BOD due to lack of data pertaining to the actual organic carbon constituents present in the leachate. The model considers DO recharge, leachate production (represented as a recharge concentration), and groundwater flow under moderately conservative assumptions.

Development of EPA HELP Model:

Leakage estimates calculated using EPA HELP were simulated under both conservative and estimated assumptions based on reported literature values for landfill cap leakage and known design parameters (Table 4-1). Conservative, in this case, refers to an overestimation of parameters that would increase the overall impact of the landfills on the groundwater chemistry of the Barnes Aquifer. Climactic data was collected from National Resources Conservation Service database and synthetically generated using default weather data in the HELP software.

Precipitation and temperature data from 9 years with complete data from 1961-1997 from Amherst, MA (NRCS, 2008) were manually input; synthetic data for solar radiation was generated from Worcester, MA (Ruffner, 1985). A generic landfill design scheme for all landfills was developed based on information included in the Dufresne-Henry contaminant transport model (2005) (Figure 4-1). The HELP simulation of the Loudeville Rd. and Oliver St. Landfills was run based on designs for the Northampton Landfill due to insufficient data at these sites. Other design parameters were estimated based on suggested values included in the HELP software and default soil characteristics

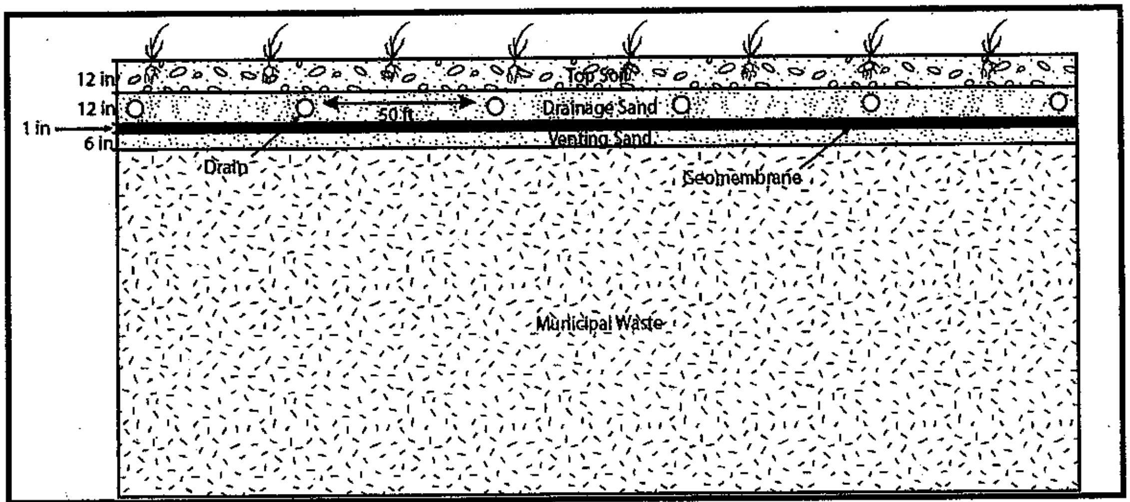

Figure 4-1. Generalized conceptual landfill design for unlined landfills simulated using HELP.

included in the software (Table 4-1, Clement, 1997). Liner placement and defect densities were estimated based on values reported from site studies (Giroud and Bonaparte, 1989). Defect parameters were estimated based on good installation for the estimated simulation and poor installation for the conservative run.

Variable	Estimated	Conservative
Topsoil Layer (inches) (cm/s)	30.48	30.48
Topsoil Layer Conductivity (cm/s)	3.70E-04	1.70E-03
Drainage Layer Thickness (cm)	30.48	30.48
Drainage Layer Conductivity (cm/s)	5.80E-03	5.80E-03
Geomembrane (cm)	2.54	2.54
Membrane Conductivity (cm/sec)	4.00E-13	4.00E-13
Venting Layer (inches) (cm/s)	6	6
Venting Layer K sat. (cm/s)	3.70E-04	3.70E-04
Municipal Waste (cm)	1920	1920
Municipal Waste Conductivity (cm/s)	1.00E-03	1.00E-03
Pinhole Density (#/acre)	1	5
Defect Density (#/acre)	1	5
Placement Quality	Good (3)	Poor (4)
Drain Length (m)	15.25	15.25
Vegetation	Fair (3)	Fair (3)
Drainage Length (m)	304.5	304.5

Table 4-1. Assumptions used in HELP landfill design parameterization

Simulations at the Northampton Landfill were run based on acres present at 5%, 25%, and 33% slopes reported by Dufresne-Henry (2005). A constant slope of 25% was assumed for the Loudeville Rd. and Oliver St. Landfills. RT3D input values are dependent on HELP outputs. Consequently, HELP results are presented below.

The HELP model was simulated for a period of 10 years in order to obtain long-term annual average values for leachate production. Results include acre based estimated conservative annual average hydrologic parameters calculated (Table 4-2). Total values for leachate production are summarized in table Table 4-3. Estimated results leachate production values assuming the landfill liner was built according to stringent guidelines and quality assurance outlined by Shroeder (1997). These values are consistent with values yielded from a HELP simulation conducted by Dufresne-Henry (2005). The conservative value represents leachate production assuming poor construction quality outlined by Shroeder (1997). These values are 6.59 times larger than estimated values indicating that construction quality has a large impact on leachate production.

Assumption	Estimated	Estimated	Estimated
Percent Slope	5	25	33
Head on Barrier (cm)	2.84	0.61	0.48
Leachate (cm/acre)	**0.94**	**0.21**	**0.17**
Leachate (L/acre)	37944.06	8600.01	6885.08
Precipitation (cm)	1091.18	1091.18	1091.18
Runoff (cm)	21.74	22.01	22.04
Evapotranspiration (cm)	52.44	52.47	52.46
Layer 2 Drainage (cm)	34.16	34.60	34.63
	Conservative	**Conservative**	**Conservative**
Percent Slope	5	25	33
Head on Barrier (inches)	3.03	0.71	0.56
Leachate (inches/acre)	**5.04**	**1.42**	**1.18**
Leachate (L/acre)	204050.95	57597.12	47575.61
Precipitation (inches)	1091.18	1091.18	1091.18
Runoff (inches)	18.93	18.96	18.96
Evapotranspiration (inches)	48.95	48.95	48.95
Layer 2 Drainage (inches)	36.38	39.99	40.25

Table 4-2. Acre based hydrologic outputs from HELP simulation

Location	Slope	Acres	Leakage C (cu.ft/d)	Leakage E (cu.ft/d)
Northampton Landfill (Unlined)	5	1	204050.8	37944.2
Northampton Landfill (Unlined)	25	7	403180.0	60200.1
Northampton Landfill (Unlined)	33	6	285453.7	41310.5
		NH Total	**892684.5**	**139454.5**
Loudeville Rd. Landfill (Unlined)	25	5.81	**334639.2**	**49966.0**
Oliver St. Landfill (Unlined)	25	24.65	**1419768.9**	**211990.4**

Table 4-3. Leachate production by landfill

Reactive Transport Model Development:

The reactive transport model simulates the microbially mediated aerobic degradation of landfill leachate. The goal of the model is to predict concentrations of BOD and DO based on constant production of leachate over 50 years. RT3D module 1 which simulates aerobic hydrocarbon degradation and dissolved oxygen concentration was used with an adjusted stoichiometry to simulate BOD. Stoichiometric parameters in RT3D are assigned based on mass of the oxygen constituent over the mass of the carbon constituent. BOD represents consumption of oxygen expressed effectively as negative oxygen resulting in the mass balance equation:

$$BOD + O_2 = 0$$

DO was assigned to all unconfined cells as a function of aquifer recharge (400mm/yr). Recharge concentration was estimated at 10mg/L based on DO results from private wells upgradient from the Northampton Landfill which were consistent with the DO of surface waters. In order to gain a background value of DO under the confined portion of the aquifer, the model was simulated with only DO recharge for 80 years

BOD was assigned to cells occupied by unlined landfills as a function of recharge determined by HELP (Table 4-4). BOD concentration was estimated as 5000mg/L based

Location	Conservative (mm/yr)	Estimated (mm/yr)	Simulated (mm/yr)	Cells
Northampton Landfill	15.75	2.43	10	4
Loudeville Rd Landfill	14.22	2.13	8	1
Oliver St Landfill	14.22	2.13	8	6

Table 4-4. BOD recharge at respective landfills and number of cells assigned at each site.

on values used by Dufresne-Henry (2005) in a similar simulation. Each landfill was assigned recharge concentration cells equaling approximately the landfills area. BOD recharge was assigned only after DO concentrations reached near steady state conditions based on recharge and the simulation was run for 50 years. The simulation assumes that all landfills are at capacity and are capped for the duration.

Results:

Groundwater flow was simulated under steady state conditions based on constant estimated hydrologic and geologic parameters. The model was calibrated to water levels recorded in 17 observation wells throughout the modeling region. Reactive transport of landfill leachate was simulated to assess the impact of leachate degradation on dissolved oxygen concentrations throughout the aquifer. Concentrations were not calibrated to water chemistry data due to lack of long term concentration data from wells in the region. As a result, concentrations represent a prediction of potential impacts based on assumed leachate production values rather than actual concentrations calibrated to field measurements.

Flow Calibration:

Groundwater flow was calibrated to hydraulic heads measured from 17 wells (Figure 5-1). Water level and well depth data were acquired from a variety of wells used in a TCE study, wells used in the previous contaminant transport model (Dufresne-Henry, 2005), and Northampton Landfill monitoring wells (Dufresne-Henry, 1986). Monitoring well density was high in the vicinity of the landfills in order to ensure that flow parameters near the contaminant source were adequately represented in the model. In addition, a number of wells were used the Southeastern portion of the modeling region in order to calibrate groundwater flow in the vicinity of groundwater divides. Precicted Hydraulic heads in the southwestern portion of the modeling region reflect a potentially high degree of error due to insufficient monitoring well data in this area.

Results indicate that the model was able to accurately calculate distribution of head in the modeling region (Figure 5-2, Table 5-1, Table 5-2). Absolute residual mean error in calibration results yielded is 2.58m. Residual mean error is -0.56m. Results indicate that there are no major outliers suggesting overall quality in head calculation. Calibration error in the vicinity of the Southampton Well and Maloney Well indicate that predicted heads are significantly greater than observed heads. Lack of sufficient monitoring wells in the southwestern portion of the model indicates that calibration error in this region may be high. In contrast, heads calculated in the Southeastern

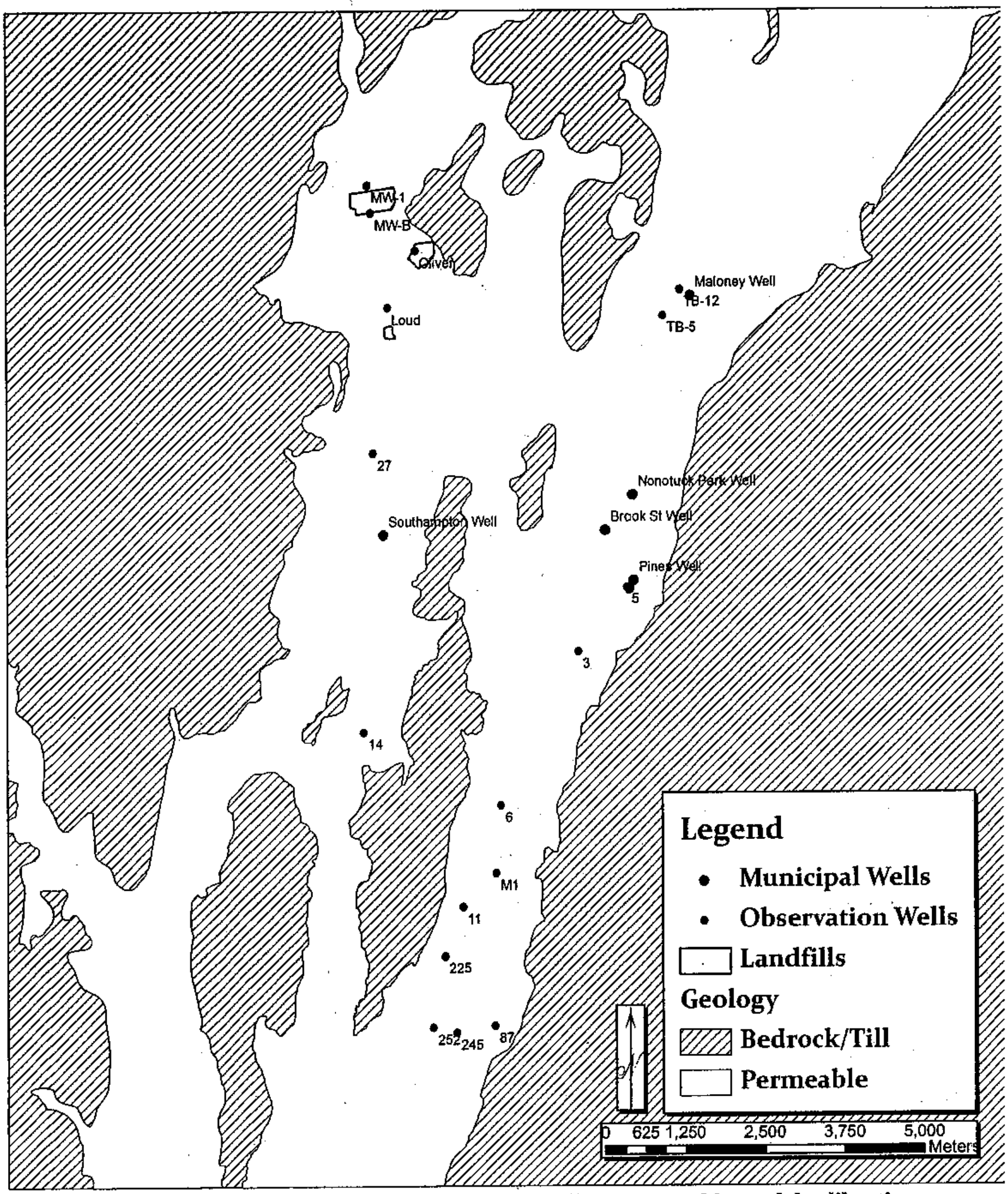

Figure 5-1. Observation wells within the modeling area used in model calibration

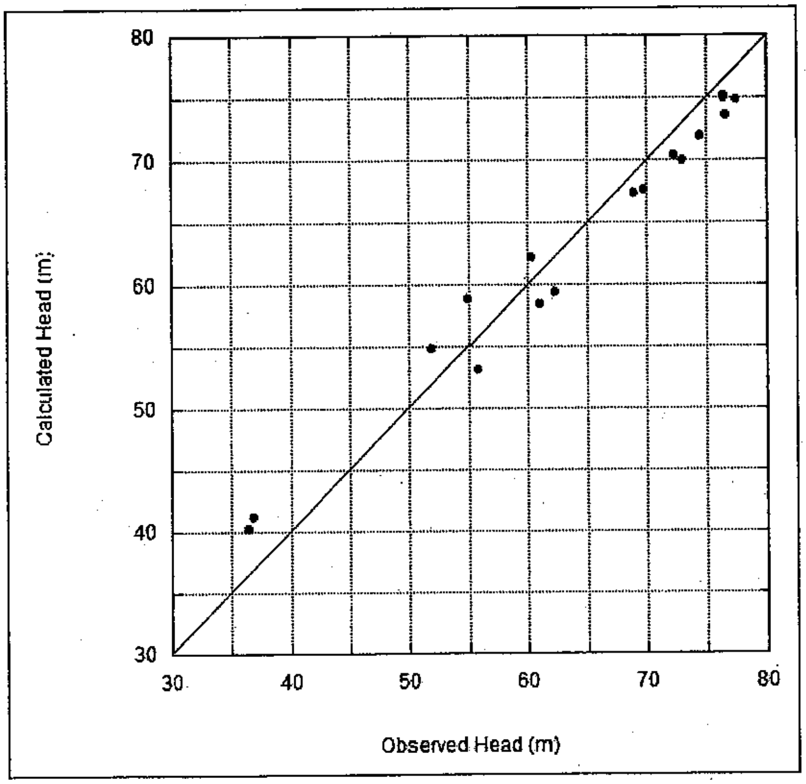

Figure 5-2. Plot of observed vs. predicted heads derived from model simulation

Residual Mean	2.58m
Absolute Mean	-0.556m
Normalized RMS	6.65%
R2	0.99

Table 5-1. Statistical results calculated from observed and predicted heads

Obs Well	Observed Head (m)	Calculated Head (m)	Difference (m)
14	54.90	58.93	4.03
27	51.80	54.90	3.10
11	74.49	72.00	-2.49
225	76.48	73.58	-2.90
245	76.45	75.07	-1.38
252	76.41	75.19	-1.22
3	61.03	58.44	-2.59
5	55.71	53.18	-2.53
6	69.68	67.66	-2.02
87	77.37	74.84	-2.53
LOUD	62.26	59.52	-2.74
M1	72.17	70.37	-1.80
MW-1	72.90	70.05	-2.85
MW-B	68.91	67.30	-1.61
OLIVER	60.30	62.18	1.88
TB-12	36.42	40.25	3.83
TB-5	36.84	41.21	4.37

Table 5- 2. Observed head, calculated head, and error by location

portion of the modeling region are low compared to observed heads. Calibration error over the entire model region is low (5.38%) compared to the overall head change over the modeling region (48m).

Equipotential head values (Figure 5-3) illustrate simulated groundwater flow directions. Calibrated results indicate that flow through the north-south trending bedrock ridge provides a hydrologic connection between the eastern and western lobes of the aquifer. Groundwater from the entire aquifer discharges to the Connecticut River. Losses to rivers cause minor sinks near the Southampton Well particularly on the clay layer boundary. Groundwater divides were successfully simulated in the Southern extent of the model. The western divide was accurately placed in the predicted location. The eastern divide was simulated further south than predicted potentially allowing for greater influx of water do to an increased recharge area. High head gradients were simulated in areas underlain by lower conductivity material. This is particularly the case in the vicinity of the Northampton Landfill and in the far Southwestern portion of the modeling region. High head gradients may be in part due to high bedrock gradients that appear to be the largest control on equipotential head in the aquifer.

Particle tracks show that connectivity through the gap in the bedrock ridge provides a conduit of flow capable of transporting water directly from the landfills to the vicinity of the Maloney Well (Figure-5-4). Particle travel times indicate that groundwater residence from the landfills range from 6570 days at the Loudeville Rd. Landfill to 9125 days at the Oliver St. Landfill (Table 5-3). Low conductivity sediments in the groundwater flow path from the Oliver St. Landfill increase travel time considerably. Groundwater stagnation on the Northeastern side of the bedrock ridge forces particles from the Oliver St. Landfill to move north of the Maloney Well. Particles from the Northampton Landfill and Loudeville Rd. Landfill are relatively unaffected by low conductivity sediments resulting in direct flow to the Maloney Well.

Particle Tracking Results	
Landfill	**Time to Maloney Well**
Oliver St	9125 Days
Loudeville Rd	6570 Days
Northampton	8000 Days

Table 5- 1. Particle tracking times

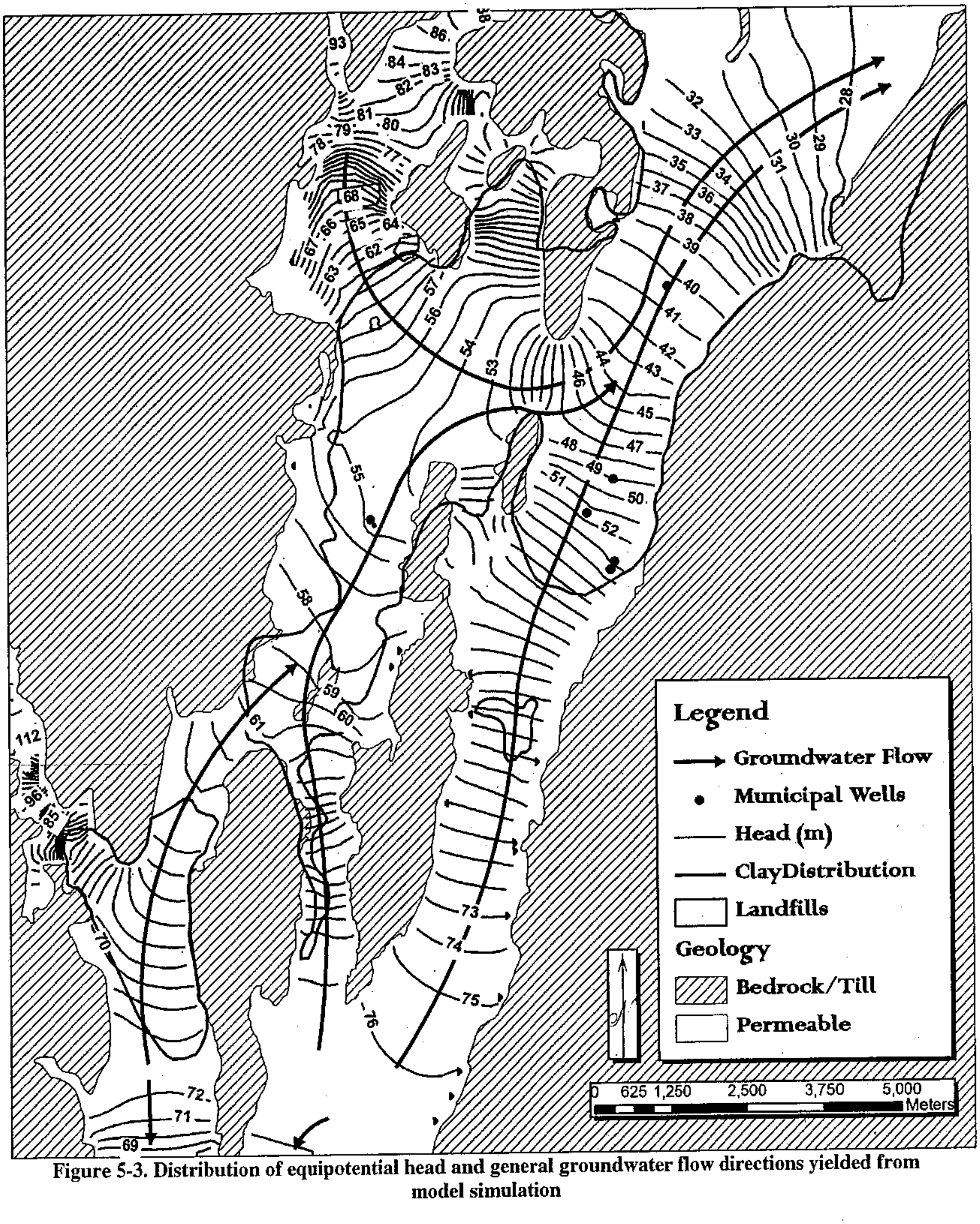

Figure 5-3. Distribution of equipotential head and general groundwater flow directions yielded from model simulation

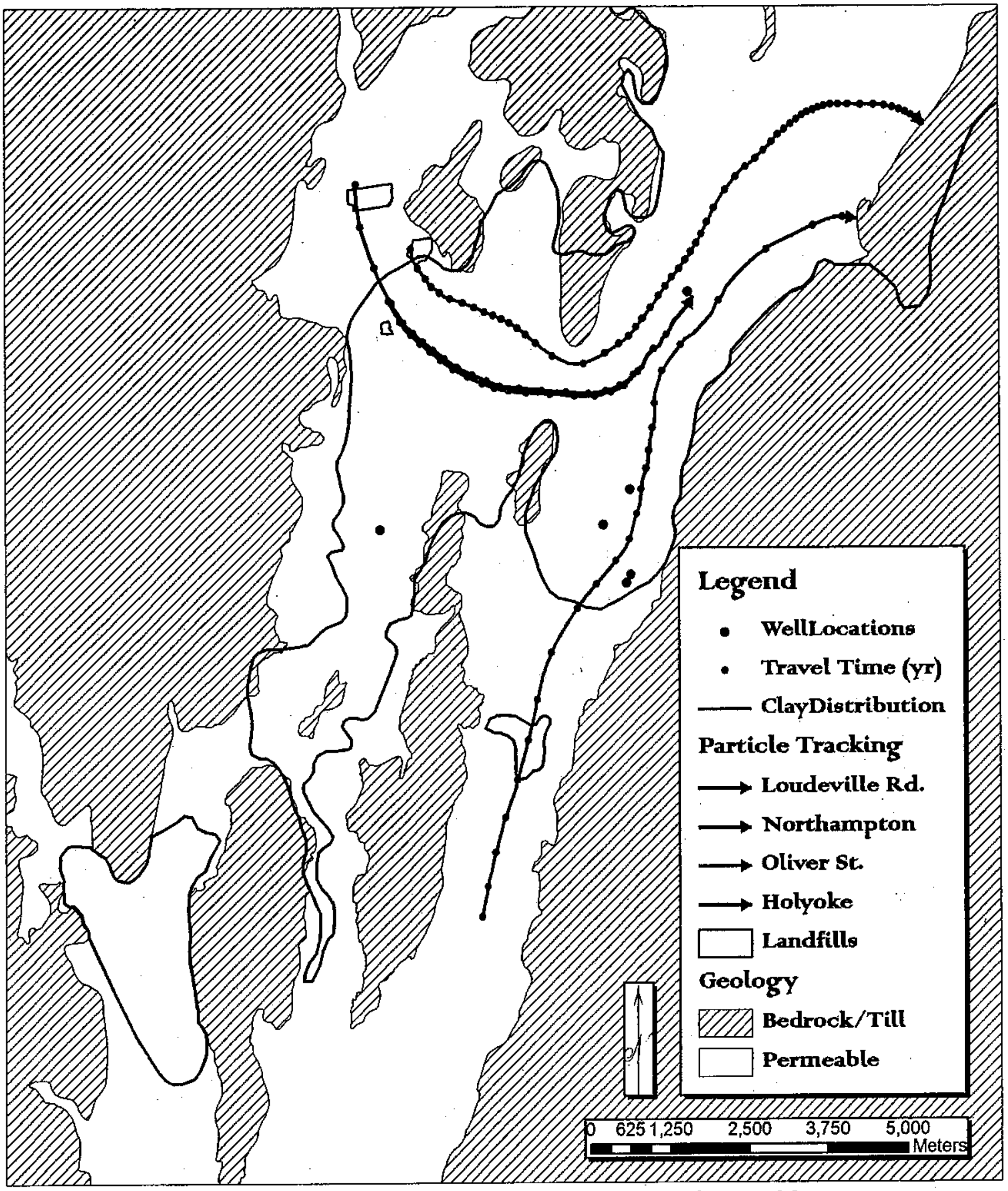

Figure 5-4. Results from particle release from landfills and southern model extent.

Contaminant Transport:

Simulation of DO recharge at approximately steady state shows that DO is distributed throughout the confined and unconfined regions of the aquifer within 80 years. To quantify background DO concentrations DO was simulated as a non-reactive conservative constituent that is transported through dispersion and groundwater flow (Figure 5-5).

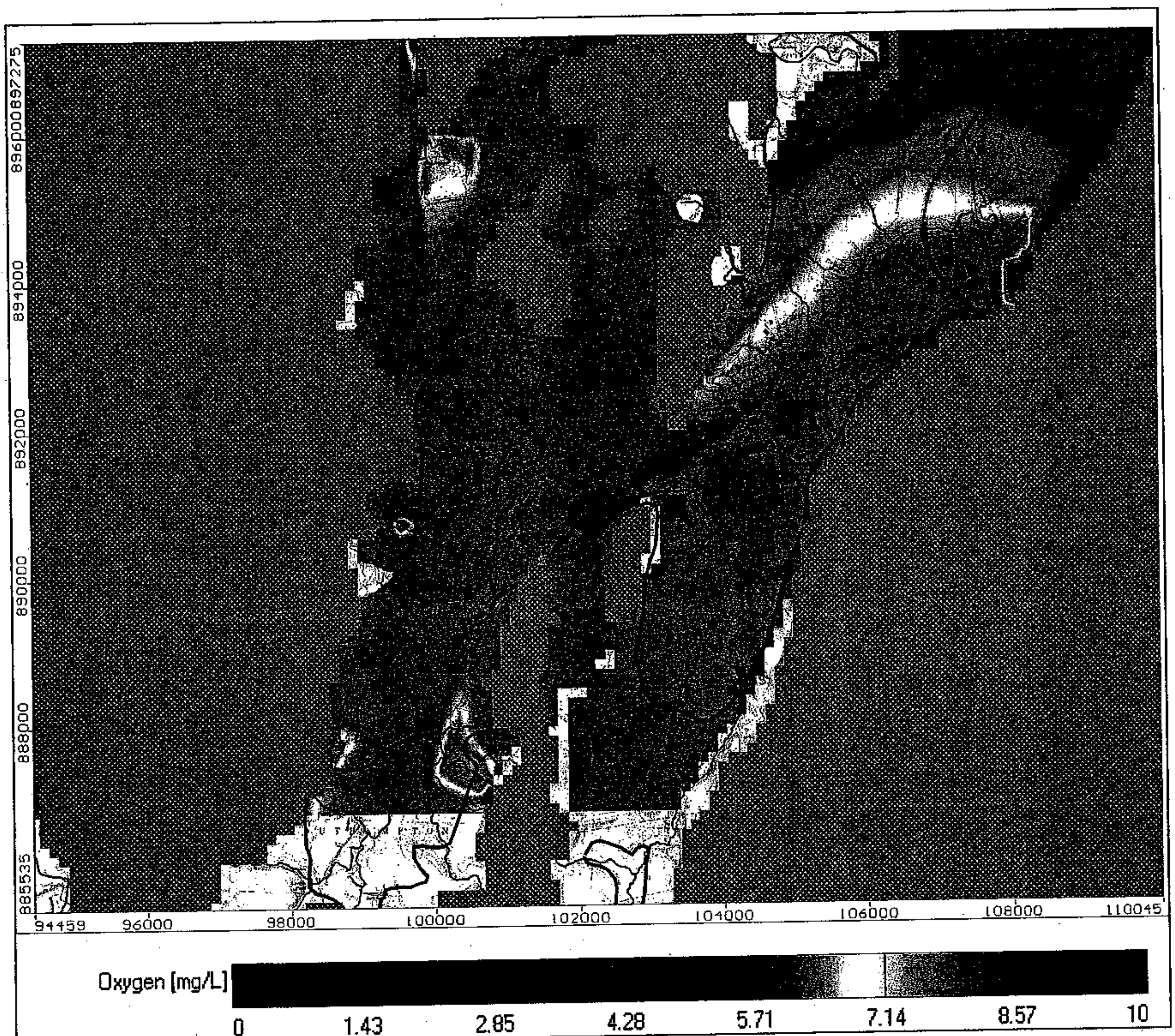

Figure 5-5. Steady state dissolved oxygen concentration

DO reaches a concentration of approximately 10mg/L at the Maloney Well (Figure 5-5). Chemistry results from the Maloney Well suggest that this value is an overestimation of DO at the site. The model shows decreased DO concentrations in areas where groundwater discharges into streams particularly in the Western portion of the modeling region. South of the Southampton Well river leakage produces DO concentrations of 0 mg/L. In confined areas where the equipotential head gradient is low, the model shows decreased DO concentrations due to a reduced influx of DO rich groundwater from unconfined regions. Groundwater stagnation in the northeastern portion of the modeling region causes DO concentrations to drop to 0 mg/L/.

Figure 5-6. Dissolved oxygen concentration after 55 year leachate release

At the assumed leachate recharge rate of 10mm/yr and 9mm/yr for the Northampton Landfill and Loudeville Rd. and Oliver St. Landfills respectively, no significant BOD plume was evident at detectable concentrations in layer 2. The upper layer showed elevated concentrations of BOD consumed within a few hundred meters of all landfills. In contrast, the model shows significantly reduced concentrations of DO extending as a plume from the landfills in the direction of groundwater flow (Figure 5-6).

The most significant plume develops at the Oliver St. Landfill which boasts the largest area and thinnest amount of underlying aquifer. The smallest impact is seen at the Loudeville Rd. Landfill that only consists of 5.81 acres. The model shows the plume continuing through the bedrock gap and impacting water chemistry in the vicinity of the Maloney Well. Though particle tracks show that groundwater from the Northampton Landfill flows directly to the Maloney Well, results from the contaminant model suggest that groundwater flow is diverted slightly to the south before entering the gap. In addition, the contaminant transport model shows that the reduced DO plume flows further north than simulated particle tracks decreasing the impact on the Maloney Well.

Analysis:

Results from the contaminant transport model provide preliminary results regarding hydrologic flow and the fate and transport of landfill leachate in the modeled portion of the Barnes Aquifer. Due to limited existing data, simulated results provide a conservative prediction of groundwater processes and contaminant migration and degradation occurring in the region rather than a representation of actual groundwater flow and contamination. Results indicate a clear hydrologic link between the Maloney Well and the Northampton, Loudeville Rd., and Oliver St. Landfills. Critical analysis of model results illustrates uncertainties in model assumptions and provides insight into where geologic and hydrologic data need to be refined.

Flow Model:

Under the assumed input parameters the groundwater flow model incorporates geologic and hydrologic data to accurately predict the distribution of heads under steady-state conditions. Results simulate a significant amount of flow over a gap in the bedrock ridge separating two major lobes of permeable materials and establish general directions of groundwater flow. Model sensitivity analysis suggests that changes in geologic and hydrologic parameters impact overall calibration quality. However, input parameters used to develop an accurate flow model required a degree of interpretation that is not necessarily supported by geologic data. Model development also required a rough estimation of input parameters that need to be refined in order to develop a more representative simulation.

Groundwater flow simulated provides a conservative prediction of groundwater flow within the context of assessing hydrologic links between the landfill region and the Maloney Well. Calibration indicates that the groundwater flow model presented represents a reasonable prediction of the hydrogeology of the region. However, limited geologic data and calibration points in the Western lobe of the aquifer forced conservative estimations of input parameters based on interpretation of the geomorphic sequence. Clay layer leakage was simulated to remove groundwater sinks due to surface water interactions and lower the heads to match observed heads in Wells 14 and 27. Data in bedrock divide was interpreted to produce a calibration that maximizes the potential flow between the landfill sites and the Maloney Well. As a result, the model demonstrates a potentially exaggerated amount of groundwater flow over the bedrock gap to the Maloney Well. However, due to both limited geologic and hydrologic data and insufficient head calibration points, modeling efforts cannot validate a more moderate interpretation of groundwater flow dynamics without discrediting the realistic potential of the simulated scenario.

Model sensitivity was assessed under modified run conditions in order to increase the probability of a solution to the flow equations. Changes included adjusting the dampening and increasing the convergence criteria in MODFLOW run settings. Increasing hydraulic conductivity resulted in no solution to groundwater flow equations under steady state conditions (Table 6-1). Decreasing hydraulic conductivity increased head calculations in monitoring wells used in model calibration. Decreasing hydraulic conductivity reduces the amount of water transmitted per unit thickness of aquifer requiring an increase in saturated thickness in order to reach steady state conditions. Increasing recharge to 635mm/yr had a similar effect, resulting in a residual mean of 5.65 m. Decreasing recharge resulted in a decrease in calculated average head elevation as aquifer units transmit less groundwater given the same hydraulic conductivity (Table 6-1).

Action	Δ Water Budget (Liters)	Residual Mean (m)	Absolute Res. Mean (m)
K sat. x 10	No Solution	No Solution	No Solution
K sat. / 10	-4506.65	4.16	4.25
Recharge : 635 mm	-44.09	5.65	5.65
Recharge : 300 mm	-65.37	-4.37	5.42
No Clay Layer Leakage	-4253797.5	8.3	8.3

Table 6-1. Results from altering conductivity, recharge, and clay layer leakage in sensitivity analysis

Clay layer leakage simulated in order to accurately calibrate head distribution may be overestimated given the thickness of the confining layer. Leakage through the clay layer was simulated as a negative recharge flux. This had the effect of removing groundwater from the southwestern portion of the aquifer decreasing the elevation of groundwater heads. Based on data used in model development, there is no indication or basis for simulating clay layer leakage at modeled magnitudes. In preliminary runs where no clay layer leakage was simulated error between calculated heads and observed heads was approximately 10 – 15 meters. These results indicate that there is potentially a major conceptual misinterpretation in groundwater flow processes in this region that result in a net imbalance in the water budget producing increased head elevations. A more appropriate conceptual interpretation of flow dynamics in this region could have significant implications on the simulated groundwater flow and contaminant transport and need to be addressed in future groundwater models.

Groundwater flow through the bedrock gap was simulated using conservative assumptions based on limited data. According to a site assessment conducted by Tighe and Bond (1956) in the vicinity of the bedrock divide, exploration wells were drilled into primarily silt and clay and were unproductive. Though these wells provide subsurface geologic data pertaining to a small part of the overall area, they indicate that hydraulic conductivities in this region may be significantly lower than those simulated or that high conductivity sediments may have been deposited as narrow melt-water stream deposits. Either of these scenarios have the potential to significantly affect flow dynamics particularly in regard to groundwater velocity and contaminant transport through this region.

Contaminant Transport:

Contaminant transport modeling reflects an effort to conservatively interpret landfill design parameters to assess the maximum potential impact of landfill leachate on regional geochemistry. However, uncertainties in flow parameters could show that reactive transport of landfill leachate may behave significantly different from the simulated results. In addition, though leachate inputs from the landfill are overestimated, regional sources of DOC, which may have a significant impact on both local and regional groundwater chemistry, are not represented.

Simulated contaminants after 50 years suggest that a plume of depleted dissolved oxygen develops as a result of the degradation of landfill leachate simulated as BOD. Results indicate, elevated BOD developed from landfill leachate rapidly decreases in concentration and is not visible in layer 2 beneath the Northampton and Loudeville Rd. Landfills. However, due to the proximity of the landfills to the confining layer that prohibits recharge of dissolved oxygen, oxygen concentrations do not significantly recover prior to transport to the Maloney Well. Though particle tracks simulated a direct flow between the Northampton and Loudeville Rd. Landfills and the Maloney Well, chemical transport results indicate that the most depleted concentrations of dissolved oxygen are present a few hundred meters to the west of the well. This phenomenon could be attributed to the fact that the majority of the landfill leachate is generated from the Oliver St. Landfill that does not have strong hydraulic connection with the Maloney Well. In addition, aquifer dispersion could impact contaminant transport dynamics simulated. However, it is important to recognize that simulated results represent an estimation of groundwater flow in the region. Slight differences in interpretation of hydrologic or geologic data could produce results that simulate direct link between all landfills and the Maloney Well.

A refined interpretation of the flow model could significantly alter contaminant travel times altering the time scale used to assess impacts of leachate on the Maloney Well. Silt layers observed in 1956 boring logs indicate that flow through the bedrock gap may occur at a much slower rate than anticipated. Hydraulic conductivity for silt range from 10^{-7} to 2×10^{-3} cm/s (Weight and Sonderegger, 2001). Hydraulic conductivity used in this region was 10^{-2} cm/s (Table 3-1). Based on the distribution of unconsolidated materials in this region, groundwater velocity through the divide could be significantly slower, potentially increasing travel time between the landfills and the Maloney Wells. Depending on the magnitude, this could significantly change the conceptual understanding of the impact of landfills on the groundwater chemistry in the Northeastern portion of the modeling region. Impacts from landfill leachate may not be observed in this region for much longer than predicted. Conversely, decreased groundwater velocity from the Landfill to the Maloney Well could increase leachate dispersion allowing for increased degradation and relative DO recharge minimizing the impact on the Maloney Well.

The EPA HELP results indicate that a significant amount of leachate is generated from the unlined landfills. The model excludes leachate production from the lined portion of the Northampton Landfill due to insufficient design data needed to accurately run the simulation. The density of pinhole defects and installation defects simulated showed that poor construction quality produces 6.59 times more leachate than fair construction quality. However, under the context of a conservative simulation, placement defects and pinhole densities were assumed to be high. According to Giroud and Bonaparte (1989), 10 or more installation defects are possible per acre with limited quality assurance. Shroeder (1997) provides a table in the HELP User Documentation suggesting a density of 10 – 20 holes/acre for poor quality landfills. Based on this information a value of 5 holes/acre was simulated in order to capture a moderately conservative estimation of leachate production. More specific design parameters, other than those reported in the Final Contaminant Transport Model (Dufrese-Henry, 2005), could provide details required to more accurately predict leachate production.

Other potential sources of organic carbon to the groundwater system were not represented in the contaminant transport. According to Keimowitz et al. (2005) reducing conditions beneath wetlands has been shown to mobilize arsenic in Saco, Maine. In their study, groundwater chemistry from observation wells in the vicinity of wetlands was shown to have DO beneath wetlands ranged from .63 mg/L to 1.3 mg/L. According to wetlands data layers provided by the EPA, there are an abundance of wetlands in the primary recharge area of the Barnes Aquifer. Simulating DOC generation and reactive transport from wetlands could provide information regarding background concentrations of DO and manganese concentrations at the Maloney Well. Generating a more realistic background concentration of DO in the vicinity of the landfills could significantly alter degradation of leachate generated from landfills.

Conclusion

The contaminant transport model simulates the movement of landfill leachate generated from three unlined landfills through the Barnes Aquifer and assesses the potential impact on a high yield municipal well. The purpose of the model was to determine if existing landfills are affecting the public water supply of Easthampton. Results from the model demonstrate a high degree of hydrologic connectivity between the municipal well and the landfill through a gap in the bedrock ridge that separates the two main lobes of the aquifer. In addition, it indicates that dissolved oxygen concentrations throughout a large portion of the confined aquifer are depleted within the 55 year contaminant transport simulation.

Model was developed to provide information regarding impacts on the mobilization of iron, manganese, and arsenic due to leachate contamination. Issues in coupling results from the reactive transport program (RT3D) with the metals transport program (MT3D) prevented the simulation from providing quantitative information concerning metals mobility. However, based on the knowledge of reducing conditions in the confined portion of the aquifer, there is some indication that landfill leachate could affect iron, manganese, and arsenic concentrations at the Maloney Well. Model refinement and development will continue to provide more conclusive results concerning these parameters.

The model effectively indicated where knowledge and assumptions pertaining to hydrologic processes and geologic information need refinement. Model calibration and sensitivity analysis indicate that site-specific data in some critical areas have a large impact on model quality, hydrologic flow, and contaminant transport. Without a long-term transient simulation that can assess groundwater fluxes in the region, it is impossible to determine whether the steady state calibration is an accurate representation of the hydrologic processes of the Barnes Aquifer. Future work should move towards developing a transient model of the area. In addition, acquiring data concerning flow dynamics in bedrock gap and clay layer leakage needs to be a priority.

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