

Modeling of Nitrate Loading and Transport in the Plymouth Aquifer

by

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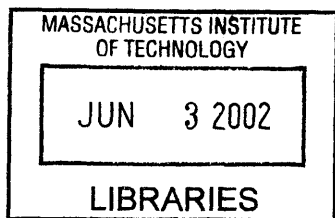
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ABSTRACT

Present study is the simulation of the nitrogen loading from different point and nonpoint sources on the Plymouth aquifer underlying the Eel river watershed and the changes in nitrate concentrations during its transport in subsurface environment.

The flow model, founded on the present USGS flow model for the aquifer, has been the basis for the transport simulation. Using Groundwater Vistas v.3 coupled with RT3D package, the reactive transport of nitrate is modeled. The parameters involved in the advection-dispersion-reaction equation are determined. The nitrogen load from different land-uses in the watershed are estimated as well as organic carbon and dissolved oxygen content of the recharge from various land-uses. The model is calibrated based on the field measurements during the groundwater-monitoring program.

The changes in nitrate concentration in groundwater under future development scenarios including new wastewater treatment facility (WWTF) and Pinehills recreational developments are simulated. The results show that an anaerobic nitrate-contaminated plume will be established underneath the infiltration beds of WWTF. This plume will expand towards Eel River and release a substantial amount of nitrate into the river. As for the Pinehills developments, exceeding the permitted limit for release of nutrients (5 mg-N/L), discharge of typical nutrient load will also inflict considerable amount of nitrogen to the Eel River system and endanger the watershed.

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1 Introduction

The purpose of this study is to assess the subsurface fate and transport of excessive nitrogen load from future developments in the Eel River watershed, Massachusetts.

The reactive transport model is based on the groundwater flow model of the watershed. In chapter 2, this flow model is introduced. The flow model is built using the Plymouth-Carver aquifer model provided by U.S. Geological Survey.

In chapter 3, the transport model is introduced. The nitrogen loads to the aquifer are defined based on the land-uses in the watershed. The advection-dispersion-reaction equation is applied to model the reactive transport. Load factors and effective parameters are defined based on similar studies on nitrogen transport. At last, the present situation simulation is used to verify and calibrate the model.

Chapter 4 represents the results of the transport model under different development scenarios including the new wastewater treatment facility and Pinehills developments. The spatial distributions of nitrate concentration in the groundwater are illustrated and the potential impacts on the water quality in the Eel River are addressed.

The summary of results are given in chapter 5 and strategies that may help maintaining the health of the watershed are represented.

1.1 Nitrogen in Ecosystems

1.1.1 Nitrogen in Nature

Nitrogen is ubiquitous ecosystems. Nitrogen constitutes about 79% of the atmosphere and is an essential element for the growth of organisms. Nitrogen has several oxidation states from -3 to $+5$, appearing in various organic substances as well as variety of inorganic compounds.

The most abundant nitrogenous inorganic compounds (beside nitrogen gas) are ammonia and nitrate. Ammonia, the most reduced form, is a toxic gas, widely used in industry. Ammonia is naturally produced in the process of ammonification of organic nitrogen. The other common form of nitrogen, nitrate, is the most oxidized form of the nitrogen. Nitrate is the final product of nitrogenous organic matter decomposition followed by nitrification under aerobic condition. Nitrate is highly soluble in water and can travel very quickly in the water bodies.

In the last 40 years, the amount of nitrogen entering the groundwater and coastal ecosystems from anthropogenic sources has greatly increased. The dominant sources of anthropogenic sources are fertilizers (more than 50%), atmospheric deposition of NO_x produced in fossil fuel combustion, agricultural sources and wastewater (Pabich, 2001).

1.1.2 Health Concerns

Nitrate contamination of water resources is one of the common problems in various parts of the world. One concern is methemoglobinemia (blue-baby syndrome), which nitrate causes in infants. Cases of infant methemoglobinemia have been reported in the United States. The majority of reported cases were in infants under the age of four months and who were fed milk formulas prepared with contaminated well water.

The methemoglobinemia hazard from drinking water with nitrate-nitrogen occurs when bacteria in the digestive system transform nitrate to nitrite and the nitrite oxidizes iron in hemoglobin of red blood cells to form methemoglobin. Methemoglobin lacks oxygen-carrying capacity and the condition known as methemoglobinemia occurs. Because infants under six months of age have a higher concentration of the digestive system bacteria known to transform nitrate to nitrite, and a lower than normal concentration of the enzyme known to reduce methemoglobin back to hemoglobin, they are at higher risk for methemoglobinemia (Skipton, 1995).

Consuming water from a source containing 10 or less mg-N/L of nitrate provides assurance that methemoglobinemia should not result from drinking water. New York state sanitary

code and USEPA (primary drinking water regulations) constrain the nitrate concentration in drinking water to 10 mg-N/L of nitrate.

Based on a study on stomach cancer mortality rates in different countries, Fine (1982) has proposed a correlation between nitrate intake and gastric cancer. It seems that under certain conditions, nitrate in drinking water reacts with eaten food, resulting in formation of nitrosamines (a carcinogenic matter). Yet, it has not scientifically clear what concentration of nitrate in drinking water causes gastric cancer, if any (Motolenich-Salas, 1997).

Other concerns, related to the environmental impacts resulting from elevated levels of nitrate in surface waters, are discussed in the next section.

1.1.3 Environmental Concerns

The growth of algae in aquatic ecosystems depends on several factors including flow regime of the water body, turbidity, temperature and nutrient supply. Eutrophication is a condition in the aquatic ecosystem where high nutrient concentrations stimulate blooms of algae (e.g., phytoplankton). Although eutrophication is a natural process in the aging of lakes and some estuaries, human activities can greatly accelerate eutrophication by increasing the rate at which nutrients and organic substances enter aquatic ecosystems from their surrounding watersheds. Agricultural runoff, urban runoff, leaking septic systems, sewage discharges, eroded streambanks, and similar sources can increase the flow of nutrients and organic substances into aquatic systems. These substances can over-stimulate the growth of algae, creating conditions that interfere with the recreational use of lakes and estuaries, and the health and diversity of indigenous fish, plant, and animal populations (USEPA, 2002).

Algal blooms cause harm in three ways. First, they cloud the water and block sunlight, causing underwater macrophytes to die. Because these macrophytes provide food and shelter for aquatic creatures (such as the blue crab and summer flounder), spawning and nursery habitat is destroyed, and waterfowl have less to eat when macrophytes die off.

Second, algae bloom restricts the recreational access to water bodies and also causes the formation of algal toxins (i.e. cyanobacteria), including neurotoxins, hepatotoxins, cytotoxins, and endotoxins. These toxins potentially cause gastroenteritis, renal malfunction, allergic reactions, and hepatitis (Herman, 2002).

Eventually, when the algae die and decompose, oxygen is used up. Dissolved oxygen in the water is essential to most organisms living in the water, such as fish and crabs (USEPA, 2002). Thus, this anoxic situation causes inhabitable conditions for that sort of organisms. Another result of anoxic situation in water is increased concentrations of ammonia, iron, manganese, and hydrogen sulfide.

Increased eutrophication as the result of nutrient release from anthropogenic sources (i.e. nitrogen and phosphorous) is one of the major problems facing some estuaries in the mid-Atlantic. Eutrophication has been experienced in the U.S. in several regional resources such as Long Island Sound (NY) and Chesapeake Bay (MD) and also a number of local sites such as Waquoit Bay (MA) and Wellfleet Harbor (MA) (Pabich, 2001).

1.2 Eel River Watershed

The Eel River watershed is less than 15 sq. mi. in area, largely untouched by the impacts of residential development or industry. The USEPA has designated the underlying Plymouth-Carver aquifer as a "sole source" aquifer. Current anthropogenic sources of nutrients in the Eel River watershed include residential developments (septic systems, lawn fertilizer) and agricultural activities (fish hatcheries, cranberry bogs). At current pollution levels, the watershed remains under eutrophication level, but has a "high vulnerability" water quality designation by the USEPA. Future developmental pressures threaten this watershed. A major residential development, golf courses, recreational areas and a new wastewater treatment facility (WWTF) for the town of Plymouth, all jeopardize water quality in the Eel River watershed, due to nutrient pollution (e.g., excess nitrogen and phosphorous introduced into the watershed via anthropogenic processes) (MERIT, 2002).

A Nutrient Technical Advisory Committee (TAC), specifically formed to evaluate nutrient

inputs and their possible impact on the health of the Eel River system, discovered that there is a significant likelihood that additional nutrient concentrations will cause eutrophication. TAC characterizes the western branch of the as mesotrophic and the eastern branch as oligotrophic (TAC, 2000). Thus, unforeseen increase in nitrogen and phosphorous loads to the watershed via the new WWTF and Pinehills developments threaten the watershed with a potentially higher degree of eutrophication.

However, TAC concludes: *“it is unknown if the increase in nitrogen will sufficiently stimulate algal production to harmful levels.”* TAC recommends reducing the phosphorus load to a zero discharge level but places no discharge limit on nitrogen. TAC predicts that future nutrient loading conditions will force the watershed into a phosphorus limiting condition. However, it seems that TAC has overlooked the current nitrogen limiting conditions in the Eel River that will be greatly affected by elevated nitrate concentration (MERIT, 2002).

The Eel River Watershed Association (ERWA), a citizens action group dedicated to the protection of the water quality of the Eel River and its tributaries in Plymouth, Massachusetts, is understandably concerned that the Nutrient Management Plan for the Town of Plymouth, which establishes the guidelines for development within the watershed, is based on incomplete and, in some cases, misinterpreted data. In response, The MIT Eel River Investigation Team (MERIT) is assembled to provide the ERWA with a study of the Eel River watershed. The MERIT study is designed to provide further information concerning the threats to water quality, and recommend mitigation solutions, before potential impacts become a reality (MERIT, 2002).

The soil in Eel River watershed is highly permeable, deep and dry. Because of this highly permeable soil, precipitation infiltrates quickly into the groundwater. Thereby, the hydrology of the watershed is dominated by groundwater. The goal of present study, which is a part of the MERIT study, is to establish a groundwater reactive transport model for analyzing nitrogen fate and transport in the aquifer underlying the Eel River watershed.

2 Flow Model

2.1 Introduction

Modeling of the groundwater flow pattern is the basis for simulation of every solute transport in groundwater. The flow is simulated using MODFLOW package developed by USGS. In this chapter, the hydrologic and hydro-geologic settings of the aquifer, which are the basis for defining the flow model parameters, are represented. The results of the flow model are explained afterwards.

2.2 Hydrologic Settings

Figure 2-1 represents the Eel River watershed. The Eel River system consists of two main branches and several man-made impoundments, including Russell-Mill and Hayden ponds on the northern branch, and Talcott, Forge, Cold Bottom, and Howland ponds on the southern branch. Many of these impoundments were constructed during the manufacturing era of the Eel River watershed. The southwestern portion of the watershed contains a number of kettle ponds including Great and Little South ponds. The kettle ponds have neither inflow nor outflow other than groundwater. The groundwater also recharges the Eel River system, and a small portion of the groundwater is discharged directly to the ocean.

Stream flow of the river was also measured at several locations for the period May 28, 1998 to February 22, 2000. The summary of stream flow measurements is given in Table 2-1 (Herman 2002).

The residence time of the ponds contributes to the potential for eutrophication. The longer residence times the more opportunity for phytoplankton blooms and subsequently the more potential for eutrophication.

Table 2-2 gives the residence time of the ponds in the Eel River system.

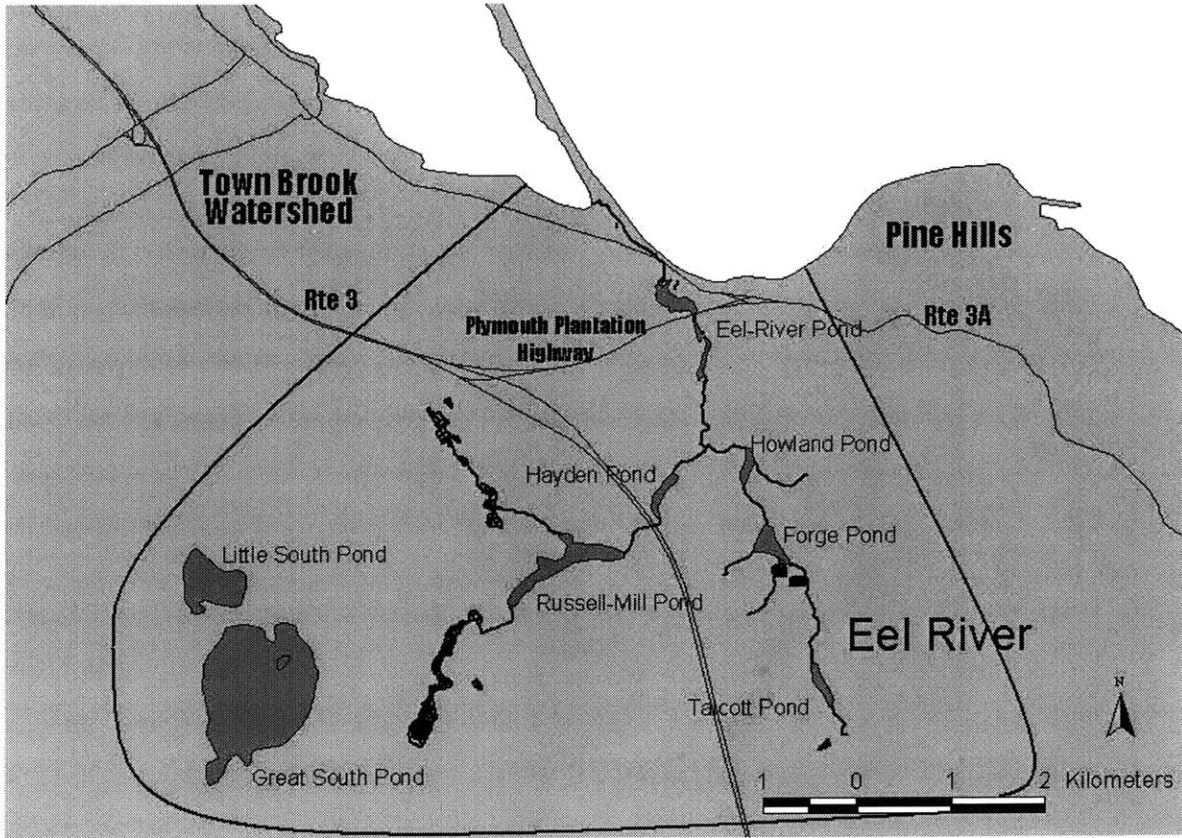


Figure 2-1: Eel River system (MassGIS, 2002)

Table 2-1: Eel River Flow Measurement Summary (Herman, 2002)

Mouth of Eel River		Western Branch						Eastern Branch	
Warren Ave. Bridge		Russell Mill Pond, Outlet		Russell Mill Pond, Inlet		Hayden Pond, Outlet		Howland Pond, Outlet	
Date	Q [cfs]	Date	Q [cfs]	Date	Q [cfs]	Date	Q [cfs]	Date	Q [cfs]
5/28/98	38.26	5/28/98	20.13	5/28/98	6.99	5/28/98	23.61	5/28/98	10.73
6/29/98	31.33	6/29/98	18.01	6/29/98	3.96	6/29/98	19.15	6/29/98	10.26
8/4/98	30.56	8/4/98	6.58	8/4/98	3.99	8/4/98	18.92	8/4/98	7.45
8/25/98	23.83	8/25/98	33.47	8/25/98	4.75	8/25/98	22.75	8/25/98	11.1
9/18/98	29.25	9/18/98	15.22	9/18/98	3.66	9/18/98	17.59	9/18/98	8.52
11/20/98	32.14	11/20/98	15.89	11/20/98	2.91	11/20/98	22.31	11/20/98	10.77
2/24/99	26.22	2/24/99	11.49	2/24/99	1.18	2/24/99	14.92	2/24/99	7.91
6/3/99	22.51	6/3/99	6.73	6/3/99	3.33	6/3/99	14.5	6/3/99	8.89
6/22/99	20	6/22/99	13.8	6/22/99	1.56	6/22/99	9.11	6/22/99	8.34
7/19/99	21.9	7/19/99	10.37	7/19/99	3.01	7/19/99	14.11	7/19/99	5.25
8/18/99	24.37	8/18/99	10.78	8/18/99	2.55	8/18/99	12.19	8/18/99	6.48
9/28/99	22.59	9/28/99	8.82	9/28/99	2.32	9/28/99	12.44	9/28/99	5.81
11/17/99	23.22	11/17/99	20.03	11/17/99	3.38	11/17/99	22.12	11/17/99	6.34
2/22/00	26.48	2/22/00	12.82	2/22/00	1.79	2/22/00	17.86	2/22/00	8.47
MINIMUM	20.00	MINIMUM	6.58	MINIMUM	1.18	MINIMUM	9.11	MINIMUM	5.25
AVERAGE	26.62	AVERAGE	14.58	AVERAGE	3.24	AVERAGE	17.26	AVERAGE	8.31
MAXIMUM	38.26	MAXIMUM	33.47	MAXIMUM	6.99	MAXIMUM	23.61	MAXIMUM	11.10

Table 2-2: The residence times of the ponds in the watershed (TAC, 2000)

Pond	Residence Time (days)
Russell-Mill pond	9
Hayden pond	1
Talcott pond	7
Forge pond	3
Howland pond	2
Eel-River pond	2

2.3 Hydro-geologic Settings

The aquifer underlying the Eel River watershed is part of the Plymouth-Carver aquifer. In 1992, the U.S. Geological Survey established a three-dimensional, finite-difference groundwater model to study the flow in the Plymouth-Carver aquifer. This section represents the hydro-geologic settings of the aquifer, which is used in afore mentioned model.

2.3.1 Geological History

Deposits of fine to coarse sand and gravel with occasional, limited lenses of silt and clay predominantly underlie the Eel River watershed. The subsurface soil consists of the Carver-Gloucester and Carver-Peat soils as classified by the U.S. Department of Agriculture Soil Conversation Service (USDA, 1986).

The surficial geology in the watershed consists of unconsolidated stratified glacial materials deposited during the last retreat of glacial ice about 15,000 years ago. The lower portion of these stratified materials is saturated with directly infiltrated water from precipitation (TAC, 2000).

The soil is highly permeable, deep and dry. This high permeability of the soil explains the important role of the groundwater flow in watershed hydrology. The groundwater locally recharges the Eel River system and regionally discharges to the ocean, flowing outward from topographically high areas in the southwestern portion of the watershed.

The aquifer underlying the Eel River is part of the Plymouth-Carver aquifer, which underlies an area of 140 sq. mi. and is the second largest aquifer in area in Massachusetts. This aquifer is recharged almost entirely from precipitation that averages about 24 inches per year. Water discharges from the aquifer by pumping, evapotranspiration, direct water table transpiration, and seepage to streams, ponds, wetlands, bays and the ocean. In 1985, water use was about 59.6 mgd, of which 82 percent was used for cranberry production (USGS, 1992).

2.3.2 Hydraulic Conductivity

Darcy's law that is the theory that is currently applied in explanation of the fluid (water) movement in a porous medium (soil), introduced in 1856 by Henry Darcy. Darcy's law states that "*The velocity of flow is proportional to the hydraulic gradient.*" Darcy's law can mathematically be expressed as follows (Freeze and Cherry, 1979):

$$q = \frac{Q}{A} = -K \frac{\partial h}{\partial l} \quad \text{Eq. 2-1}$$

Where,

q = specific discharge, volumetric flow rate per unit surface area [L/T]

Q = volumetric flow rate [L^3 / T]

A = surface area perpendicular to flow direction [L^2]

K = hydraulic conductivity [L/T]

$\frac{\partial h}{\partial l}$ = hydraulic gradient [unitless]

Hydraulic conductivity, the ratio of specific discharge to hydraulic gradient in Darcy's law with the units of velocity, depends on properties of both the medium and the fluid and represents the capacity of the medium to transmit the fluid. It can range in value over about 12 orders of magnitude, with the lowest value for unfractured igneous and metamorphic rocks and the highest for value for gravels and some karstic or reef limestones and permeable basalts. The range in hydraulic conductivity within a given rock type is largest for the crystalline rocks and smallest for the sedimentary material (Domenico and Schwartz, 1990).

Hydraulic conductivity is commonly anisotropic in glaciofluvial deposits as the result of the lithologic and textural differences and generally decreases with depth as the mean grain size decreases. It tends to be greater in the horizontal direction than in the vertical direction because of near-horizontal bedding. The ratio of horizontal-to-vertical hydraulic conductivity of coarse-grained deposits is commonly around 10:1, while it could be as great as 1000:1 in fine-grained deposits (Guswa, 1985).

There are several methods to evaluate the hydraulic conductivity. Field-testing, laboratory testing and empirical correlations are common methods to determine the hydraulic conductivity. In the USGS flow model of the aquifer; the hydraulic conductivity values are obtained from 33 aquifer tests conducted for public and industrial supplies, and from lithologic data collected at USGS and public test-well sites.

Average horizontal hydraulic conductivity of stratified sand and gravel deposits, interpreted from aquifer test data, ranges from 16 to 95 m/d with an average of 57 m/d. These hydraulic conductivities are consistent with results of aquifer tests in similar deposits on nearby Cape Cod and Mattapoisett River Valley (USGS, 1992).

Figure 2-2 shows the spatial distribution of the hydraulic conductivity zones in the upper layer of the Plymouth-Carver aquifer.

2.3.3 Porosity

Total porosity is defined as the ratio of void space of the rock (medium) to the total volume of the rock (medium)(Freeze and Cherry, 1979):

$$n = \frac{V_V}{V_T} \qquad \text{Eq. 2-2}$$

where V_V is the void volume and V_T is the total volume.

Porosity may be primary (interstitial) porosity, which corresponds to the void spaces resulting from placement of grains or porous nature of the grains themselves, or secondary porosity, which represents the void spaces resulting from fractures or solution of the rock (Meinzer, 1923).

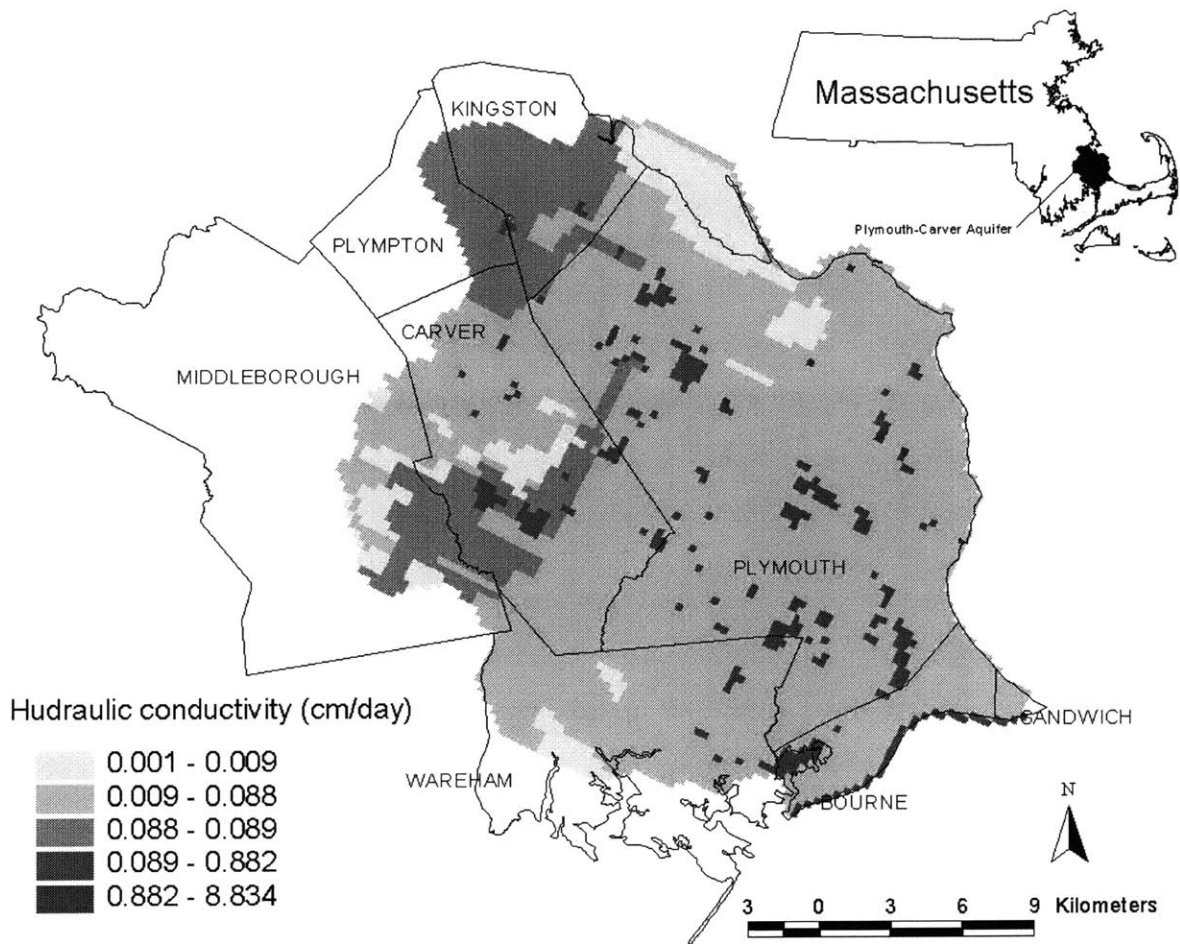


Figure 2-2: Spatial distribution of the hydraulic conductivity (USGS, 1992)

Interstitial porosity can range from about 26 to 47% through different arrangements and packing of ideal spheres (Graton and Fraser, 1935). Total porosity can range from near zero to more than 60% (Domenico and Schwartz, 1990).

The interconnected portion of pore space, which is effective in groundwater flow, is called effective porosity. Effective porosity can be over one order of magnitude smaller than total porosity, with the greatest difference occurring for fractured rocks (Domenico and Schwartz, 1990).

The pore space is the only available space for the groundwater flow. Therefore, to calculate the seepage velocity that is the actual transport velocity of the dissolved matter, the specific discharge rate should be divided by effective porosity.

Based on the conducted studies on the Cape-Cod aquifer, which has the same geological source as Plymouth-Carver aquifer, the effective porosity of this sand and gravel aquifer is set to 0.39 (Garabedian et al, 1991).

2.3.4 Model Boundaries

In the Plymouth-Carver aquifer model, the natural hydrologic boundaries, i.e. streams and topography-based basin boundaries, were used to define the model boundaries. The withdrawals by pumped wells, leakage through streambeds, and discharge to the ocean, were also included in the USGS model.

The steady state flow pattern of the USGS model is used to define the boundary conditions of the groundwater flow simulation in the aquifer underlying the Eel River watershed.

2.3.5 Saturated Thickness

Saturated thickness, the part of the aquifer thickness in which the groundwater flows, is the difference between water table and bedrock elevations, assuming bedrock is impervious.

The saturated thickness of the aquifer can be calculated based on maps of the water table and bedrock surface. The values of saturated thickness are highest along the buried bedrock valley and its tributaries (more than 60 m) and decrease to less than 6 m, along the southwestern boundary of the Plymouth-Carver aquifer (USGS, 1992). In the Eel River watershed, the groundwater table elevation ranges from sea level to 38 m above sea level, with the saturated thickness of the aquifer greater than 45 m in many areas (TAC, 2000).

2.3.6 Confining Units

The aquifer is mostly unconfined, and the water table is at atmospheric pressure and can rise or fall freely as the response to changes in recharge and discharge.

In the coastal area between the Eel River and Kingston, overlying units of glaciolacustrine silt, and clay and underlying bedrock locally confine water in the aquifer (Williams, 1974). These confining units do not appreciably extent inland and are limited to the coastal area. Water in the confined sand and gravel is under artesian pressure; this artesian condition also causes many springs in coastal areas, notably around coastal areas in Town Brook watershed (Figure 2-1) (USGS, 1992).

Drillings at the northern end of the Pine Hills (Figure 2-1) has identified a confining unit of relatively low vertical and horizontal conductivity that probably was deposited in a small, ice-contact glacial lake (USGS, 1992).

2.3.7 Groundwater Recharge

Recharge to the aquifer is primarily from infiltrating precipitation on the stratified glacial deposits and, to a small extent, from infiltration of precipitation runoff from adjacent till deposits in upland areas and recharge from kettle ponds.

Because the soil is very permeable in the area, precipitation can infiltrate quickly and leave very little surface runoff. From continuity stand point, the averaged groundwater recharge rate should be equal to the averaged groundwater discharge rate to Eel River. However, it is obvious that a localized high volume recharge in the lower watershed can create flow under the river to the sea (TAC, 2000).

The effective groundwater discharge, i.e. the amount of groundwater discharging to the Eel River, was monitored over a period of 2 years from 1969 to 1971. The effective groundwater discharge ranged from 19 to 23.2 cfs with the average of 22.2 cfs. Total groundwater discharge can be calculated by adjusting the effective groundwater discharge to account for losses due to evapotranspiration and pumping water supplies. Total groundwater discharge

after adjustment is 24.2 cfs. Dividing this discharge rate (that is the recharge rate) by the area of the watershed, i.e. 13.62 sq. mi., gives a unit area recharge rate of 24.18 in/yr (USGS, 1992).

The actual groundwater recharge may be greater than this estimation due the losses not considered in the calculations such as evapotranspiration from the area close to the ponds where the water table is less than 10 meter deep. In this estimation, it was assumed that the recharge rate through all of the area is the same. However, fine-grained deposits have less recharge rate than the coarse-grained deposits. Thus, the actual recharge rate through the coarse-grained deposits probably ranges from 26 to 28 in/yr. The recharge rate to glacial till is estimated to be approximately 7 in/yr (USGS, 1992).

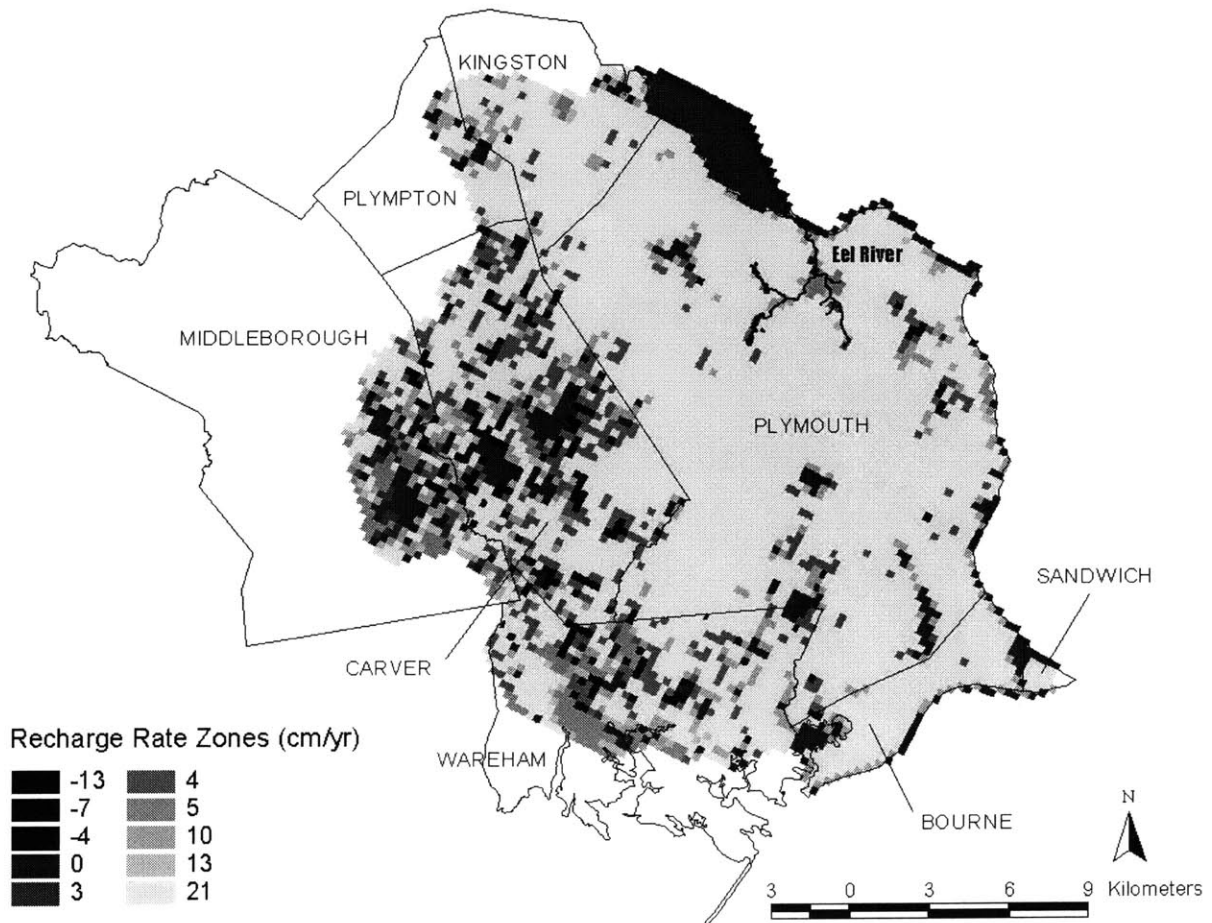


Figure 2-3: Spatial distribution of recharge-zones (USGS, 1992)

Based on current monitoring at the mouth of the river at Warren Avenue, the average annual flow in the Eel River shows a small increase up to approximately 26.62 cfs (Table 2-1). Since the flow model has been calibrated on the basis of 1969-1971 period measurements, and the flow rate in the Eel River has not considerably changed, the original flow rates (estimated in 1969-1971) are maintained in the flow model.

There are 10 different recharge zones in the Plymouth-Carver area. Figure 2-3 shows the spatial distribution of recharge zones in the Plymouth-Carver aquifer.

2.4 Flow Model Results

Based on the information provided in the previous sections, the groundwater flow model in the Plymouth aquifer is established.

Groundwater in the Plymouth-Carver aquifer discharges to the rivers and streams that drain the aquifer, to ponds, wetlands, bogs, and directly to the ocean along the coast. In Eel River watershed, the groundwater discharges to the Eel River and the ocean.

The average discharge to the Eel River is about 27 ft³/s (0.75 m³/s), based on stream flow measurements in the period May 28, 1998 to February 22, 2000 by Camp Dresser & McKee, Inc. (Herman, 2002).

The present flow pattern is shown in Figure 2-4. The water table contour lines and flow directions show that the general direction of groundwater flow in the Eel River watershed is from the Great South pond in the west of the Eel River toward points of groundwater discharge. Eel River greatly influences the configuration of the water table and consequently, the groundwater flow direction.

To be mentioned is the noticeable distortion of this generalized flow pattern caused by large, deep ponds in the watershed, such as Little South pond and Great South pond. These ponds act as zones of very high conductivity within the aquifer. Groundwater discharged into the ponds at their upgradient ends reenters the aquifer at their downgradient ends.

Water table slope (hydraulic gradient) ranges from 5 to 75 ft/mi (0.001 to 0.014). Hydraulic gradient is smaller in areas underlain by very conductive material than in areas underlain by less conductive material.

The spatial variations in seepage velocity are shown in Figure 2-5. The groundwater velocity is the representative of the specific discharge of the groundwater. The flux of any species is the product of the concentration and the specific discharge.

The seepage velocity in the Eel River watershed is changing from less than 1.5×10^{-5} ft/s (40 cm/d) underneath the junction of two branches to 1.5×10^{-4} ft/s (4 m/d) in the south of the Forge pond. The average seepage velocity is about 7.5×10^{-5} ft/s (2 m/d).

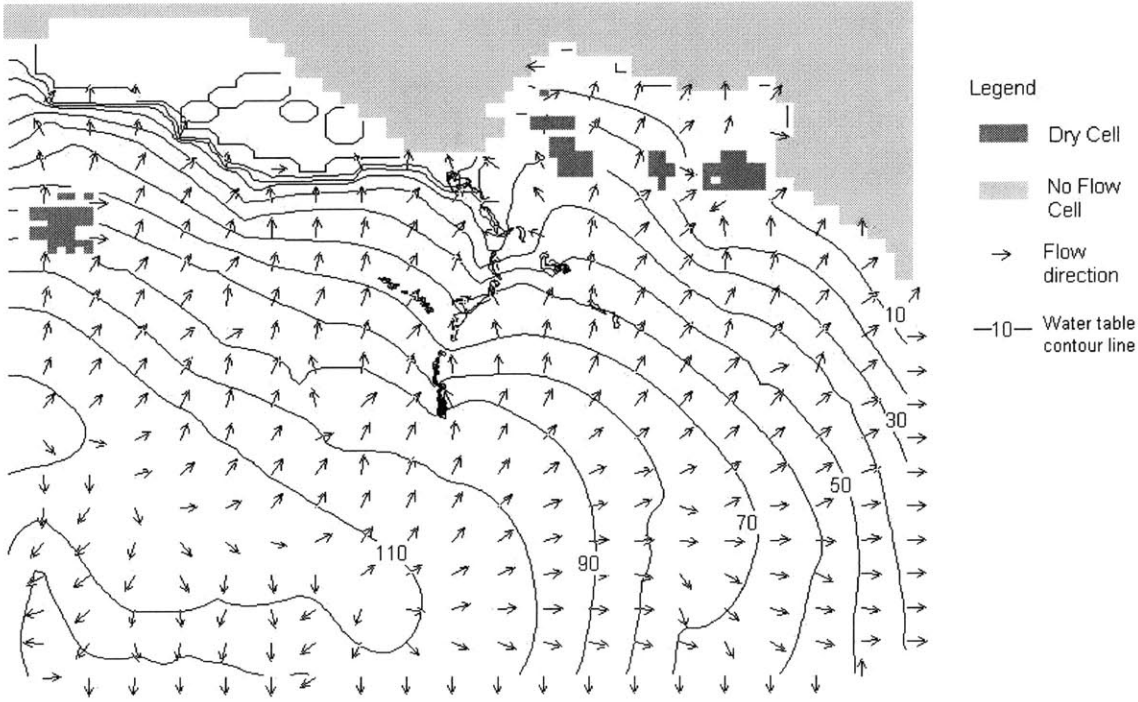


Figure 2-4: Flow pattern, present situation

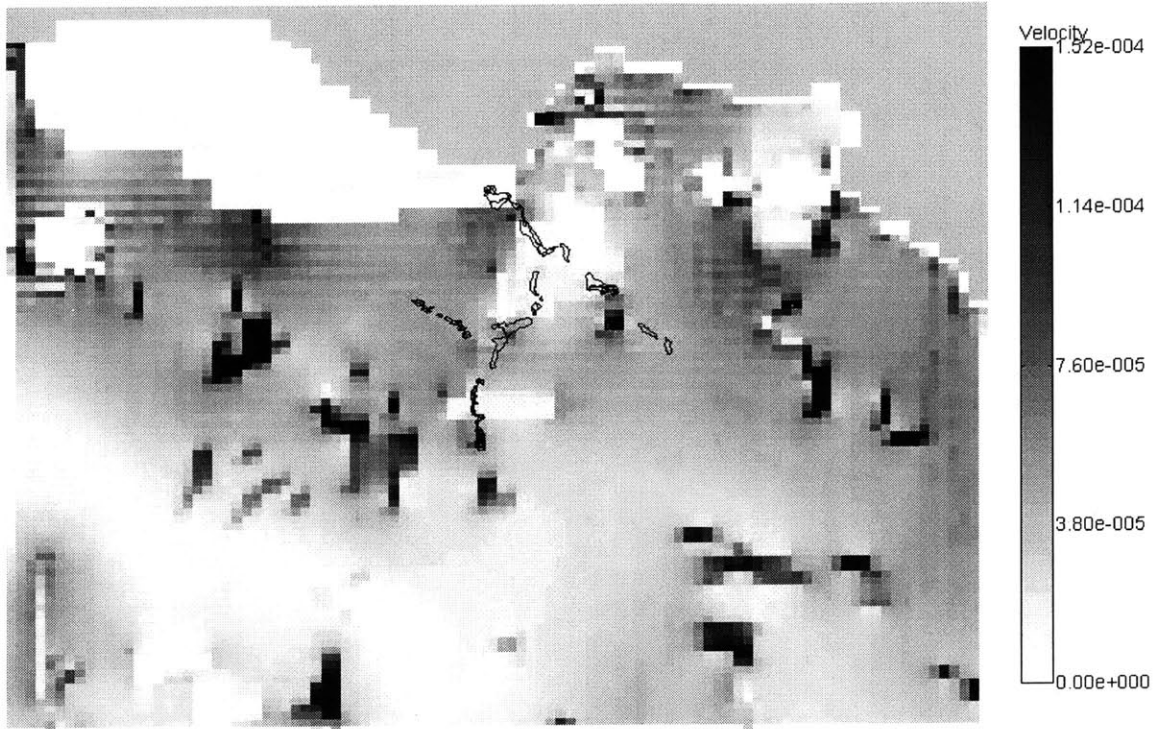


Figure 2-5: Spatial variation in seepage velocity, present situation

3 Transport Model; Setup and Calibration

The reactive transport model is built on the basis of the flow model shown in the previous chapter. Defining the transport parameters and calibration of the model is the next step before simulation of development scenarios.

The present situation simulation is used to calibrate the model. After calibration of the model, different development scenarios are defined. These development scenarios include two operation phases of WWTF and Pinehills developments, which will be represented in the next chapter.

In this chapter first the procedure of setting up the model is explained and then the present situation simulation and model calibration are discussed.

3.1 *Setting up the model parameters*

Every substance, entering the subsurface environment, undergoes several physical, chemical and biochemical processes, which affect the spatial and temporal variations in concentration of the substance. The common approach is to formulate these processes in as the advection-dispersion-reaction equation. This equation states that the total rate of the concentration change at any point of the medium, $\partial C / \partial t$, is the sum of changes due to physical processes (advection and dispersion) and internal sources-sink processes.

The advection term can be expressed in term of seepage velocity of the groundwater (\vec{v}) and concentration gradient ($\vec{\nabla} C$). The mathematical formulation of dispersion/diffusion term is the spatial gradient of the dispersion flux ($D \cdot \vec{\nabla} C$). On the other hand, the internal source-sink process includes all the chemical and biochemical processes, which transform the substance (r).

The overall advection-dispersion-reaction equation in three dimensions can be expressed as following (Hemond and Fechner-Levy, 2000):

$$\frac{\partial C}{\partial t} = -\vec{v} \cdot \vec{\nabla} C + \vec{\nabla} \cdot (D \vec{\nabla} C) + r \quad \text{Eq. 3-1}$$

Where,

$$\vec{\nabla} = \frac{\partial}{\partial x} \cdot \hat{i} + \frac{\partial}{\partial y} \cdot \hat{j} + \frac{\partial}{\partial z} \cdot \hat{k} \quad \text{Eq. 3-2}$$

In the following section, each of these terms is explained.

3.1.1 Advection

The concentration of the chemical may spatially vary in the medium (aquifer), and this concentration gradient along with the flow in medium (groundwater flow) causes the temporal change in the concentration at the point of interest. This process is called advection (Hemond and Fechner-Levy, 2000).

The flow model provides the transport model with the spatial distribution of the hydraulic head and flow pattern.

3.1.2 Dispersion/Diffusion

The coefficient of hydrodynamic dispersion (D) is the sum of the coefficients of bulk diffusion (D_d) and mechanical dispersion (D_m). The coefficient of bulk diffusion can be estimated reasonably within an order of magnitude for granular media, based on the laboratory experiments. The coefficient of mechanical dispersion is controlled by seepage velocity and medium grain size. As the seepage velocity increases, the mechanical dispersion dominates the diffusion process and the molecular diffusion coefficient becomes negligible (Domenico and Schwartz, 1990).

The effect of medium grain size is represented by dispersivity of the medium (α). In the anisotropic medium, the coefficients of mechanical dispersion in three dimensions are formulated as following:

$$D_L = \alpha_L \cdot v \quad , \quad D_H = \alpha_H \cdot v \quad \text{and} \quad D_V = \alpha_V \cdot v \quad \text{Eq. 3-3}$$

Where,

' D_L ', ' D_H ' and ' D_V ' are respectively longitudinal, transverse horizontal and transverse vertical hydrodynamic dispersion coefficients [L^2/T],

' α_L ', ' α_H ' and ' α_V ' are respectively longitudinal, transverse horizontal and transverse vertical dispersivities of the medium [L],

and ' v ' is the groundwater seepage velocity [L/T].

The efforts to measure the dispersivity show that longitudinal dispersivity increases infinitely with scale. This process, called macro-dispersion indicates that heterogeneity at the macroscopic scale contributes significantly to dispersion, because it creates local-scale variability in velocity. The resulted longitudinal dispersivity is about one hundredth of the scale (plume size). And the transverse horizontal and vertical dispersivities are respectively, one and two order of magnitude smaller than longitudinal dispersivity. All in all, field tracer tests are the common means to estimate the dispersivity coefficients of the aquifer (Gelhar, 1992).

Since the Plymouth-Carver aquifer is quite similar to the sand-and-gravel aquifer underlying Cape Code, the dispersivity values from a study on the Cape Code aquifer are used for this model; longitudinal dispersivity of 0.96 m, transverse horizontal dispersivity of 1.8 cm and transverse vertical dispersivity of 1.5 mm.

3.1.2.1 Sink and Source

Sink and source term (r) accounts for internal production or decomposition of the chemical. There are different mathematical approaches for modeling these processes. Selection of the appropriate model depends on the nature of the ongoing chemical and biochemical reactions. The external production of the chemical (i.e. dissolved in the recharge) is also included in this term.

3.1.2.1.1 Nitrogen sources

The major sources of nitrogen in water bodies include wastewater septic systems, fertilizers, sewer exfiltration, storm water and effluents from wastewater treatment plants. During its travel through the unsaturated zone, nitrogen transforms from organic nitrogen and ammonia to nitrate.

The nitrogen content of septic effluent mainly consists of ammonia, nitrate and organic nitrogen (particulate and dissolved). During infiltration through the unsaturated zone, nitrogen undergoes several processes including ammonification (DON), nitrification (ammonia), assimilation (ammonia and nitrate), sorption (ammonia), filtration (PON), volatilization (ammonia) and denitrification (nitrate). 75% of the nitrogen content of the effluent, which reaches the water table, mainly consists of nitrate and ammonia (93%). Figure 3-1 shows these processes (Desimone, 1998). The nitrogen load from other sources is also mainly in the form of nitrate by the time it reaches the groundwater.

3.1.2.1.2 Heterotrophic denitrification

Nitrate is one of the stable forms of nitrogen. There are natural processes, which attenuate the nitrogen concentration, such as dilution and plant uptake. The most important process, which results in nitrate removal from groundwater, is heterotrophic denitrification.

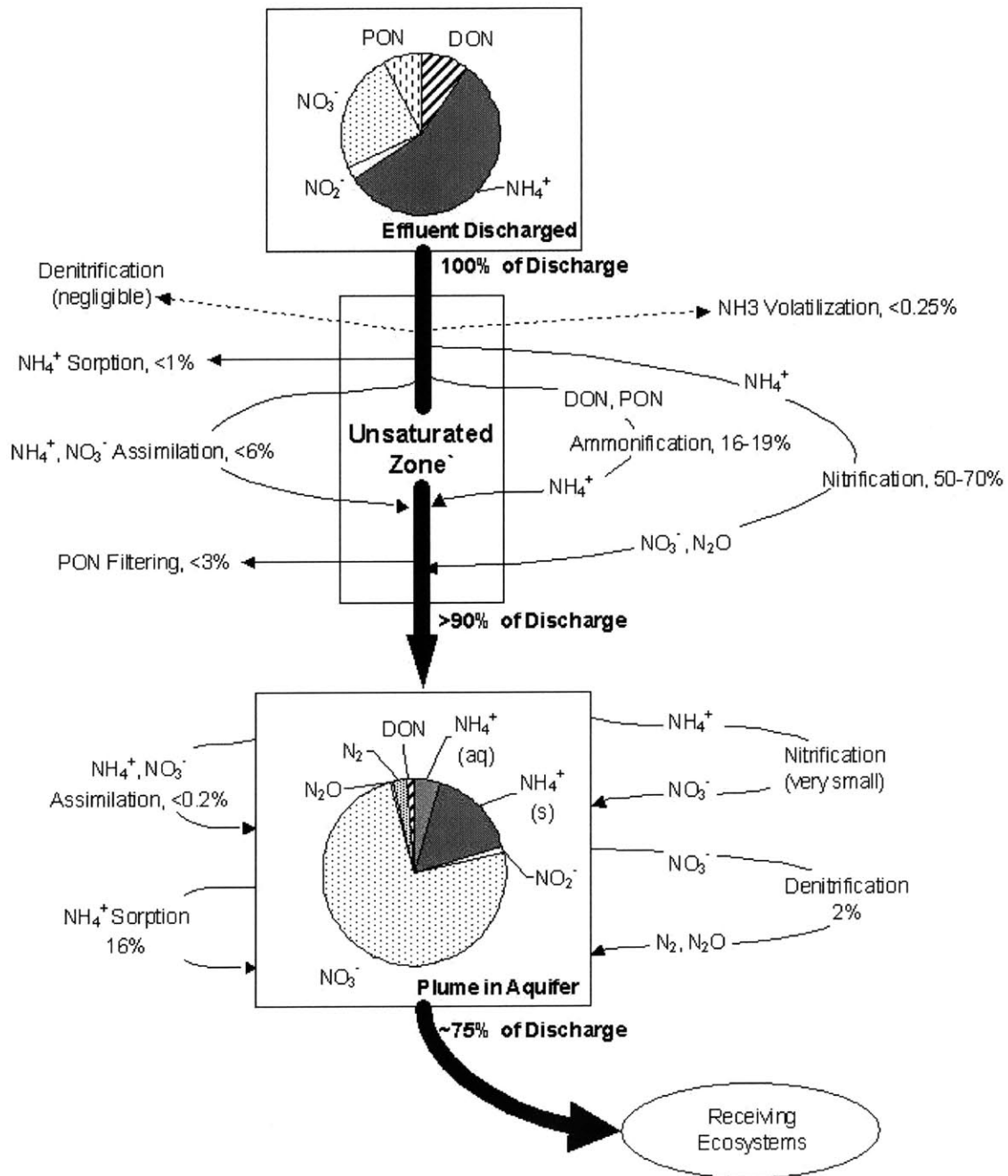


Figure 3-1: Nitrogen transformation in unsaturated zone (Desimone, 1998)

Denitrification is a complex biological process performed primarily by ubiquitous facultative heterotrophs. In denitrification, nitrate is the electron acceptor and the organic carbon serves as the common electron donor. After oxygen, nitrate is the most energetically favorable common e^- acceptor in ecosystems. In the presence of oxygen, denitrification rates are very small. At concentrations above 0.2 mg/L, DO inhibits the reductases required to catalyze the reactions. However, denitrification process may still proceed. At higher concentrations, 2.5 to 5 mg/L, oxygen suppresses several nitrogen reductase genes and as the result denitrification halts. At these higher concentrations, the nitrate reductase and nitrous oxide reductase are repressed before the nitrate reductase is affected, resulting in production of NO_2 and N_2O ; both gases are known effective in the atmospheric greenhouse problem (Johnson, 2002).

To provide an anoxic (or suboxic) environment in favor of denitrification, organic carbon is very important. Aerobic decay of organic carbon, consuming the dissolved oxygen, provides the anaerobic condition necessary for denitrification. However, organic carbon is required for denitrification itself. Therefore, it is important to keep account of the dissolved oxygen and organic carbon as well as nitrate concentration.

However, other conditions are required for denitrification to begin; including a viable population of denitrifying bacteria, sufficient concentrations of nitrogen oxides (NO_3^- , NO_2^- , NO and N_2O) as intermediate electron acceptors (Pabich, 2001).

3.1.2.1.3 Mathematical Model for Reactions

The biochemical processes that result in nitrate removal from the groundwater are in fact the organic carbon decay processes. First the organic carbon is oxidized by available dissolved oxygen in the groundwater. This aerobic decay of organic carbon, which highly depends on dissolved oxygen concentration, can be formulated as follows (Clement T. P., 1998):

$$r_{HC,O_2} = -k_{O_2} [HC] \frac{[O_2]}{K_{O_2} + [O_2]} \quad \text{Eq. 3-4}$$

Where,

$[HC]$ and $[O_2]$ are, respectively, concentrations of organic carbon and dissolved oxygen in the groundwater $[ML^{-3}]$,

r_{HC,O_2} is the rate of oxidation of organic carbon by oxygen $[ML^{-3}T^{-1}]$,

k_{O_2} is the first order rate constant for aerobic decay of organic carbon $[T^{-1}]$, and

K_{O_2} is the Monod half saturation constant $[ML^{-3}]$.

After depletion of oxygen, denitrification starts. Denitrification rate depends on the organic carbon concentration as well as nitrate concentration. As mentioned before, denitrification rate decreases drastically in presence of oxygen. Clement T. P., 1998 proposes that the rate of organic carbon decomposition, caused by denitrification, can be formulated as follows (Clement T. P., 1998):

$$r_{HC,NO_3} = -k_{NO_3} [HC] \frac{[NO_3]}{K_{NO_3} + [NO_3]} \frac{K_{i,O_2}}{K_{i,O_2} + [O_2]} \quad \text{Eq. 3-5}$$

Where,

$[NO_3]$ is the concentrations of nitrate (as nitrogen) in the groundwater $[ML^{-3}]$,

r_{HC,NO_3} is the rate of oxidation of organic carbon by nitrate $[ML^{-3}T^{-1}]$,

k_{NO_3} is the first order rate constant for denitrification $[T^{-1}]$,

K_{NO_3} is the Monod half saturation constant for nitrate $[ML^{-3}]$, and

K_{i,O_2} is the oxygen inhibition constant $[ML^{-3}]$.

Rates of electron acceptor utilization can be expressed as the corresponding rate of organic carbon oxidation multiplied by the appropriate yield coefficient (Y):

$$\frac{d[O_2]}{dt} = Y_{O_2/HC} r_{HC,O_2} \quad \text{Eq. 3-6}$$

$$\frac{d[NO_3]}{dt} = Y_{NO_3/HC} r_{HC,NO_3} \quad \text{Eq. 3-7}$$

In the next section, these parameters are estimated.

3.1.2.1.3.1 Half-saturation Concentrations

Different half saturation coefficients for nitrate (K_{NO_3}) are estimated in different studies. The value of 0.66 mg/L is assigned to this parameter, based on the study by Hooker et al, (1994) and Peyton (1994) from batch kinetic data, and the value is consistent with the other reported values (Clement, 1997).

There are also different half saturation coefficients for oxygen used in different studies. A multi-species transport model was established in Kassel University, Germany, to describe the interaction of oxygen, nitrate, organic carbon and bacteria. This model is very similar to the model that is used in the present study. Therefore, the value of 0.2 mg/L is specified for oxygen half saturation coefficient (Kinzelbach and Schafer, 1991).

3.1.2.1.3.2 Oxygen Inhibition Constant

The oxygen inhibition constant is the DO concentration below which denitrification will start. In other words, it is the threshold DO concentration above which denitrification stops.

Different values of K_{i,O_2} are used in different studies. Based on EPA documentations for surface water quality modeling, the value of 0.1 mg/L is set for oxygen inhibition constant (EPA, 1985).

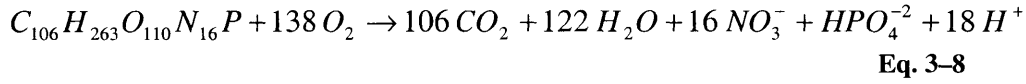
3.1.2.1.3.3 Reaction Rates

The reaction rate is highly dependent on the environmental factors such as temperature and pH. To have a good estimate of the reaction rates, site-specific experiments should be performed, which are beyond the scope of this study. Therefore, the reaction rates are estimated based on literature values and research on similar environments. The value of 1.5 d^{-1} is estimated for both aerobic decay and denitrification rate constants (Kinzelbach and Schafer, 1991).

3.1.2.1.3.4 Yield Ratios

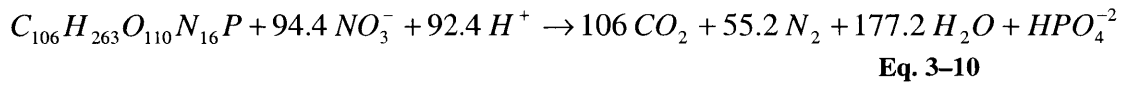
The aerobic decay of the biomass is a biochemical reaction. To define the ratio of required oxygen to decompose one gram of organic carbon, the general formulation of the organic

matter is used. The following reaction is the simplified representation of the aerobic decay of the organic matter.



$$Y_{O_2/HC} = \frac{O_2}{HC \text{ as } C} = \frac{138 \times 32}{106 \times 12} = 3.47 \quad \text{Eq. 3-9}$$

The denitrification process is also a biochemical reaction and can be simplified as following:



$$Y_{NO_3/HC} = \frac{NO_3 \text{ as } N}{HC \text{ as } C} = \frac{94.4 \times 14}{106 \times 12} = 1.04 \quad \text{Eq. 3-11}$$

These yield values relate the mass of decomposed organic carbon to required mass of oxidant, which is the dissolved oxygen in aerobic decay and nitrate in denitrification.

The overall mass balance equations can be expressed as follows:

$$\begin{aligned} -\bar{V} \cdot \bar{\nabla}[HC] + \bar{\nabla} \cdot (D\bar{\nabla}[HC]) + [\text{carbon input by recharge}] - k_{O_2}[HC] \frac{[O_2]}{K_{O_2} + [O_2]} \\ - k_{NO_3}[HC] \frac{[NO_3]}{K_{NO_3} + [NO_3]} \frac{K_{i,O_2}}{K_{i,O_2} + [O_2]} = 0 \end{aligned} \quad \text{Eq. 3-12}$$

$$\begin{aligned} -\bar{V} \cdot \bar{\nabla}[O_2] + \bar{\nabla} \cdot (D\bar{\nabla}[O_2]) + [\text{oxygen input by recharge}] \\ + Y_{O_2/HC} \left(-k_{O_2}[HC] \frac{[O_2]}{K_{O_2} + [O_2]} \right) = 0 \end{aligned} \quad \text{Eq. 3-13}$$

$$\begin{aligned} -\bar{V} \cdot \bar{\nabla}[NO_3] + \bar{\nabla} \cdot (D\bar{\nabla}[NO_3]) + [\text{nitrate input by recharge}] \\ + Y_{NO_3/HC} \left(-k_{NO_3}[HC] \frac{[NO_3]}{K_{NO_3} + [NO_3]} \frac{K_{i,O_2}}{K_{i,O_2} + [O_2]} \right) = 0 \end{aligned} \quad \text{Eq. 3-14}$$

3.2 Present Situation Simulation

3.2.1 Concentrations in the recharge

3.2.1.1 Base Concentrations

The species concentrations for different recharge zones are defined based on the land-use data. In Table 3-1, different land-uses underlain by the aquifer are shown. There are 21 different land-use categories, which are aggregated from 104 categories originally defined in 1971. The most recent updated data in 1999 are applied in this study (MassGIS, 2002).

Table 3-1: Land-use category definitions (MassGIS, 2002)

Code	Category	Definition
1	Cropland	Intensive agriculture
2	Pasture	Extensive agriculture
3	Forest	Forest
4	Wetland	Nonforested freshwater wetland
5	Mining	Sand, gravel & rock
6	Open Land	Abandoned agriculture, power lines, areas of no vegetation
7	Participation Recreation	Golf, tennis, playground, skiing
8	Spectator Recreation	Stadiums, racetracks, fairgrounds, drive-ins
9	Water Based Recreation	Beaches, marinas, swimming pools
10	Residential	Multi-family
11	Residential	Smaller than ¼ acre lots
12	Residential	¼ - ½ acre lots
13	Residential	Larger than ½ acre lots
14	Salt Wetland	Salt marsh
15	Commercial	General urban, shopping center
16	Industrial	Light & heavy industry
17	Urban Open	Parks, cemeteries, public & institutional green-space, vacant undeveloped land
18	Transportation	Airports, docks, divided highway, freight, storage, railroads
19	Waste Disposal	Landfills, sewage lagoons
20	Water	Fresh water, coastal embayment
21	Woody Perennial	Orchard, nursery, cranberry bog

The spatial distribution of the land-use categories, gathered from MASSGIS website, are shown in Figure 3-2. As can be seen, the area is mainly undeveloped and the forest and open-land land-use are the predominant land-use categories in the Eel River watershed area. The developed parts of the watershed are around Plymouth harbor. Providing the northward groundwater flow direction in that area (Figure 2-4) the recharges from these areas discharge directly to the harbor and have a minor effect on the groundwater quality in Eel River.

Any of the 10 different recharge zones (Figure 2-3) consists of different land-use categories. To specify the concentration of each species in the recharge, the mass load from that land-use should be divided by the recharge rate. This resulted in 113 different zones of recharge; each of them has a unique combination of the recharge rate and the species concentrations.

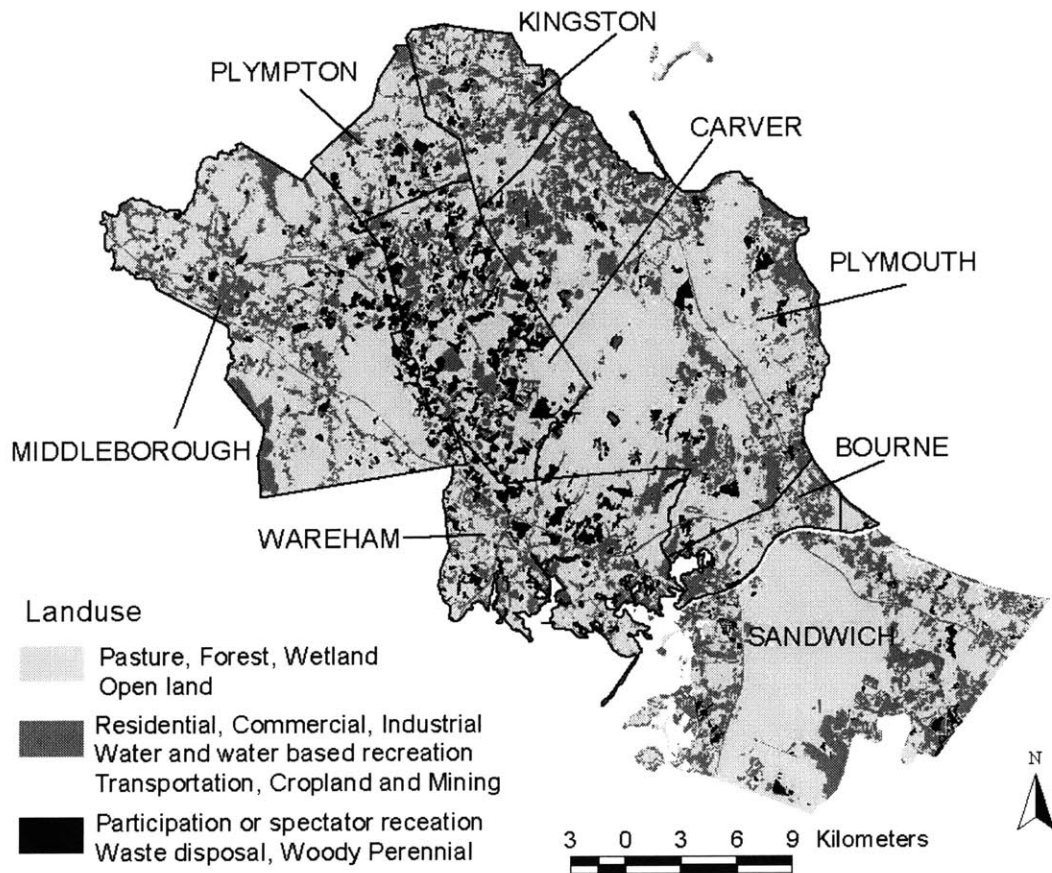


Figure 3-2: Spatial distribution of land-use categories (MassGIS, 2002)

The nitrogen concentrations in the recharge from the flow model are defined based on the load values estimated by the technical advisory committee. Table 3-2 demonstrates the load values of total dissolved nitrogen for different land-use categories.

In estimation of the concentration of organic carbon in the recharge, the typical ratio of organic nitrogen to organic carbon in the wastewater is used. This ratio is given in Table 3-3.

To evaluate the typical ratio of organic nitrogen to total nitrogen, the typical ratio of 30% is used (Desimone, 1998). Table 3-2 shows the resulting organic carbon loads from different land-uses.

Table 3-2: Load values for different land-use categories (in part from TAC, 2000)

Code	Category	TDN (kg/ha.yr)	DOC (kg/ha.yr)
1	Cropland	8.5	26.01
2	Pasture	5	15.3
3	Forest	0.6	1.83
4	Wetland	3	9.18
5	Mining	14.8	45.3
6	Open Land	0.6	1.83
7	Participation Recreation	38	116.34
8	Spectator Recreation	38	116.34
9	Water Based Recreation	11.1	33.99
10	Residential	9.5	29.07
11	Residential	9.5	29.07
12	Residential	9.5	29.07
13	Residential	9.5	29.07
14	Salt Wetland	3	9.18
15	Commercial	15.1	46.23
16	Industrial	15.1	46.23
17	Urban Open	5	15.3
18	Transportation	15	45.9
19	Waste Disposal	34	104.07
20	Water	11.1	33.99
21	Woody Perennial	23	70.41

Table 3-3: Typical unit loading factors from individual residences in US (Metcalf & Eddy, 2002)

Component	Range	Typical value
TKN	9 – 21.7 (g/d.Capita)	13.3 (g/d.Capita)
TOC	80 – 192 (g/d.Capita)	136 (g/d.Capita)
N/TOC	0.047 – 0.271	0.098

To calculate the concentrations of TOC and TDN in the recharge, the load values are divided by recharge rates of the flow model.

The base recharges, which are mainly from precipitation, are oxygen-saturated. Therefore, DO concentrations in these recharges are set to 12 mg/L.

These concentrations are based on precipitation recharge and need to be modified to account for the septic effluent.

3.2.1.2 Modified Concentrations

The concentrations in the precipitation-based recharges should be modified to account for the effect of the septic effluents from residential land-uses.

To do this, the recharge from septic systems is calculated based on the number of households in one acre, 3.2 people per household, and the typical septic effluent of 55 gpd/capita. Table 3-4 shows the resulted recharge rates from septic systems for different residential land-uses.

Table 3-4: Septic effluent rate from different residential land-uses

Code	Description	Families/acre	People/acre	Flow (gpd/acre)	Flow (ft/s)
10	Multi-family	8	25.6	1408	4.32×10^{-3}
11	Smaller than ¼ acre lots	4	12.8	704	2.16×10^{-3}
12	¼ - ½ acre lots	3	9.6	528	1.62×10^{-3}
13	Larger than ½ acre lots	2	6.4	352	1.08×10^{-3}

The typical values for DO, DOC and nitrogen concentrations in the septic effluent are 0 mg/L, 73 mg/L and 35 mg/L, respectively.

The final concentrations of these species in the recharge from residential land-uses are calculated by weight averaging of the concentrations in recharges from precipitation and septic effluent:

$$\text{Modified Conc} = \frac{\text{Base Conc} \times \text{Base Recharge Rate} + \text{Sep. Conc} \times \text{Sep. Effluent Recharge Rate}}{\text{Base Recharge rate} + \text{Sep. Effluent Recharge Rate}}$$

Eq. 3-15

The table of the 113 different recharge zones with related recharge rate, DOC, TDN and DO concentrations can be found in the appendix.

Simulation Results and Calibration

Based on the information in the previous sections, the land-use data, transport model parameters such as porosity and dispersion coefficients, and other parameters involved in biochemical processes are defined.

The goal of this study is to evaluate the long term consequences of the present development plans in groundwater quality. Therefore, it is essential to reach steady state in the simulation of both the flow pattern and the transport processes.

The flow model is run for steady state conditions. The resulting steady state flow pattern is the base for the reactive transport simulation. RT3D (reactive transport in three dimensions) is utilized to model the fate and transport of the nitrate. Since RT3D does not support the steady state simulation, the transient transport is simulated for about 1250 days. The results show that this time span is long enough to reach the equilibrium in transport processes.

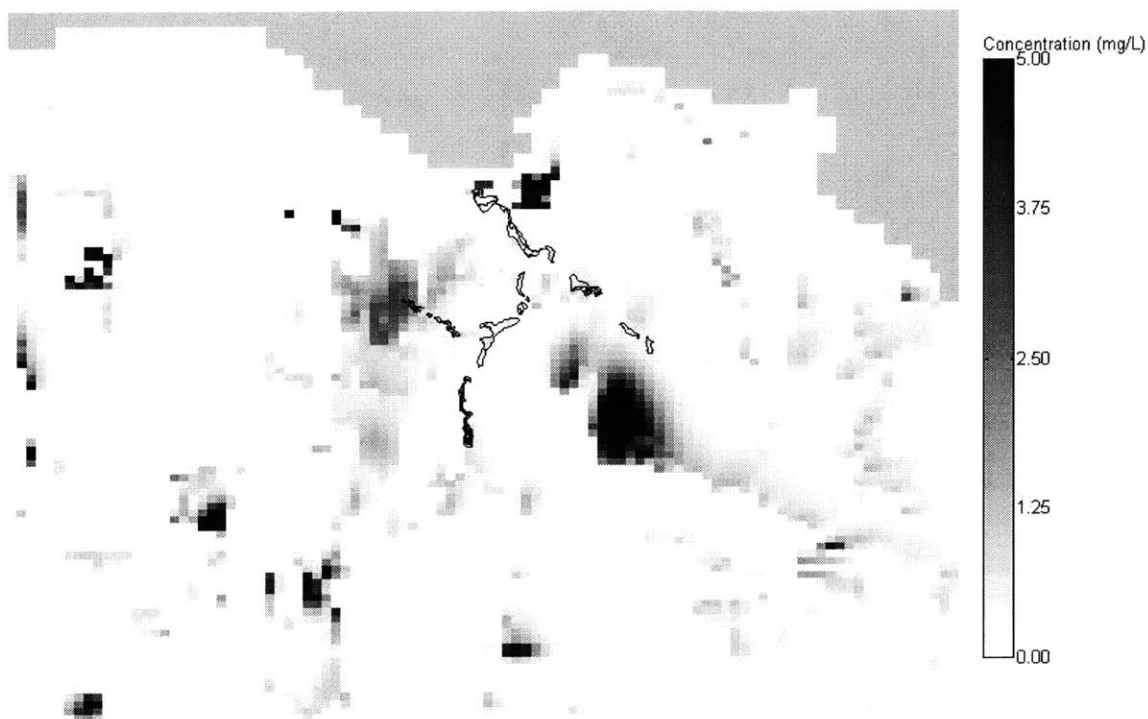


Figure 3-3: Spatial distribution of nitrate concentration, present situation

Figure 3-3 shows the spatial variations of nitrate concentration in the uppermost layer of the model. Since the major part of the Eel River watershed is undeveloped, the nitrate concentrations are small and the watershed is almost free from nitrate. There is a spot of high nitrate concentration south of Eel River that is related to a participation-recreation land-use (land-use code of 7). The high nitrogen load that this land-use inflicts to the aquifer, may explain this spot. The average concentration in the groundwater is less than 1 mg/L.

Figure 3-4 illustrates the spatial distribution of dissolved oxygen concentration in the groundwater. The major part of the Eel River watershed remains aerobic with dissolved oxygen concentration above 3 mg/L.

Under the aerobic condition, the ortho-phosphate presumably is highly absorbed to iron compounds in the aquifer material. Since the aquifer underlying the Eel River watershed remains aerobic, the phosphate is absorbed to the soil and become immobile.

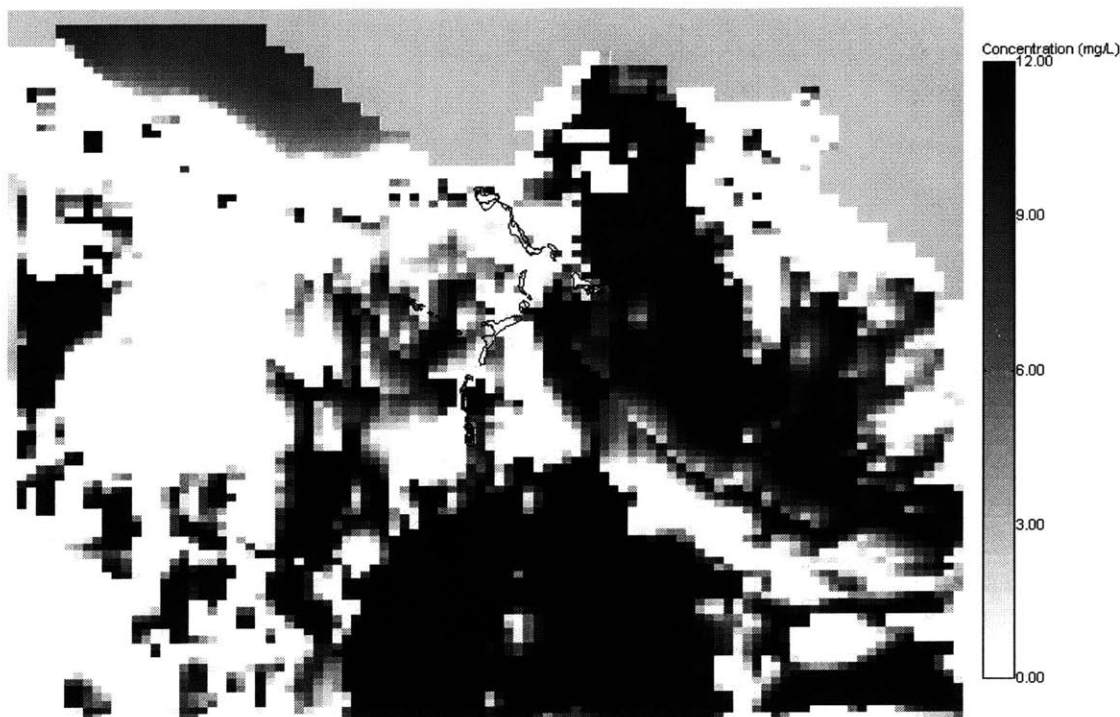


Figure 3-4: Spatial distribution of dissolved oxygen concentration, present situation

To calibrate the transport model, the results of the simulation for the present situation are compared with the field measurements at a number of observation wells. Figure 3-5 shows the approximate locations of observation wells.

Comparison of model results and field observation for total dissolved nitrogen and dissolved oxygen are given in Figure 3-6 and Figure 3-7, respectively. Dissolved oxygen concentration results from model are quite consistent with field measurements. The total dissolved nitrogen concentration results are also close to field observation. Thus, this transport model can be used as the basis for the analysis of groundwater quality under different land-use change scenarios.

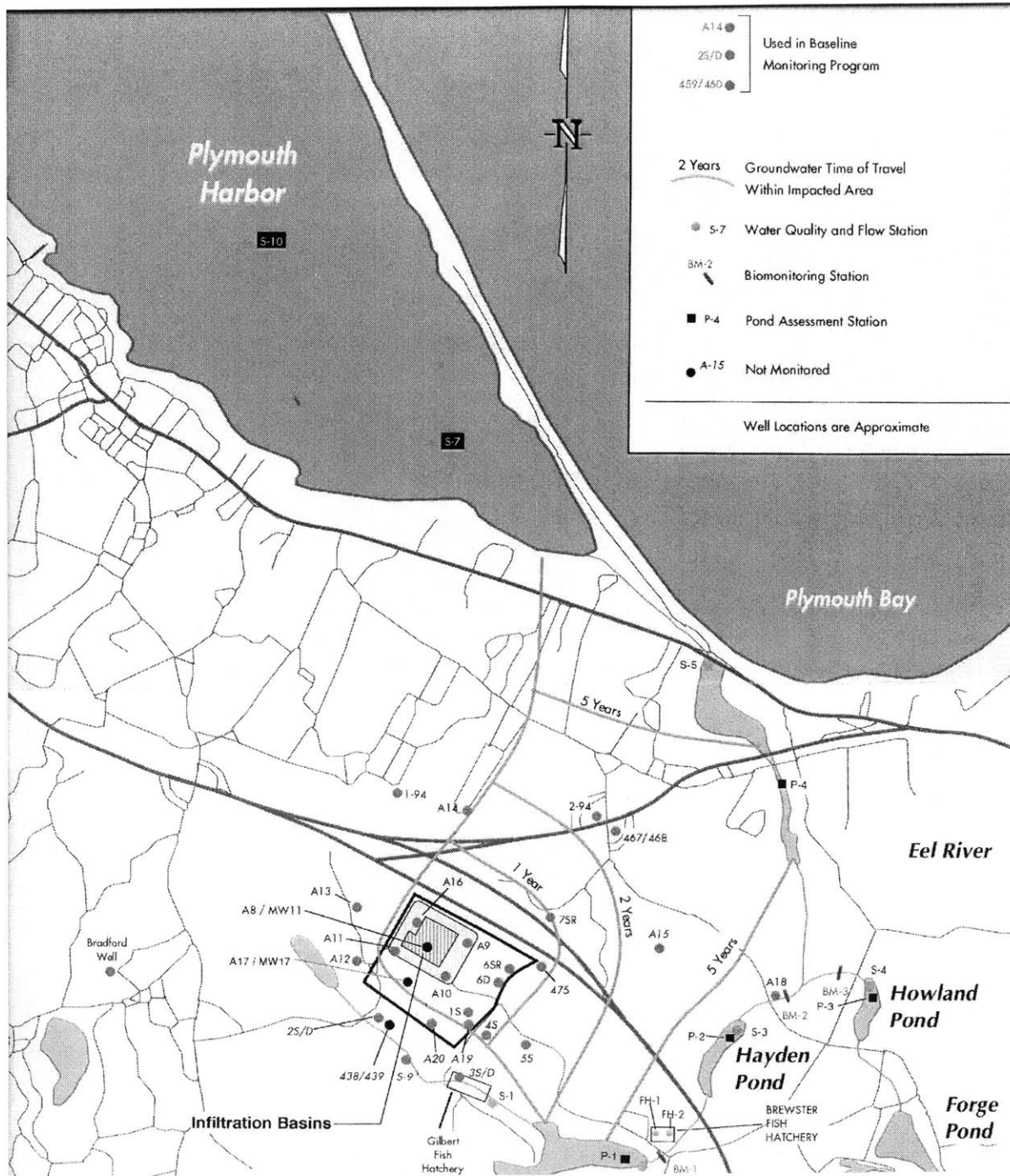


Figure 3-5: Locations of Observation wells and WWTF (CDM, 2000)

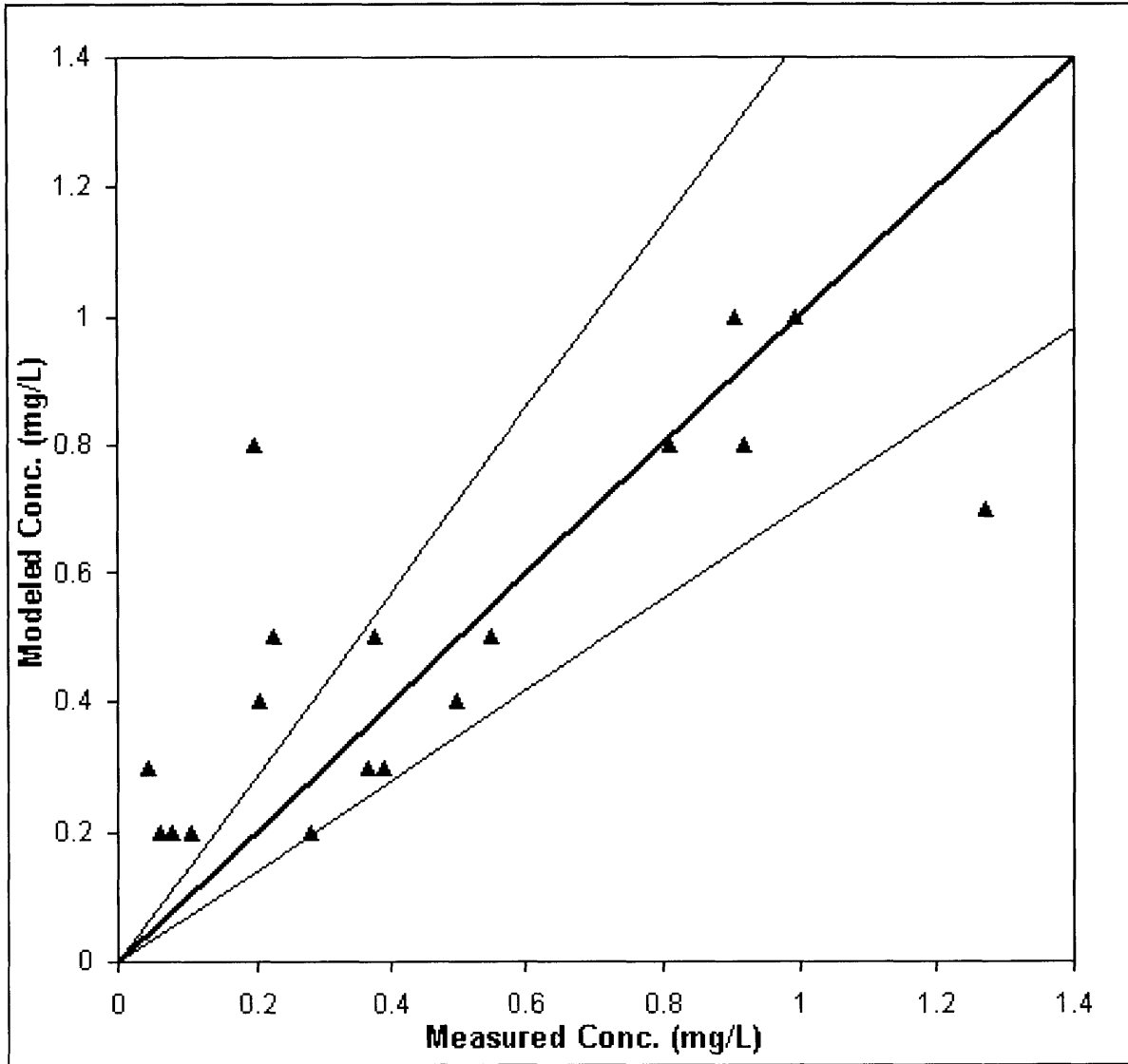


Figure 3-6: Comparison of total dissolved nitrate for calibration

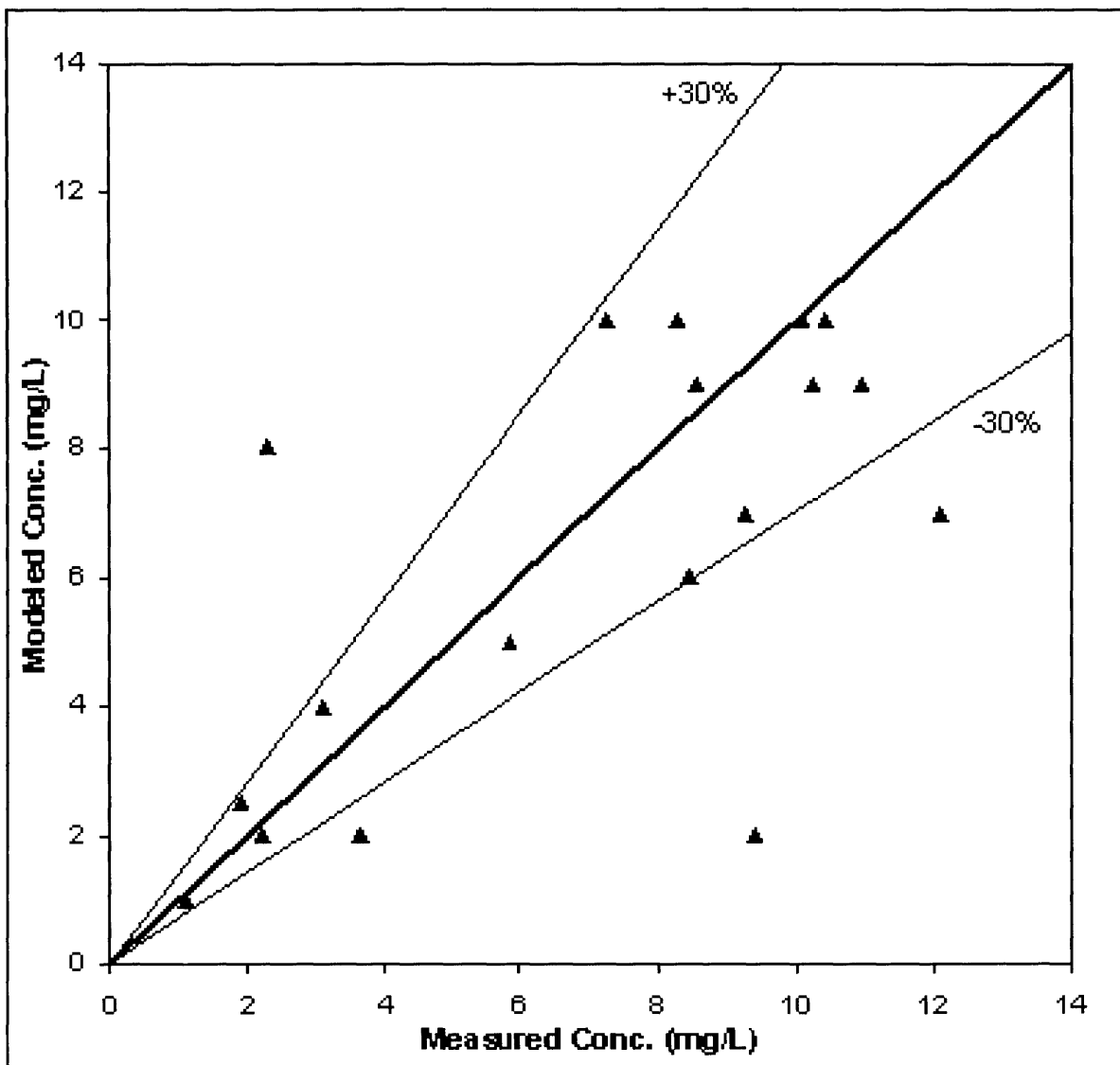


Figure 3-7: Comparison of dissolved oxygen for calibration

4 Future Development Scenarios

4.1.1 Wastewater Treatment Facility

The new wastewater treatment facility is the most important nutrient point source in the Eel River watershed. The WWTF is located in Camelot Industrial Park, between Route 3 and Warren Wells Brook (Figure 3-5). The WWTF has the ultimate capacity of 3.0 mgd of which, 1.75 mgd will be discharged to the existing ocean outfalls and the remaining 1.25 mgd will be discharged to infiltration beds. However, the initial discharge to infiltration beds is 0.75 mgd (CDM, 1997). In this section, the effects of WWTF during its two operation phases on the flow pattern and nitrate concentration are evaluated.

4.1.1.1 Assumptions

The WWTF has two operation phases, with different discharge rates to infiltration beds. In the first phase, WWTF will have a discharge rate of 0.75 mgd (2800 m³/d). This discharge rate will increase in the second phase by 67% to 1.25 mgd (4700 m³/d). However, the concentrations of constituents in the effluent will remain the same. The typical concentrations in the effluent are given in Table 4-1.

There will be 6 “100m-by-100m” infiltration cells in the Phase 1. In the Phase 2, the number of infiltration beds will increase to 10 cells. Thus, the recharge rate to the aquifer (i.e. the effluent discharge rate divided by the area of infiltration beds) will remain constant in the two phases (CDM, 1997).

From the modeling point of view, since the recharge rates and concentrations are constant, a new discharge zone (zone 114) will be representative of the WWTF discharge characteristics in both the two phases. The only change from phase one to the second phase is the area of the infiltration beds (i.e. the number of model cells assigned to this new zone).

Table 4-1 represents the discharge rate and species’ concentrations for the two operation phases of the plant.

4.1.1.2 Results

Released to infiltration beds, the 0.75 mgd effluent of the WWTF in the first phase will raise the water table and change the flow pattern of the groundwater as well as the species concentrations. These changes after starting the second operation phase and increasing the effluent discharge rate by 67% to 1.25 mgd will be more remarkable. The changes in the flow pattern and water table rise due to the effluent of WWTF in the first and second operation phases are shown in Figure 4-1.

Table 4-1: Wastewater treatment facility effluent characteristics (Permit, 2000)

Phase	Area (ha)	Discharge Rate			DOC (mg/L)	DO (mg/L)	TN (mg./L)
		mgd	ft ³ /s	ft/s			
1	6	0.75	1.16	1.8 x 10 ⁻⁶	2	4	10
2	10	1.24	1.93	1.8 x 10 ⁻⁶	2	4	10

Under the infiltration beds, in the Phase 1, the water table will rise about 10 feet to 70 feet above sea level. In the Phase 2, water table will rise another 10 feet and reach the 80 feet above sea level. WWTF will also distort the flow pattern and cause more groundwater discharge to Eel River and Russell-Mill pond.

Another significant change in flow pattern is the change in the seepage velocity. Figure 4-2 represents the spatial changes in the seepage velocity for the two operation phases of the WWTF. The treatment plant changes the naturally uniform groundwater flow-pattern to a high-velocity groundwater flow from beneath the infiltration beds towards the Russell-Mill pond and the mouth of the Eel River. The average seepage velocity can increase to as fast as 2.1 m/d in the Phase 1, and eventually 2.7 in the Phase 2.

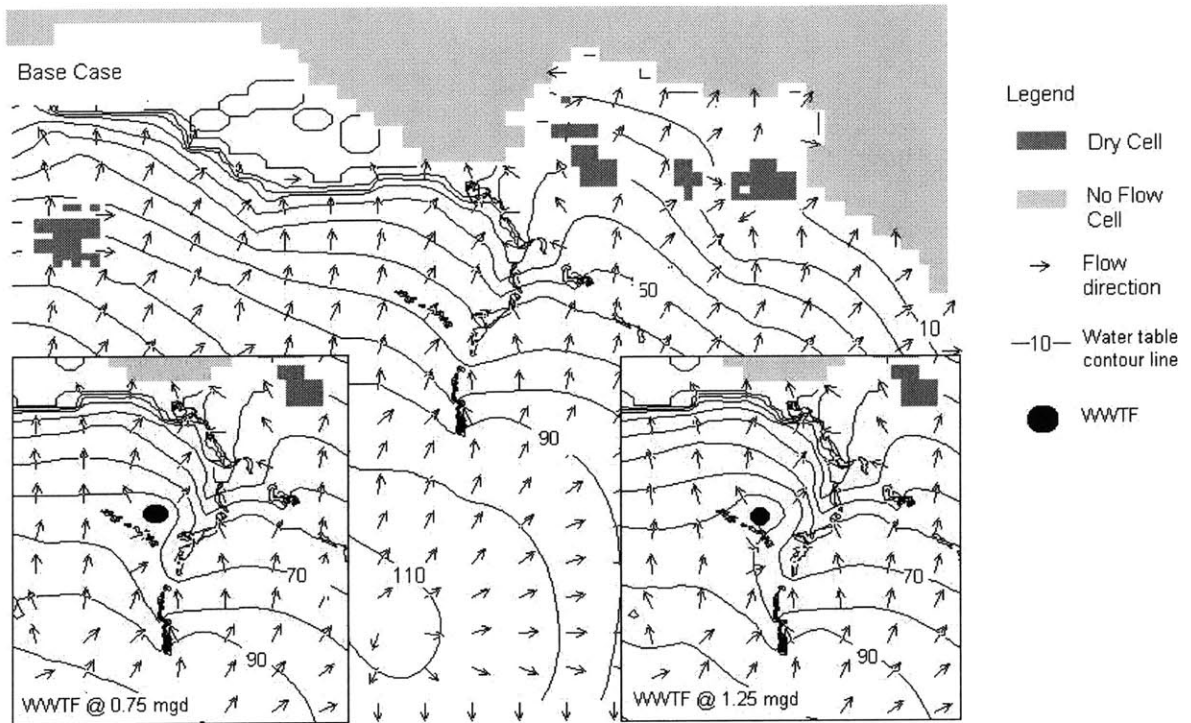


Figure 4-1: Groundwater flow pattern, WWTF

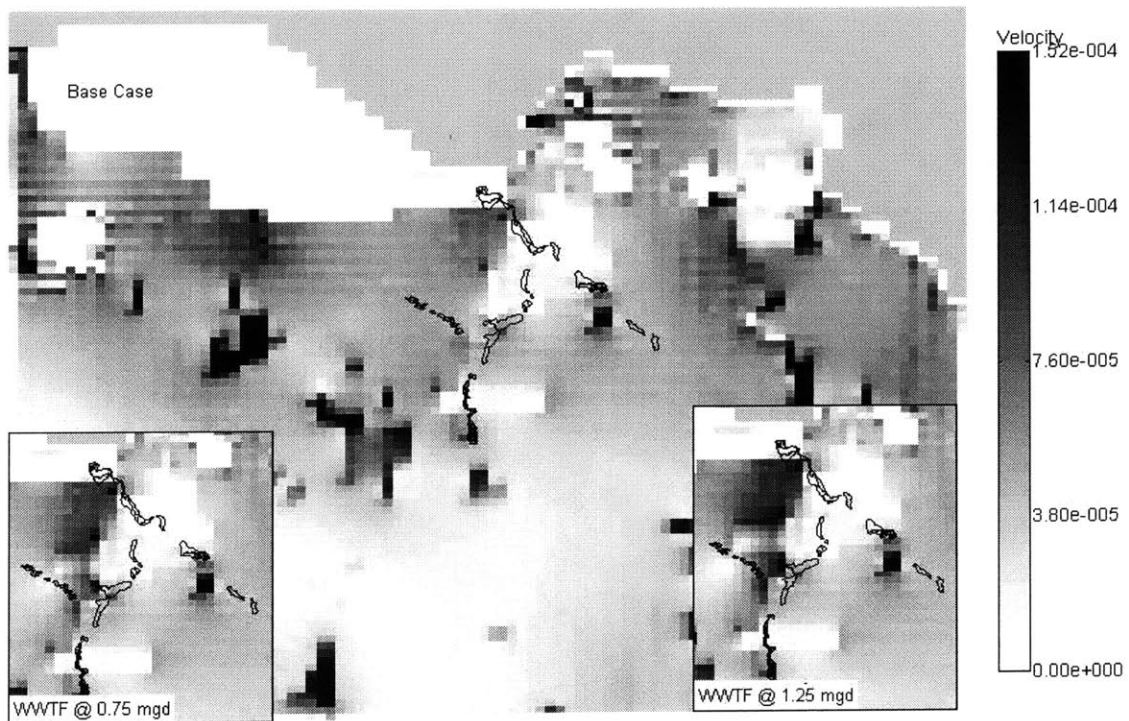


Figure 4-2: Spatial variations in seepage velocity, WWTF

Figure 4-3 demonstrates the spatial distribution of the nitrate concentration after the WWTF starts operation. In the Phase 1, a plume of nitrate-contaminated groundwater will be established underneath the infiltration beds. The concentration of nitrate in the plume reaches as high as 9.1 mg/L. The plume will reach the Russell-Mill pond very quickly. The nitrate concentration in the discharged groundwater to the pond is 8.1 mg-N/L. The plume will also extend later and reach the river upstream of the Eel-River pond and discharge at 2.9 mg-N/L to the river (after about 3 years).

In the Phase 2, the plume will extend more, but the peak concentration will not change. As for the discharged groundwater to the Russell-Mill pond and the river, the nitrate concentration will increase to 8.5 and 4.3 mg-N/L, respectively.

As mentioned before, the flux of the species is the product of concentration and specific discharge, and the latter is proportional to the seepage velocity. Above-mentioned results, suggest that the plume of nitrate-contaminated groundwater, which reaches the Russell-Mill pond and Eel River, will discharge a considerable amount of nitrate to the Eel River system.

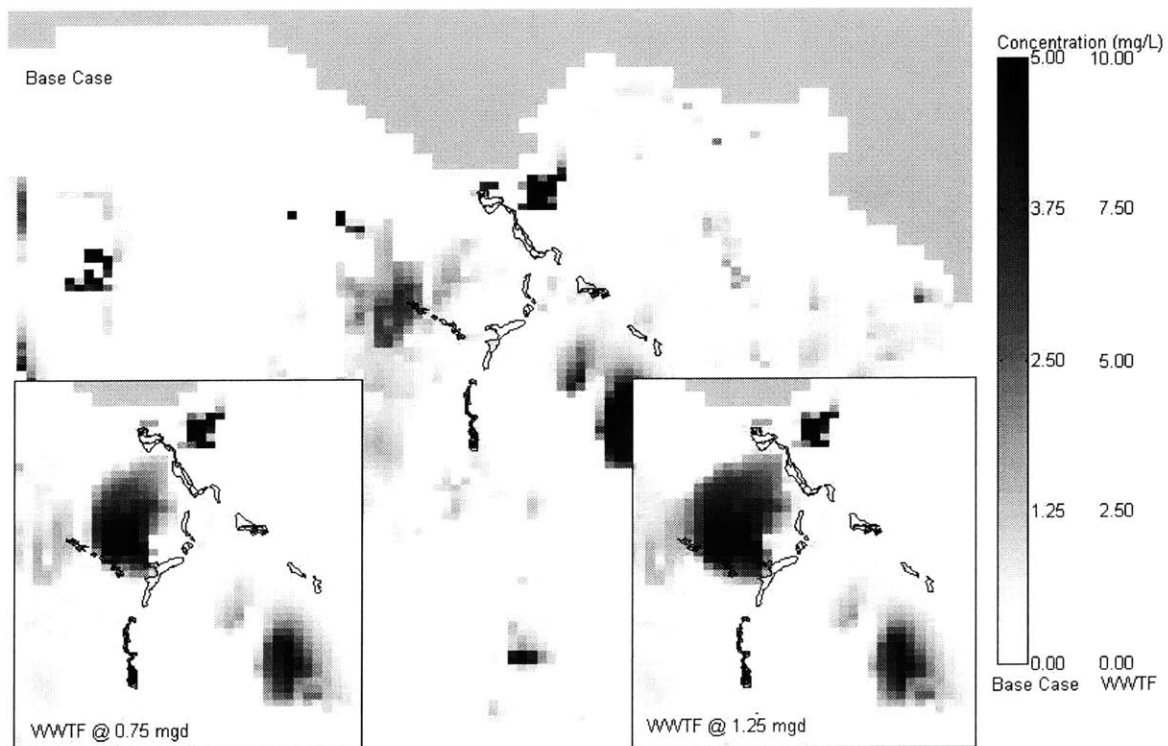


Figure 4-3: Spatial distribution of nitrate concentration, WWTF

The other important result is distribution of dissolved oxygen concentration. As mentioned before, oxygen is an important factor in sorption of ortho-phosphate to the iron content of the soil (TAC, 2000). The spatial variation in dissolved oxygen concentration is represented in Figure 4-4. The results show that an anaerobic plume will be built beneath the infiltration beds. The dissolved oxygen concentration falls to even below 0.2 mg/L, providing the condition for phosphorous release from soil. Although the technical advisory committee's nutrient management plan suggest to eliminate the phosphorous release from WWTF to zero, it is probable to have some phosphorous release through the effluent from WWTF. Considering the anaerobic condition of the aquifer, underneath the infiltration beds, this released phosphorous is not likely to be absorbed to the soil. Consequently, phosphorous release even in very small concentrations along with high effluent discharge rate, will result in a considerable phosphorous load, and threat the health of the Eel River.

It is strongly recommended to perform a precise analysis of phosphorous fate and transport in this aquifer.

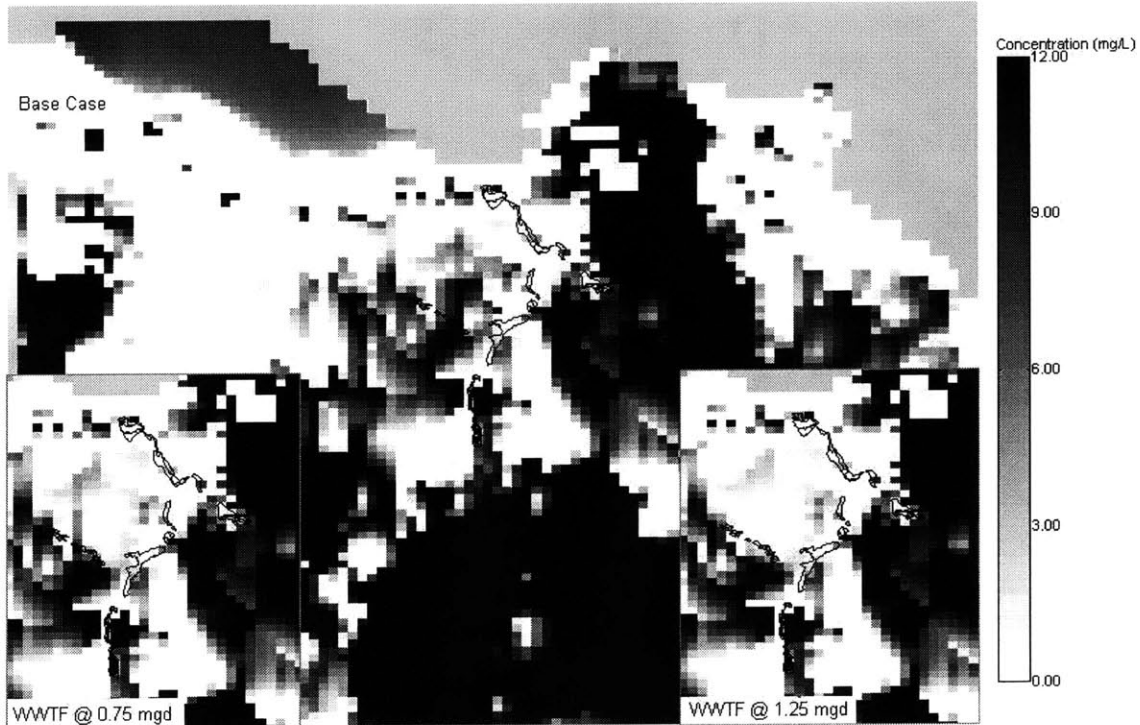


Figure 4-4: Spatial distribution of dissolved oxygen concentration, WWTF

4.1.2 Pinehills developments

Pinehills is a recreational residential development, 998 acres in area, located in the south of the Talcott pond, next to route 3. This development consists of a number of golf courses and houses (Figure 4-5). The septic effluent from the majority of houses will be treated in the private wastewater treatment facility located in the south of the vicinity. However, the nitrogen load from golf courses and grass lawns will be discharged to the groundwater. In this section, the effects of this development are reflected.

4.1.2.1 Assumptions

The Pinehills developments cover an area of 998 acres, discharging a total of 1.78 mgd (650.35 mgal/yr) to the groundwater (Permit, 2000). As for the nitrogen load from this activity, the final detailed plan for the houses and golf courses are not available. Therefore, to have an estimate of the future changes in groundwater quality, two different scenarios are simulated.

In the first scenario, it is assumed that the recharge characteristics from the Pinehills will be the same as of the recharge from similar recreational activities (zone 96, appendix 1) with the nitrate concentration of 18 mg-N/L. In the second scenario, it is assumed that the recharge will meet the permitted limit of 5 mg-N/L (Permit, 2000).

The simulations of these scenarios are built on the model for the WWTF with 1.25mgd discharge. However, the effect of the Pinehills development is independent of the WWTF.

4.1.2.2 Results

The spatial distributions of the nitrate concentration under the 2 scenarios are illustrated in Figure 4-6 and Figure 4-7. If the Pinehills discharge the typical nitrate load of the recreational land-use, a plume of nitrate contamination will be built up and this plum will extend to the east and northeast, effecting the groundwater discharging to the Talcott pond. The maximum concentration in the plume will reach 11.75 mg-N/L.

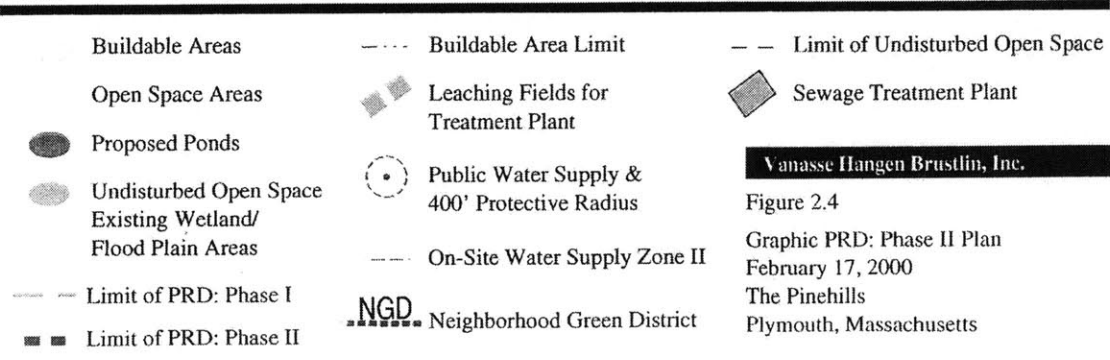
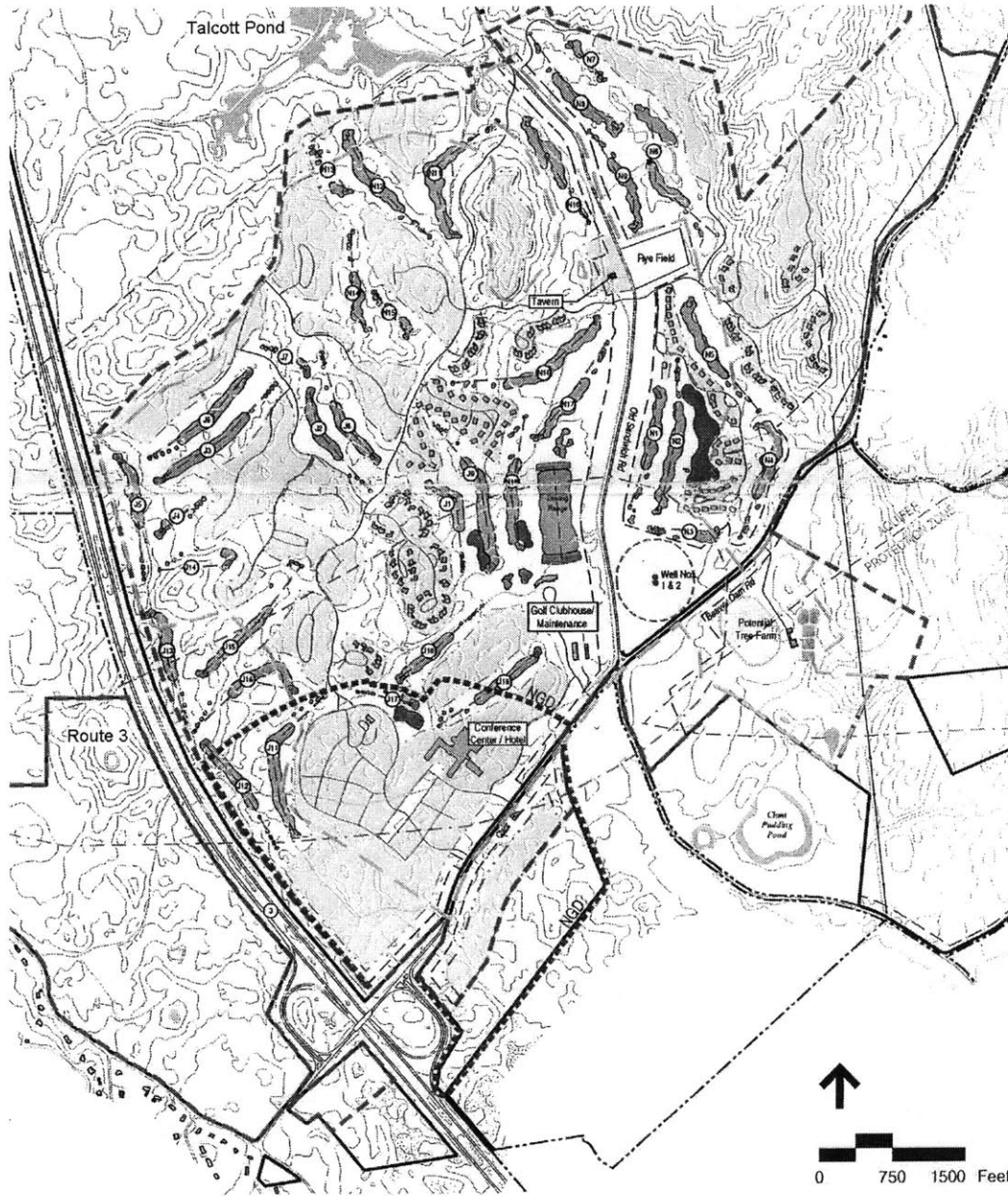


Figure 4-5: Pinehills recreational residential developments, Vanasse Hangen Brustlin, 2000

However, if the permitted nitrate concentration in the discharge, 5 mg-N/L, is met, the plume will be eliminated and nitrate concentration in the groundwater will be reduced to less than 1 mg-N/L.

The scenarios for the Pinehills developments are not very inclusive and more data are required for more specific simulation. Though, these simulations show that before any development and change in land-use, a specific impact assessment on the groundwater quality is a need.

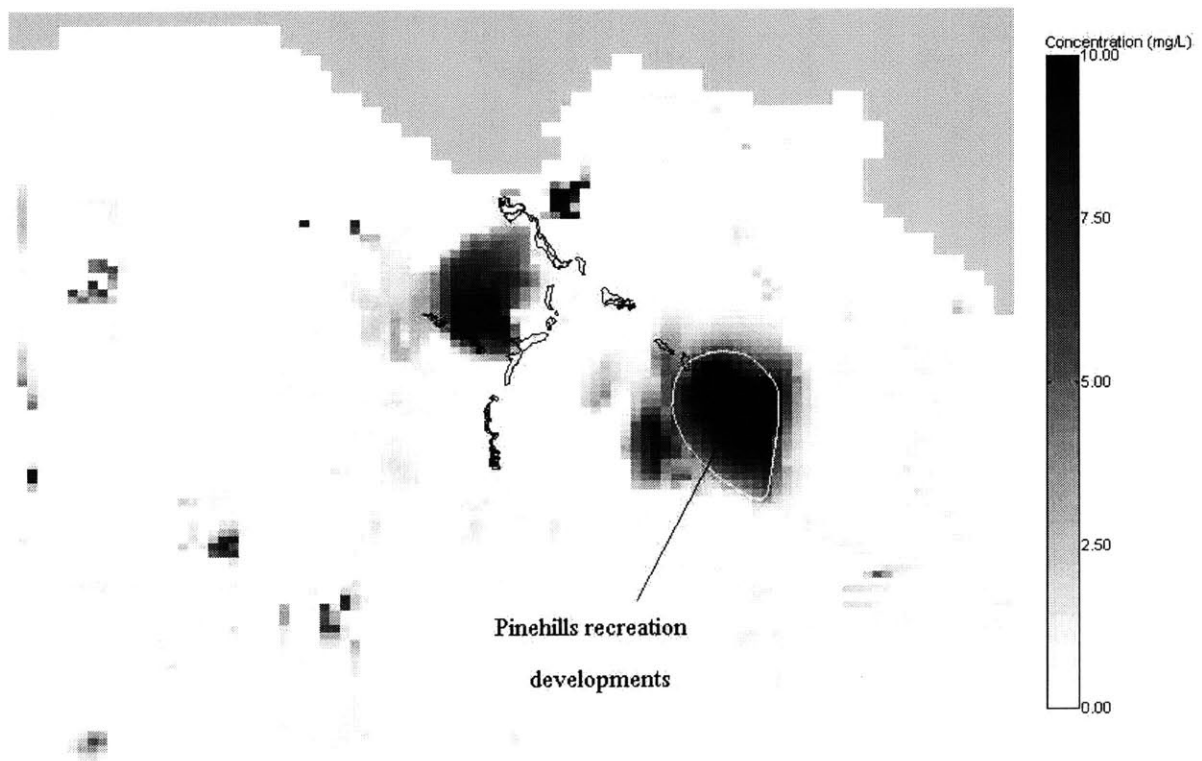


Figure 4-6: Spatial distribution of nitrate concentration, Pinehills developments, typical load

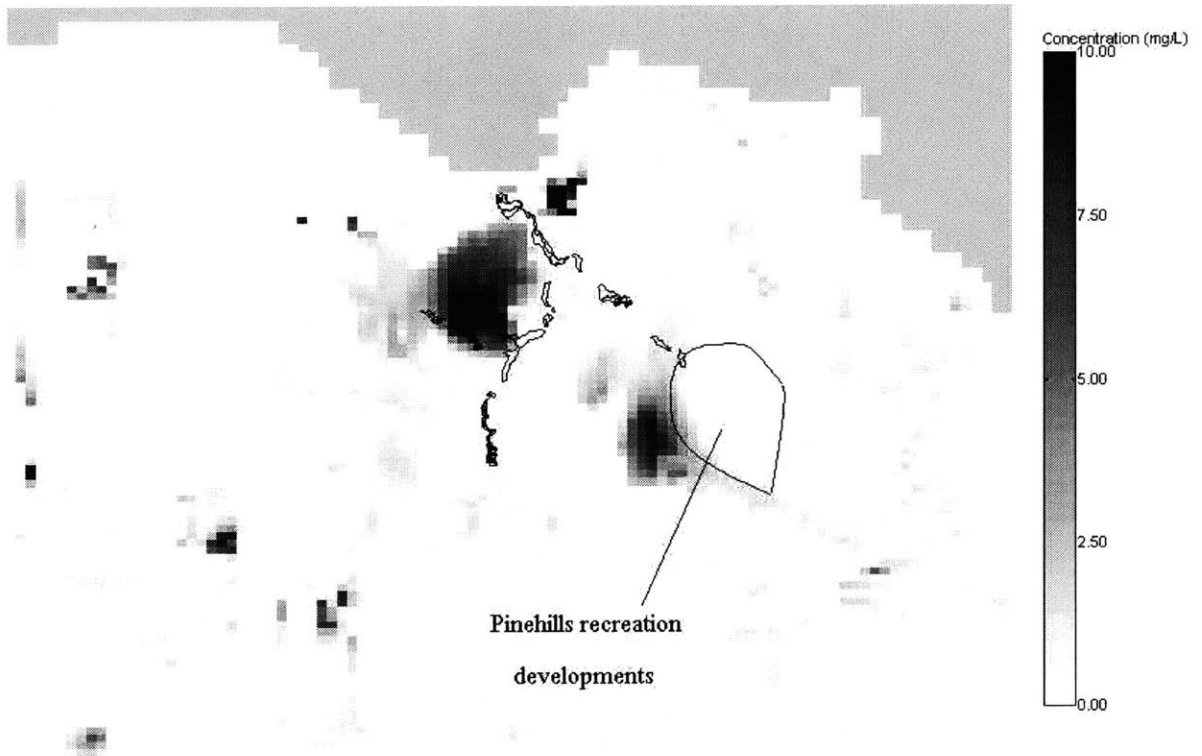


Figure 4-7: Spatial distribution of nitrate concentration, Pinehills, permitted load

5 Conclusion

This study is a small-scale evaluation of probable impacts resulting from nitrogen release to groundwater. Although it was believed that *"the potential nutrient related effects would be most likely restricted to the region within the breakwater adjacent to Eel River discharge (TAC, 2000),"* the results of this study show that the new WWTF and Pinehills recreational developments, without appropriate considerations, will impose enormous amount of nitrogen to the Eel River system and increase the risk of eutrophication in the river.

The WWTF will raise the water table, change the flow pattern and cause more groundwater discharge to the Eel River. An anaerobic plume of nitrate-contaminated groundwater will also be established under the infiltration beds. The plum will extend towards the ocean and reach the Russell-Mill and Eel-River ponds. This increased groundwater discharge to the ponds, along with its high nitrate concentration, drastically increases the nitrogen load to the Eel River.

The proposed nutrient management plan for the watershed suggests that zero-phosphorous-release policy shifts the watershed to phosphorous limiting situation and eliminates the eutrophication danger of the Eel River. However, there is no guarantee that the WWTF's discharge will be free of phosphorous and achieving absolutely phosphorous-free discharge is not feasible. The anaerobic conditions, which will be established beneath the infiltration beds prohibits the sorption of the released phosphorous compounds to soil and this released phosphorous can reach the Eel River.

As for the Pinehills developments, exceeding the permitted limit for nutrient release will result in a plume of nitrate contamination. This plume will reach the Talcott pond and elevate the nitrate level in the Eel River.

Since the Eel River is currently mesotrophic in western branch and oligotropic in the eastern branch, these elevated levels of nitrogen load to Eel River has potential to cause the severe eutrophication of the Eel River. Releasing tremendous amount of nitrogen load to river

provides the ponds with the condition required for eutrophication. Thus, phosphorous release by any means is likely to result in algae bloom and eventually cause eutrophication.

This study was limited to nitrate fate and transport in the subsurface environment. To properly evaluate the impacts of nutrient release from future developments in the watershed, a thorough analysis of phosphorous fate and transport in the groundwater is also required. That study can show if there is any place to be concerned about phosphorous availability to cause eutrophication.

Additionally, a risk assessment study is required to estimate the scope of potential impacts resulting from nutrients released from the new sources. That evaluation of the scope of the probable impact can give better understanding of the problem, providing the decision makers with the necessary information to prevent possible unfavorable consequences.

6 Appendix

Table of recharge zones in the simulation of present situation

Zone	Landuse	Recharge (ft/s)	Concentration (lb/ft ³)		
			Carbon	Oxygen	Nitrogen
1	0	-4.54E-08	0.00E+00	0	0.00E+00
2	0	-2.27E-08	0.00E+00	0	0.00E+00
3	0	-1.37E-08	0.00E+00	0	0.00E+00
4	0	0.00E+00	0.00E+00	0	0.00E+00
5	11	0.00E+00	0.00E+00	0	0.00E+00
6	20	0.00E+00	0.00E+00	0	0.00E+00
7	4	0.00E+00	0.00E+00	0	0.00E+00
8	2	0.00E+00	0.00E+00	0	0.00E+00
9	21	0.00E+00	0.00E+00	0	0.00E+00
10	7	0.00E+00	0.00E+00	0	0.00E+00
11	3	0.00E+00	0.00E+00	0	0.00E+00
12	12	0.00E+00	0.00E+00	0	0.00E+00
13	14	0.00E+00	0.00E+00	0	0.00E+00
14	9	0.00E+00	0.00E+00	0	0.00E+00
15	15	0.00E+00	0.00E+00	0	0.00E+00
16	1	0.00E+00	0.00E+00	0	0.00E+00
17	17	0.00E+00	0.00E+00	0	0.00E+00
18	13	0.00E+00	0.00E+00	0	0.00E+00
19	6	0.00E+00	0.00E+00	0	0.00E+00
20	16	0.00E+00	0.00E+00	0	0.00E+00
21	18	0.00E+00	0.00E+00	0	0.00E+00
22	14	9.24E-09	2.12E-03	1.37E-03	6.91E-04
23	17	9.24E-09	3.54E-03	1.37E-03	1.15E-03
24	20	9.24E-09	7.08E-03	1.37E-03	2.56E-03
25	21	9.24E-09	1.42E-02	1.37E-03	5.29E-03
26	13	9.24E-09	4.56E-03	1.18E-08	2.18E-03

Zone	Landuse	Recharge (ft/s)	Concentration (lb/ft ³)		
			Carbon	Oxygen	Nitrogen
27	3	9.24E-09	4.72E-04	1.37E-03	1.31E-04
28	0	9.24E-09	0.00E+00	1.37E-03	0.00E+00
29	13	1.29E-08	4.56E-03	1.64E-08	2.18E-03
30	3	1.29E-08	3.37E-04	1.37E-03	9.37E-05
31	1	1.29E-08	4.21E-03	1.37E-03	1.40E-03
32	17	1.29E-08	2.53E-03	1.37E-03	8.22E-04
33	20	1.29E-08	5.05E-03	1.37E-03	1.83E-03
34	0	1.29E-08	0.00E+00	1.37E-03	0.00E+00
35	6	1.29E-08	0.00E+00	1.37E-03	9.37E-05
36	7	1.29E-08	2.02E-03	1.37E-03	6.25E-03
37	16	1.29E-08	7.58E-03	1.37E-03	2.48E-03
38	11	1.29E-08	4.56E-03	8.20E-09	2.18E-03
39	18	1.29E-08	6.74E-03	1.37E-03	2.47E-03
40	5	1.29E-08	0.00E+00	1.37E-03	2.43E-03
41	12	1.29E-08	4.56E-03	1.09E-08	2.18E-03
42	19	1.29E-08	1.68E-02	1.37E-03	5.59E-03
43	21	1.29E-08	1.01E-02	1.37E-03	3.78E-03
44	2	1.29E-08	2.53E-03	1.37E-03	8.22E-04
45	4	1.29E-08	1.52E-03	1.37E-03	4.93E-04
46	17	1.80E-08	1.82E-03	1.37E-03	5.92E-04
47	11	1.80E-08	4.56E-03	1.14E-08	2.18E-03
48	10	1.80E-08	4.56E-03	5.72E-09	2.18E-03
49	12	1.80E-08	4.56E-03	1.53E-08	2.18E-03
50	2	1.80E-08	1.82E-03	1.37E-03	5.92E-04
51	3	1.80E-08	2.43E-04	1.37E-03	6.75E-05
52	16	1.80E-08	5.46E-03	1.37E-03	1.79E-03
53	7	1.80E-08	1.46E-03	1.37E-03	4.50E-03
54	14	1.80E-08	1.09E-03	1.37E-03	3.55E-04
55	18	1.80E-08	4.86E-03	1.37E-03	1.78E-03
56	15	1.80E-08	5.46E-03	1.37E-03	1.79E-03
57	21	1.80E-08	7.28E-03	1.37E-03	2.72E-03
58	13	1.80E-08	4.56E-03	2.29E-08	2.18E-03

Zone	Landuse	Recharge (ft/s)	Concentration (lb/ft ³)		
			Carbon	Oxygen	Nitrogen
59	1	1.80E-08	3.03E-03	1.37E-03	1.01E-03
60	4	1.80E-08	1.09E-03	1.37E-03	3.55E-04
61	20	1.80E-08	3.64E-03	1.37E-03	1.32E-03
62	0	1.80E-08	0.00E+00	1.37E-03	0.00E+00
63	6	1.80E-08	0.00E+00	1.37E-03	6.75E-05
64	13	3.56E-08	4.56E-03	4.53E-08	2.18E-03
65	17	3.56E-08	9.17E-04	1.37E-03	2.98E-04
66	4	3.56E-08	5.50E-04	1.37E-03	1.79E-04
67	14	3.56E-08	5.50E-04	1.37E-03	1.79E-04
68	6	3.56E-08	0.00E+00	1.37E-03	3.40E-05
69	11	3.56E-08	4.56E-03	2.26E-08	2.18E-03
70	9	3.56E-08	7.34E-04	1.37E-03	6.62E-04
71	12	3.56E-08	4.56E-03	3.02E-08	2.18E-03
72	3	3.56E-08	1.22E-04	1.37E-03	3.40E-05
73	0	3.56E-08	0.00E+00	1.37E-03	0.00E+00
74	18	3.56E-08	2.45E-03	1.37E-03	8.95E-04
75	21	3.56E-08	3.67E-03	1.37E-03	1.37E-03
76	20	3.56E-08	1.83E-03	1.37E-03	6.62E-04
77	7	3.56E-08	7.34E-04	1.37E-03	2.27E-03
78	2	3.56E-08	9.17E-04	1.37E-03	2.98E-04
79	16	3.56E-08	2.75E-03	1.37E-03	9.01E-04
80	15	3.56E-08	2.75E-03	1.37E-03	9.01E-04
81	0	4.46E-08	0.00E+00	1.37E-03	0.00E+00
82	12	4.46E-08	4.56E-03	3.78E-08	2.18E-03
83	16	4.46E-08	2.20E-03	1.37E-03	7.20E-04
84	18	4.46E-08	1.95E-03	1.37E-03	7.15E-04
85	13	4.46E-08	4.56E-03	5.67E-08	2.18E-03
86	6	4.46E-08	0.00E+00	1.37E-03	2.72E-05
87	21	4.46E-08	2.93E-03	1.37E-03	1.10E-03
88	7	4.46E-08	5.86E-04	1.37E-03	1.81E-03
89	17	4.46E-08	7.33E-04	1.37E-03	2.38E-04
90	4	4.46E-08	4.40E-04	1.37E-03	1.43E-04

Zone	Landuse	Recharge (ft/s)	Concentration (lb/ft ³)		
			Carbon	Oxygen	Nitrogen
91	20	4.46E-08	1.47E-03	1.37E-03	5.29E-04
92	3	4.46E-08	9.77E-05	1.37E-03	2.72E-05
93	1	7.13E-08	7.64E-04	1.37E-03	2.54E-04
94	10	7.13E-08	4.56E-03	2.27E-08	2.18E-03
95	19	7.13E-08	3.06E-03	1.37E-03	1.01E-03
96	7	7.13E-08	3.67E-04	1.37E-03	1.13E-03
97	20	7.13E-08	9.17E-04	1.37E-03	3.31E-04
98	2	7.13E-08	4.59E-04	1.37E-03	1.49E-04
99	21	7.13E-08	1.83E-03	1.37E-03	6.86E-04
100	16	7.13E-08	1.38E-03	1.37E-03	4.51E-04
101	14	7.13E-08	2.75E-04	1.37E-03	8.95E-05
102	9	7.13E-08	3.67E-04	1.37E-03	3.31E-04
103	12	7.13E-08	4.56E-03	6.04E-08	2.18E-03
104	4	7.13E-08	2.75E-04	1.37E-03	8.95E-05
105	6	7.13E-08	0.00E+00	1.37E-03	1.70E-05
106	13	7.13E-08	4.56E-03	9.07E-08	2.18E-03
107	5	7.13E-08	0.00E+00	1.37E-03	4.42E-04
108	11	7.13E-08	4.56E-03	4.53E-08	2.18E-03
109	18	7.13E-08	1.22E-03	1.37E-03	4.48E-04
110	3	7.13E-08	6.11E-05	1.37E-03	1.70E-05
111	0	7.13E-08	0.00E+00	1.37E-03	0.00E+00
112	17	7.13E-08	4.59E-04	1.37E-03	1.49E-04
113	15	7.13E-08	1.38E-03	1.37E-03	4.51E-04

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