MINISTRY OF ENVIRONMENT

Action Research Project

Piloting and Strengthening Adaptation Capacity to Climate Change in the Zarqa River Basin

Assessment of the impact of sanitation management and farming practices on groundwater resources at Al-Kfair village

Deliverable 5

UNDP

PART (I): Surface Water Model

MODELING HYDROLOGY AND NUTRIENT FATE AND TRANSPORT IN AL-KFAIR WATERSHED

CHAPTER 1

INTRODUCTION

Nonpoint source water pollution causes widespread water quality problems worldwide. The U.S. Environmental Protection Agency (USEPA) has concluded that the risks of nonpoint source pollution are much greater than the risks posed by point sources (USEPA, 2000). Agriculture is the largest contributor of nonpoint source pollutants in US, affecting 50-70% of the waters assessed by the USEPA (USEPA, 2000). Agricultural practices such as tilling make soil available to flow overland with storm waters, causing soil erosion. Soil erosion not only influences soil fertility that reduce crop productivity, but also adds sediments and degrades the quality of receiving waters. Sediments also act as a carrier and a storage agent for other kinds of pollutants such as phosphorous, nitrogen and organic compounds. Watershed models can be used in the assessment of these pollutant loads and to identify specific areas that have high potential for pollutants.

1.1 STATEMENT OF THE PROBLEM

Sediments and nutrients carried by the agricultural runoff are identified as the major source of Non-Point Source (NPS) pollution particularly in agricultural watersheds, such as Al-Kfair . NPS pollution can originate from several locations in an agricultural watershed as opposed to point sources that are easily identifiable due to its localized nature. Point source discharges affect the water quality conditions during times of low flow when there is less water to dilute incoming effluents. NPS contribute pollutant loads that are washed off and transported mainly during precipitation events when higher surface runoff exist. NPS pollutants that accumulate in water bodies create water quality problems and make several beneficial use impairments.

1.2 OBJECTIVES OF THE STUDY

The overall objective of this study is to model the hydrological processes and assess the impact of land management practices on water quality and quantity of the Al-Kfair

watershed using Annualized AGricultural Non-Point Source (AnnAGNPS) Pollutant Loading (PL) model.

The specific objectives of the study are to:

- ✓ Investigate the adaptability of AnnAGNPS model for the watersheds in Al-Kfair watershed.
- ✓ Identify the areas susceptible to soil erosion within the watershed and estimate sediment loading in an effort to prioritize the subwatersheds for treatment/management.
- ✓ Assess agricultural nutrient loadings that are responsible for water quality degradation.

1.3 STRUCTURE OF THE REPORT

This report is mainly divided into six chapters to present the key tasks that are required to perform this simulation study.

Chapter 1 includes an introduction to the existing problem in the Al-Kfair watershed and describes the specific objectives of the study.

Chapter 2 presents a literature review on the topics that are relevant to the study. This includes an introduction to non-point source pollution, watershed models that are mostly used for non-point source pollution modeling, AnnAGNPS model applications and sensitivity of the model. It also includes an introduction to the agricultural Best Management Practices that are used to minimize the impacts of NPS pollutants on surface water quality.

Chapter 3 presents a review of the AnnAGNPS model. Some of the concepts and major processes within the AnnAGNPS model are reviewed in this chapter.

Chapter 4 describes, the development of the computer-based model to represent the Al-Kfair watershed. This includes an overview of the watershed, modules used and methodology adopted in developing the necessary input database for the model. This chapter further describes the parameter selection process conducted in the study and the methodology used in the modeling of agricultural best management practices.

Chapter 5 presents the results and review of various simulations performed for sensitivity analysis, assessing model loadings and their contributing areas.

Chapter 6 presents the conclusions derived from the results of the simulation study and recommendations for further actions with a view to monitor non-point source pollution within the Al-Kfair watershed.

CHAPTER 2

LITERATURE REVIEW

2.1 NON-POINT SOURCE POLLUTION

Non-point source (NPS) pollution is the introduction of pollutants into a system through an indirect or unidentified route. It is generated in a variety of land uses including agriculture, construction and forestry practices. NPS pollution is often associated with precipitation events, snowmelts or irrigated land activities that generate surface runoff, pick up pollutants and deposit them into water bodies. The term non-point source is used to distinguish it from point source pollution, which comes from localized, easily identifiable sources such as sewage treatment plants or industrial facilities.

NPS pollutants have particular characteristics that separate them from point source pollutants. They are (Ambrosio et al., 2001 and Leeds et al., 1994):

- NPS pollutants enter receiving waters in a dissipated manner at intermittent intervals mostly due to variations in meteorological conditions
- NPS pollution occurs over a large area of land and moves overland before reaching waterways
- NPS pollution sources are difficult to monitor at the point of origin
- The effect of NPS pollution may be minor when considered individually but will be significant when considered collectively

The extent and significance of NPS pollution is related to parameters that are uncontrollable by humans such as climatic conditions (storm intensity and frequency) and geographical conditions (soil type, erodibility).

NPS pollution has been associated with water quality standard violations and the contamination of aquatic ecosystems that lead to unsafe drinking water, destroyed habitat,

fish kills, property loss and many other environmental and human health problems (Ambrosio et al., 2001). According to the 3rd World Water Forum - 2003, every year at least five million people die from water related diseases worldwide (Pollution Probe, 2006).

A report (MOE, 2004), indicated that "Water treatment alone cannot ensure that we can meet our needs for good quality water. Even with the best water treatment technology money can buy, a community is at risk if it relies on a water source that is susceptible to contamination." Once water sources get contaminated, it is often very difficult to remove them, because some of the contaminants may stay for decades or even centuries (MOE, 2004).

National Water Quality Inventory Report to US Congress - 2000, indicated that 45% of the surveyed waters in the United States were contaminated with pollutants such as sediment, nutrients, bacteria and metals. Primary sources of impairment were identified as runoff from agricultural lands, municipal point sources and hydrologic modifications such as channelization and dredging (USEPA, 2002). A study on economic impact of erosion on surface waters (Osterkamp et al., 1998) indicated that annual costs of damages due to physical, chemical and biological sediment discharge in North America exceed $16 billion.

2.2 SOIL EROSION PROCESS

Soil erosion and sediment delivery are the key processes controlling NPS pollution in agricultural watersheds. Furthermore, erosion reduces production potential by removing nutrients needed for crop production, deteriorates the soil structure by deposition and increases flood hazards by reducing the infiltration rate and water holding capacity of the soil. Sedimentation reduces capacity of downstream channels and waterbodies, destroys fish and wild life habitat and increases cost of maintaining downstream waterbodies such as harbours.

During precipitation events, the energy from the impact of raindrops and the shear force of water flowing over the land surface causes detachment of soil particles. The detached soil particles with nutrients, such as nitrogen and phosphorous attached to them, are transported to waterways by the surface runoff while a portion of the detached soil particles deposit in the field before reaching streams. Sediment transport is a function of rainfall intensity, sediment characteristics and hydraulic parameters. Renard et al., (1997) noted that sediment transport is largely a function of topography and runoff velocity while deposition is a function of runoff velocity and sediment particle sizes.

Erosion occurred by water is generally recognized in three different forms: sheet and rill erosion, stream bank erosion and stream bed erosion. Sheet erosion is a process in which soil detached by rainfall energy, is moved across the soil surface by sheet flow, often in the early stages of runoff. Rill erosion occurs as runoff water begins to concentrate in small channels or streamlets. Inter-rill erosion takes place between rills. Sheet and interrill erosion can go unnoticed because it removes sediment in a uniformlayer. For a susceptible soil, rill erosion is immediately visible since flow concentrates in many small streamlets or rills.

Sheet and rill erosion carry mostly fine-textured, small particles and aggregates. These sediments will contain higher proportions of nutrients, pesticides, or other adsorbed pollutants than are contained in the surface soil as a whole. Sheet and rill erosion are generally active only during or immediately after rainstorms or snowmelt.

2.3 WATERSHED MODELS

As non-point source pollution (NPS) problem has gained more attention throughout the world in recent years, various methods have been developed to evaluate the magnitude and extent of NPS pollution. Most of these methods involved development of computer-based models to analyze stormwater quality and quantity in watersheds. Generally, models

are simplified mathematical representations of real systems or processes that can be used for simulations or predictions. Watershed models simulate the generation and movement of stormwater runoff as well as the pollutants it carries from the source areas to downstream waterbodies. NPS pollution models can be used in estimating loadings of chemicals, sediments and nutrients that degrade water quality, establishing critical source areas and ranking alternative measures. Hence they form effective tools for watershed planning and management (Novotny, 2003). Use of these models have been limited due to difficulties involved in simulating large areas having heterogeneous properties such as land use, land cover, soils, and topography and gathering large amounts of input data. Linking models with GIS technology, which has the capability to handle large volumes of spatial and non-spatial data, has helped in overcoming many of these difficulties.

2.3.1 Classification of models

Models are generally classified based on their functionality, method by which inputs and outputs are manipulated and whether they simulate single event or continuous processes. There are two general types of models, physical or analogue models and mathematical models. Physical models are simpler physical representations of complex systems that are assumed to have similar properties to the prototype system or reduced-dimension representation of real world system. Diskin (1970) stated that the "mathematical models are simplified systems that are used to represent real-life systems and may be substitutes of the real systems for certain purposes." According to Woolhiser and Brakensiek (1982), mathematical model is a "symbolic, usually a mathematical representation of an idealized situation that has the important structural properties of the real system." Mathematical models can be subdivided into analytical and numerical models. Analytical models provide a direct solution of the governing equations for homogeneous systems. Numerical models simulate

more complex systems by solving the governing equations that represent the physical processes approximately (Woolhiser and Brakensiek, 1982).

Models can also be classified as deterministic or stochastic depending on the character of the model outputs. If all the input data, parameters and processes are considered free of random variation and known with certainty, then the model is referred to as a deterministic model. Deterministic hydrologic simulation models are well established and widely used in watershed management activities. Stochastic models have the capability of representing the random variability of input parameters, whereby known probability distributions are used to determine statistical probabilities of output parameters.

According to the degree of spatial variability mat is represented and simulated, models may be classified as lumped parameter or distributed parameter models. Lumped parameter models ignore spatial variability of land uses, soil types, and other land surface properties within a computational unit. Effective parameter values are typically estimated based on area weighted averages. A distributed parameter model is one in which the spatial variations of characteristics are considered explicitly. In this approach, the watershed is divided into relatively smaller elements that may be considered as homogeneous units or cells. Each unit is modeled separately and the output is obtained by routing the flow or loadings from cell to cell. In these models, changes in the watershed and their effects on the output can be modeled effectively. During the past decade, distributed parameter models are linked with GIS in reducing manual data input requirements.

Hydrologic and water quality models account for water, sediment and chemical transport through watersheds. NPS models are concerned with generation, transport and tracking of pollutant loadings such as sediment, nutrients and chemicals into waterways. These models typically simulate either on an event basis or on a continuous basis. An event model simulates over a single storm that may range from a few hours to few days.

Continuous simulation models are useful in analyzing long-term effects of hydrological changes and watershed management practices since they take into consideration of both during and between precipitation events.

2.3.2 NPS pollution models

To model NPS pollutant loading from a watershed, a model of sufficient complexity to simulate the diffuse nature of NPS pollution is required. During past three decades, several watershed-scale, hydrologic and NPS pollution models have been developed. These models cover a large range of complexity depending on the extent to which hydrologic, sediment erosion and chemical processes are modeled in a mechanistic manner or based on empirical procedures (Donigian and Huber, 1991). These models employ wide ranges of techniques, from simple annual loading functions to detailed process simulation models. Most of the commonly used models were developed in 1970s and 1980s. Since 1980s and early 1990s, most of the research in model development was on integration of geographic information systems (GIS) and remote sensing data and development of graphical user interfaces (Borah and Bera, 2003).

It is necessary to have a clear understanding of an appropriate model for an application and for a certain watershed. Parsons et al. (2001) noted that if a model is to be truly practical and applicable for the purpose, a potential user has to understand: the original purpose of the model, under what conditions the model will perform correctly, the accuracy that can be expected under the best conditions and the limitations of the model. The advantages and limitations of three distributed parameter models: AnnAGNPS, ANSWERS-2000 and SWAT and a lumped parameter model HSPF are discussed below.

2.3.3 AnnAGNPS pollutant loading model

AnnAGNPS is a continuous simulation, multi event modification of single event model AGNPS, which was first developed in the early 1980's by the Agricultural Research Service (ARS) in cooperation with Natural Resources Conservation Authority (NRCS) (Bingner and Theurer, 2003). AnnAGNPS was first released in 1998 and is intended to be used as a tool to evaluate non-point source pollution from agricultural watersheds ranging in size up to 300,000 ha. In AnnAGNPS, watershed is subdivided into homogeneous land areas (cells) based on land use, soil type and land management. A separate Window based flow network generator using DEMs can be used to subdivide the watershed into hydrologically derived cells of different shapes. The model can be used to examine current conditions in a watershed or to compare effects of different conservation alternatives within a watershed (Bingner and Theurer, 2003).

AnnAGNPS calculations are performed on a daily time step. AnnAGNPS simulates water, sediment, nutrient and pesticide transport at the cell and watershed level. Each day the applied water and resulting runoff are routed through the watershed system before the next day is considered. Runoff is calculated using SCS Runoff Curve Number (RCN) equation where the curve numbers are modified daily, based upon soil moisture, crop stages and tillage operations. Separate input files for watershed data and simulation period climate data allows for quick changing of climate data. Overland sheet and rill erosion of sediment for each cell is determined using RUSLE (Renard et al., 1997). Peak flow calculations are performed using TR-55 graphical peak discharge method. Sediment routing is calculated based upon transport capacity relationships using the Bagnold stream power equation (Bingner and Theurer, 2003). A daily mass balance for Nitrogen (N), phosphorous (P) and organic carbon (C) is calculated for each cell. Nutrients and pesticides are subdivided into soluble and sediment attached components for routing. Each nutrient component is decayed based upon

the reach travel time, water temperature, and an appropriate decay constant (Theurer and Bingner, 2005).

The limitations of the model are identified as: all runoff and associated sediment, nutrient and pesticide loads for a single day are routed to the watershed outlet before the next day's simulation begins regardless of how many days this may actually take. Also point sources are limited to constant loading rates for entire simulation period.

2.3.4 ANSWERS-2000

ANSWERS (Areal Non-point Source Watershed Environment Response Simulation) - 2000 is a continuous simulation, distributed parameter, physically-based model developed in mid 1990s for evaluating the effectiveness of agricultural and urban BMPs in reducing sediment and nutrient delivery to streams in surface runoff. The original ANSWERS was developed in late 1970s as an event-based, distributed parameter model (Dillahaetal., 2001).

The model divides the area simulated into a uniform, square grid cells. Typical cell sizes range from 0.4 to 1 ha. with smaller cells providing more accurate simulations. Within a cell, all properties such as soil, land use and management are assumed homogeneous. The model can simulate BMPs such as conservation tillage, ponds**,** grassed waterways and tile drainage. The model simulates soil detachment, transport and deposition. N and P are simulated using correlation relationship between chemical concentration, sediment yield and runoff volume (Zhen, 2002).

Limitations of the model are identified as (Parsons et al, 2001): the sediment detachment submodel is empirical and out of date, the model does not simulate snow pack and melt and is thus not suitable for use in areas with significant winter snow accumulation and snowmelt.

2.3.5 SWAT model

Soil and Water Assessment Tool (SWAT) is a continuous, distributed parameter river basin or watershed scale model developed by Agricultural Research Services of US Department of Agriculture (Neitsch et al. 2002). Model can be used to predict the impact of land management practices on water, sediment and agricultural pollutants in large complex watersheds. The model emerged mainly from SWRRB (Simulator for Water Resources in Rural Basins) (Arnold et al., 1993) and inherits features from CREAMS (Chemicals, Runoff and Erosion from Agricultural Management Systems), GLEAMS (Groundwater Loading Effects on Agricultural Management Systems), and EPIC (Erosion - Productivity Impact Calculator) models (Borah and Bera, 2003). SWAT model divides the watershed into a number of subwatersheds or subbasins called Hydrologic Response Units (HRU's). HRU's are lumped land areas having unique land cover, soil and management combinations.

The model uses a Geographic Information System (GIS) interface to facilitate the automatic development of input parameters that are required to **operate** the model (Srinivasan and Arnold, 1994). SWAT model has hydrology, sediment, chemical and microbiological components and uses SCS runoff curve number and other empirical relationships to compute runoff volumes and peak flows. Erosion caused by rainfall and runoff is computed using Modified Universal Soil Loss Equation (MUSLE), a modified version of USLE, developed by Wischmeier and Smith (1978). SWAT does not make a distinction between sediment originating over the landscape (sheet, rill and gully erosion) and sediment originating within the stream system (bed and bank erosion). This may result in attributing all of the sediment load to the USLE parameters which are only related to sheet and rill erosion and considered as a serious deficiency with the model (Bingner et al., 2005).

2.3.6 HSPF model

Hydrological Simulation Program - FORTRAN (HSPF), developed by USEPA is a one-dimensional stream network, lumped parameter, continuous simulation model that can simulate watershed hydrology and water quality for both conventional and toxic organic pollutants. HSPF produces a time history of the runoff flow rate, sediment load, and nutrient and pesticide concentrations, along with a time history of water quantity and quality at any point in a watershed (Donigian and Huber, 1991). HSPF simulates three sediment types (sand, silt, and clay) in addition to a single organic chemical and transformation products of that chemical.

HSPF is originally developed from the Stanford watershed model. HSPF needs extensive input data and requires considerable effort when applied to a watershed and is not user friendly. In HSPF**,** overland flow is treated as a turbulent flow process and simulated using the Chezy-Manning equation. Reach routing is performed by using kinetic wave method. A number of pollutant transport processes can be modeled including chemical partitioning, hydrolysis and volatization. Since HSPF performs simulations on a lumped parameter basis, magnitudes of parameters are to be determined by calibration and hence model requires considerable amount of time for calibration.

2.3.7 Model selection

In 1993, Natural Resource Conservation Service (NRCS) completed a study of thirty-eight available water quality models. Two watershed scale, agricultural non-point source pollution models: AGNPS (precursor to AnnAGNPS) and SWRRB (precursor to SWAT) were selected for further analysis. AGNPS contained US Department of Agriculture, NRCS approved science and it was selected for further development (Bingner et al., 2005).

The processes such as gully erosion and tile drains in agricultural fields are unique processes in AnnAGNPS pollutant loading model (Bingner et al., 2005). Model's capability to display pollutant loadings in GIS environment is also advantageous. For the current study, AnnAGNPS pollutant loading model was selected on the basis of its capability of identifying and evaluating the sources of water, sediment and nutrient within the watershed.

2.4 AGNPS/AnnAGNPS MODEL APPLICATIONS

AGNPS has been used as a simulation model for prediction of non-point source pollution, studying the impacts of land use management on water quality and assessment of BMPs for more than a decade, in United States and in several other countries.

Bhuyan et al. (2003) applied the model to assess nutrient loadings from five watersheds in Kansas, US and concluded that the model is useful as a decision support system for resource managers. Since AGNPS assumes a uniform rainfall over the entire watershed, which is certainly not the case in a large watershed, they recommended that the large watershds be divided into smaller sub-watersheds to increase estimation and prediction accuracy. Ma et al. (2002) applied AGNPS to a watershed in Kalamazoo, Michigan in the US to evaluate the parameters that are most sensitive for phosphorous sediment loading. Their results showed that phosphorus sediment is the most sensitive to soil texture while sediment loss is the most sensitive to the SCS curve number.

Vennix and Northcott (2004) applied the AGNPS model to prioritize vegetative buffer strips placement on reducing sediment loading in an agricultural watershed in Michigan, US. Results of this study may be helpful for watershed managers to implement vegetative buffer strips in site-specific areas within the watershed to employ efficient implementation of conservation management programs. Smith (2002) applied AGNPS model to identify critical source areas of sediment and nutrient runoff in a watershed in South Dakota, US. Also the

model was used to develop management alternatives to reduce sediment and nutrient loads. Sugiharto et al. (1994) applied the AGNPS to evaluate twenty different management practices in reducing sediment and phosphorus yields. Kausman and Mitchell (1997) applied AGNPS to assess erosion and sediment in Indonesian watersheds. They found that AGNPS model simulations gave realistic results for erosion rates and sediment yields. Grunwald and Norton (1999) applied AGNPS model to predict runoff and sediment yield in two small-ungaged watersheds in Germany. Results indicated that there was considerable under and over prediction of surface runoff and sediment yield. The model has been applied in many countries in different circumstances to estimate surface runoff, soil erosion, sediment yield and nutrient loading due to non-point source pollution (Lenzi and Luzio, 1997; Mohammed et al. 2004).

Mitchell et al. (1993) evaluated the suitability of the model for predicting surface runoff and sediment yield from watersheds in Illinois, US. Results indicated that variation of runoff between modeled and observed data was reasonable but sediment yield varied about five times the observed data. Despite these discrepancies, it was concluded that the model is a valuable tool for water quality management. Pekarova et al. (1999) tested the possibility of applying AGNPS model in Slovakia and simulated runoff and nutrient loading in surface water in two experimental watersheds. Comparison of results indicated that the AGNPS was suitable to model runoff and nutrient loading with sufficient accuracy.

AGNPS model has been tested and validated for Ontario watersheds. Leon et al. (2003) applied the model for a southern Ontario watershed to validate the model for nutrients and runoff. The results showed that the model is well suited for applications in Southern Ontario. Perrone and Madramootoo (1997) applied AGNPS to a watershed in Quebec to determine its predictive capability with respect to surface runoff, peak flow, and sediment yield. Booty et al. (2005) used AGNPS to study surface water quality conditions during dry

and wet weather in a watershed in Ontario. Results of the study may be used to establish a methodology for assessing the sensitivity of water quality to the climatic changes.

AnnAGNPS model has had fewer applications. Baginska et al. (2003) applied AnnAGNPS in a watershed in Sydney, Australia to model nutrient transport. Bingner et al. (2005) applied AnnAGNPS to Upper Auglaize watershed in Ohio, US for assessing and reducing pollution from agricultural runoff and other non-point sources. AnnAGNPS has been applied in a watershed in Mississippi, US to estimate sediment yields to develop water quality targets (Simon et al., 2002). Results showed that flows and sediment loads estimated from the AnnAGNPS were in close agreement with the measured data. AnnAGNPS model has been applied in a Southern Ontario watershed (Das et al., 2006) to evaluate the hydrology and sediment loadings from non-point sources. Model results showed that simulated runoff was under predicted and sediment yield was over predicted. Their calibration and validation results showed that the AnnAGNPS model is capable of simulating the runoff and sediment yield fairly well for a cold and temperate region like Ontario.

2.5 MODEL SENSITIVITY

Extensive number of input parameter requirement is a well-known problem that exists in hydrological models, especially with distributed models. They contain hundreds of parameters that represent hydrologic and water quality processes in watersheds. Due to spatial variability and some other constraints, model input parameters always contain uncertainty to some extent. Therefore it is important to understand the input parameters that are sensitive to model output and will be beneficial in model development and application. This knowledge will result in better estimated values and thus reducing model uncertainty.

Sensitivity analysis is a formalized procedure that can be employed to identify parameters that have a significant influence on model simulation results. Those parameters, to which the model output is highly sensitive, require a special care in their estimation in order

to produce reliable conclusions. Newham et al. (2003) stated that sensitivity analysis might be used to increase the confidence in a model and its predictions. Sensitivity analysis is an important part of the model validation, which draws attention to those components where further research and development should be focused to enhance model performance. Though several researchers have accepted the importance of sensitivity analysis, there is no single, well accepted procedure for sensitivity analysis.

Vieux and Needham (1993) investigated the effects of grid cell size on sediment yield predictions. Results indicate that grid cell sizes are the most important factor affecting sediment yield. Sediment yield is mostly dependent on flow-path length and as the grid cell sizes increase, stream meanders get shortened and cause increased sediment yield. Therefore, cell sizes should not be selected arbitrarily and should be based on a scale appropriate for capturing the spatial variability of the watershed. Qiu et al. (1997) reported AGNPS outputs with two different cell resolution (100x100 meter and 200x200 meter). Results showed that cell resolution did not cause significant difference in estimating the soil loss. However, estimated nutrient loading was 20 percent higher in the 200 meter cell discretization than the 100 meter cell discretization.

Baginska et al. (2003) applied AnnAGNPS model to predict export of nitrogen and phosphorous from a small experimental watershed in Sydney, Australia. They applied a model independent, nonlinear parameter estimation code PEST for sensitivity testing and to assess the relative importance of key parameters of the model. Results indicated that predicted phosphorous loads had a high level of sensitivity to assigned pH values for topsoil and also increase of pH value by one unit resulted in up to 34% increase in model generated particulate phosphorus load. Yuan et al. (2003) applied the AnnAGNPS model to a watershed in Mississippi, US and performed a sensitivity

analysis on parameters that are critical to nitrogen loading. Their study revealed that initial nitrogen concentration in the soil and crop nitrogen uptake had the most impact on the nitrogen loadings.

Yuan et al. (2003) performed a sensitivity analysis to identify parameters that most significantly affect on N loading in an agricultural watershed in Mississippi, US. Yuan et al. (2003) concluded that, initial organic and inorganic N concentrations in the soil and crop N uptake had the most impact on N loadings.

Fisher et al. (1997) pointed two types of sensitivity analysis that are recognized in the watershed modeling. The first method, attribute sensitivity analysis, is widely used in the model validation and is commonly called, sensitivity analysis. This method is used to examine percentage change in model output results by changing input parameters. This is performed by systematically changing individual input parameters one at a time by some constant percentage and percentage variation of output is observed. There are several limitations in this method. The second type, resolution sensitivity analysis, is appropriate to any distributed model where the parameters are being sampled over space or time. In this method, sensitivity of a model is assessed by varying the sampling interval of input parameters in geographical space. Input spatial data are subjected to random mixing spatially, to varying degrees, such that the organized landscape become disorganized. They applied spatial sensitivity analysis for two models AGNPS and ANSWERS. The results showed that model outputs are insensitive to the spatial pattern of single input variables and the most of output variables showed absolutely no change as a result of the mixing. Only infiltration-related inputs produced variations in sediment and nutrient yield output.

CHAPTER 3

REVIEW OF THE AnnAGNPS MODEL

3.1 ANNAGNPS MODEL REVIEW

AnnAGNPS is a continuous, distributed parameter watershed model developed by US Department of Agriculture (Bingner and Theurer, 2005). This model was developed based on the original single event model Agricultural Non-Point Source (AGNPS) (Young et al., 1989) developed by Agricultural Research Services (ARS) and NRCS to predict non-point source pollutant loadings within agricultural watersheds. AnnAGNPS is suitable to simulate long term sediment and chemical transport from ungaged agricultural watersheds. Source accounting function is one of the distinctive features of the model. AnnAGNPS can generate loadings for the entire simulation period at user defined locations and calculate the contribution of each location as a ratio to the loadings at watershed outlet. This feature is particularly useful in identifying critical areas in a watershed and can be used to assist in determining BMPs and for risk and cost/benefit analyses (Bingner and Theurer, 2005).

3.2 ANNAGNPS MODEL STRUCTURE

A number of modules that are supplied with AnnAGNPS model can be used in the preparation of AnnAGNPS database. The input output structure and the suite of modeling components contained within AnnAGNPS model are shown in Figure 3.1. The basic modeling components are hydrology, sediment, nutrient and pesticide transport. The model requires physical parameters of the watershed, soil data, climate data, land use and management data.

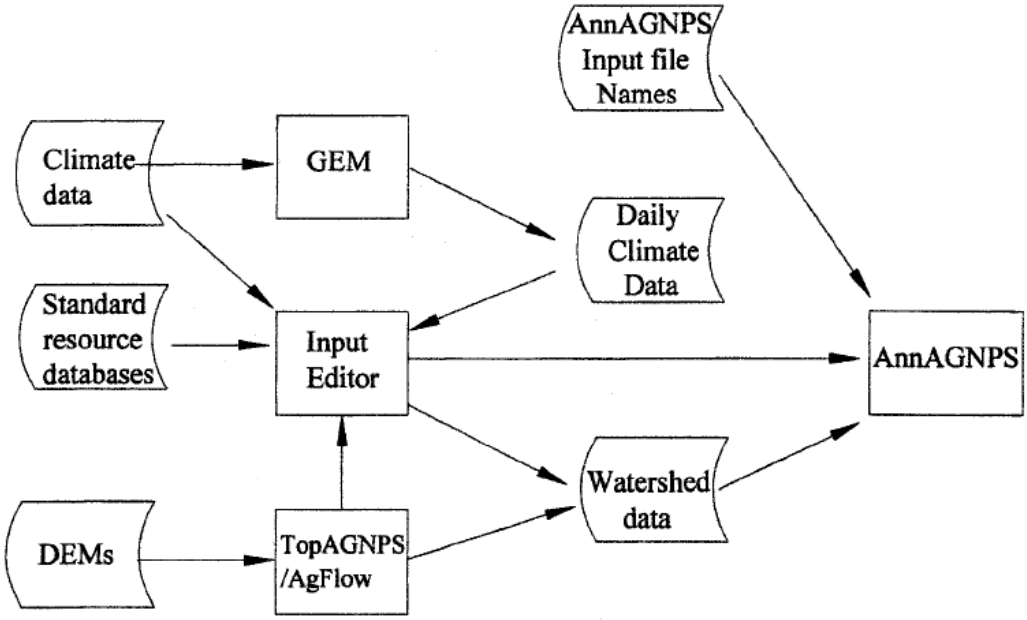

Figure 3.1: Input-Output structure of the AnnAGNPS model (adapted from Bingner and Theurer, 2005).

The physical parameters of the watershed such as cell and stream network information can be extracted from watershed digital elevation models (DEMs) using TOP AGNPS. AGFLOW is used to determine the topographic related input parameters for AnnAGNPS and to format the TOP AGNPS output in the form needed by AnnAGNPS (Bingner and Theurer, 2005). These physical parameters of the watershed are held constant throughout the simulation period. Climate data can be either generated using the Generation of Weather Elements for Multiple Applications (GEM) program or can be generated manually using historical data. Graphical input editor assists in developing the AnnAGNPS database. Seasonal data will change according to human activities and will rarely change during a season such as soil, land use and management data are typically specified manually for each cell using AnnAGNPS input editor. AnnAGNPS-ArcView interface program has been utilized to facilitate TOP AGNPS and AGFLOW programs and to export data to AnnAGNPS input editor. Output processor has been used to analyze the results from AnnAGNPS by generating summary of the results in tabular or GIS format (Bingner and Theurer, 2005).

3.3 AnnAGNPS THEORY

3.3.1 Watershed concept

The spatial variability within a watershed is accounted for by dividing the watershed into many homogeneous drainage areas referred to as "cells." Each cell is homogeneous in soil type, land use and land management and represents the landscape within its respective drainage area boundary. A daily soil water balance is maintained, so runoff can be determined when a precipitation event that includes rainfall, snowmelt or irrigation application occurs (Bingner and Theurer, 2005). Simulated drainage areas are integrated together by a network of "reaches" which collectively represent the stream system in the watershed. Basic AnnAGNPS watershed concept is illustrated in Figure 3.2.

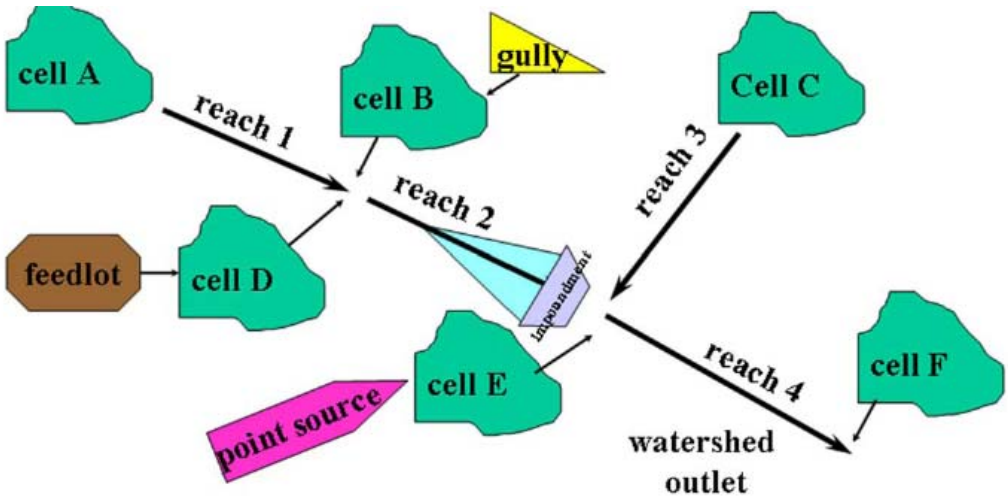

Figure 3-2: Major processes simulated within AnnAGNPS.

3.3.2 Major processes

a. Water Balance

The hydrology component of the model is based on a water balance equation. A schematic of water balance is shown in Fig. 3.3. The erosion calculation is based on whether there has been any runoff for each day. The amount of soil moisture is used to determine the effect of the SCS curve number and is thus the basis for the surface and subsurface runoff in the system (Bingner and Theurer, 2005). AnnAGNPS simulates the soil profile into two layers. First layer is the top 200 mm and the second layer is from the bottom of the first layer to depth of the soil profile. Water balance in a soil layer can be shown as in the following figure. Soil moisture for each time step (i. e., daily time step) is calculated using the equation 3.1.

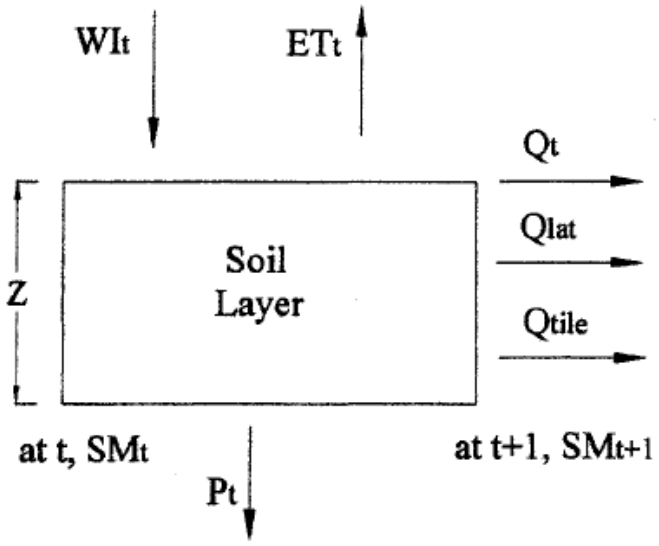

Figure 3-3: Water balance in soil Layer.

$$SM_{t+1} = SM_t + \frac{WI_t - Q_t - P_t - ET_t - Q_{jat} - Q_{tile}}{Z}$$

where SM_t and $SM_{t+}i$ are the moisture contents for each soil layer at the beginning (t) and end of the time period (t+1), WI_t is the water input consisting of precipitation, snowmelt or irrigation water (mm), Q_t is surface runoff (mm), P_t is percolation of water out of soil layer (mm), ET_t is potential evapotranspiration (mm), Qj_{at} is subsurface lateral flow (mm), Qtiie is tile drainage flow (mm), Z is thickness of soil layer (mm) and t is the time period.

b. Surface Runoff

The Soil Conservation Service (SCS) curve number technique is used within AnnAGNPS to determine the surface runoff from a field. The model first calculates a number of parameters in soil moisture calculations such as soil porosity and hydraulic properties, that will remain constant throughout the simulation period. Additional curve number parameters are calculated to vary the curve number for a given day between the dry and wet condition curve numbers based on soil moisture storage.

The average curve number **(CN2)** can change due to operation events, which make significant changes to the land surface such as crop harvesting or during the active growth phase of a crop. These operations primarily change the ground cover and affect the hydraulic properties of the soil and have impact on runoff. The model calculates SCS curve numbers for each cell for a given day corresponding to dry condition or wilting point (CN1) and wet condition or field capacity (CN3) as a function of **CN2.** The actual curve number (CN) associated in calculating runoff is allowed to vary depending on the available soil moisture content.

Surface runoff for each cell is calculated for the current day using the retention variable, S:

$$S = 254\left[\frac{100}{CN} - 1\right] \qquad (3\text{-}2)$$

where S is retention variable (mm) and is related to the soil and cover conditions of the cell through the CN; CN is the SCS curve number. With the value of S calculated for the current day, runoff is calculated as,

$$Q = \frac{[WI - 0.2S]^2}{WI + 0.8S} \qquad (3.3)$$

where Q is runoff (mm) and WI is water input to soil (mm). This equation is valid as long as WI is greater than 0.2 S, otherwise Q is set to zero. WI is equal to the snowmelt amount if a snowpack exists, or the daily precipitation, if no snow is present plus any irrigation water applied. Runoff volume for a cell (Q_t) is obtained by multiplying Q by the cell area.

c. <u>Potential Evapotranspiration</u>

The model uses the Penman equation, a commonly accepted method, to calculate potential evapotranspiration (ET_P). The equation uses standard climatological records of solar radiation (sunshine), air temperature, humidity and wind speed and is given as (Bingner and Theurer, 2005).

$$ETp = \frac{1}{Hv}\left\{\left[\frac{\wedge}{\Delta + \gamma}\right](R - G) + \left[\frac{\gamma}{\Delta + \gamma}\right]W(esat - e)\right\} \qquad (3.4)$$

where ETP is potential evapotranspiration (mm), Hv is latent heat of vaporization (MJ/kg) which is a function of the mean air temperature for a day, A is the slope of saturation vapor pressure-temperature curve, y is psychrometric constant (kPa/ °C), R is net radiation (MJ/m2), G is soil heat flux (MJ/m2) and is calculated as a function of air

temperature for the current day and three previous days, W is wind function, esat is saturation vapor pressure that is a function of air temperature, e is actual vapor pressure (kPa) and is a function of relative humidity. W is calculated using wind speed. Volume of evapotranspiration for a cell (ETt) is obtained by multiplying ETP by the cell area.

d. Subsuface Flow

The model calculates the lateral subsurface flow and tile drain flow to determine the contribution of subsurface flow from each cell to the corresponding reach. Subsurface flow calculation is done only when there is an impervious layer present in the soil profile. In case of tile drainage, the model assumes that a steady constant flow occurs through the soil to the drains (Figure 3.4). When the water table is above tile drains, the model

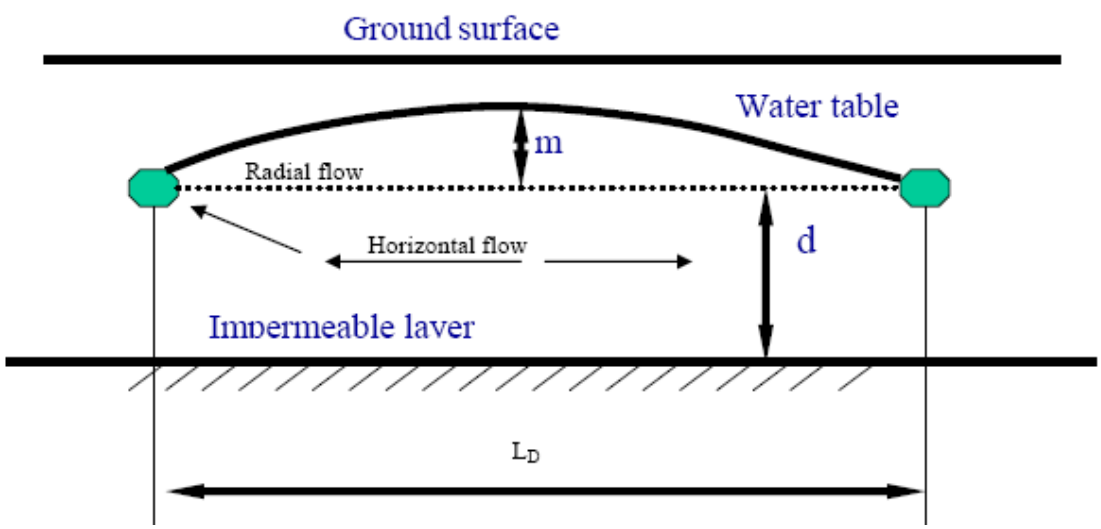

Figure 3-4: Schematic for Houghoudt Tile Flow

calculates drainage flux through pipes using the widely applicable Hooghoudt's equation (Bingner and Theurer, 2005) shown below:

$$q_{drain} = \frac{8K_s d_e m + 4K_s m2}{L^2} \qquad (3.5)$$

where **qdrain** is drainage flux (mm per time period), K_s is saturated lateral hydraulic conductivity (mm per time period), **L** is the distance between tile drains (m), m is midpoint water table height above tile drains (m) and **de** is the equivalent depth of impermeable layer below the tile drain which is a function of **L,** d and radius of tile drain tube. The total tile drainage flow out of each cell (Qtu_e) to corresponding reach is obtained by multiplying q^**am** for the cell by area of the cell.

When the water table is below the depth of the tile drainage system, the model calculates lateral flow using Darcy's equation. Darcy's equation is widely used and provides an accurate description of subsurface flow. In the model, only the saturated flow condition is considered and subsurface flow is assumed to be homogeneous through the entire soil profile.

$$q_{lat} = -Ks\frac{dh}{dl} \qquad (3.6)$$

where qua is subsurface lateral flow (mm per time period), K_s is saturated hydraulic conductivity for each soil layer (mm per time period), *dh/dl* is hydraulic gradient where stream length *dl* represents the length of the cell. The total volume of lateral flow out of each cell **(Qiat)** is obtained by the product of **qiat** and the lateral flow cross section area.

e. <u>Channel Hydrology</u>

AnnAGNPS uses TR55 methodology to calculate peak water discharge for each cell. Flowpath in a cell is divided into a section of overland flow, followed by a section of

shallow concentrated flow and a section of concentrated or open channel flow. Length of overland flow and shallow concentrated flow is assumed to be no longer than a maximum length of 50 m each. The length of the in-cell, concentrated flow is the remainder of the in-cell flow length. In-cell time of concentration for flow to each cell outlet ($T_{Cj}j_{n\_ce}u$) is calculated using travel time for flow in these sections using the equation 3.7:

$$Tc, in\_cell = Tt, ov + Tt, cf \qquad (3.7)$$

where $T_{c>}$**j„_ceii** is time of concentration for the in-cell processes, **Tt,$_0$v** is travel time for the overland flow period, **Tt,$_S$cf** is travel time for the shallow concentrated flow period and **Tt,$_C$f** is travel time for the concentrated flow period. Time of concentration to channel reach outlet is the maximum value of time of concentrations for all reaches flowing into a reach being considered.

AnnAGNPS assumes a triangular shaped hydrograph. The time to base of the hydrograph (i.e. duration of each surface runoff event) is calculated using equation 3.8,

$$t_b = 20(R_q D_a / Q_p) \qquad (3.8)$$

where Q_p is peak discharge (m^3/s), Rq is total runoff volume from upstream drainage area (mm), D_a is total drainage area (ha.) and tb is the time base of the hydrograph (s). The ratio of initial abstraction, I_a to 24-hour precipitation is needed to calculate the peak discharge for each cell during each runoff event. The model calculates initial abstraction using the equation 3.9,

$$I_a = (P_{24} + 2\,Q_{24}) - (5\,Q_{24}\,P_{24} + 4\,Q_{24}^2)^{0}\text{-}^{5} \qquad (3\ 9)$$

where Q_{24} is 24-hour runoff. The model calculates the peak discharge for each runoff event using the following equation,

$$qp = 2.7778\ \text{X}\ 10''3P24\ \ Da\left(\frac{a + (cTc) + (eTc2)}{l + (6Tc) + (tfTc2) + (frc3)}\right) \qquad (3.10)$$

where qp is peak discharge (m^3/s), D_a is total drainage area (ha.), T_c is time of concentration (hr.) and *a, b, c, d, e* and/are the unit peak discharge regression coefficients for a given **(I_a/P24)** and rainfall distribution type.

f. <u>Sediment</u>

AnnAGNPS uses Revised Universal Soils Loss Equation (RUSLE) (Renard et al., 1997) technology to predict sheet and rill erosion from cells. RUSLE technology within AnnAGNPS calculates LS, C, P factors for each cell in the watershed and a K factor for each soil in the watershed. When factors other than rainfall are held constant, soil losses from agricultural fields are directly proportional to a rainstorm parameter called EI. The value of EI for a given rainstorm equals the product of total storm energy (E) times the maximum 30-minute rainfall intensity **(I30)** (Renard et al., 1997). RUSLE uses EI value to determine the erosion within a cell and is calculated for a given rainfall distribution type and the rainfall amount using equation 3.11,

$$EI = \frac{A \exp(2.119 \log(P) \exp(0.0086 \log(24)))}{\exp(51og(24))} \qquad (3.11)$$

where EI is energy intensity, P is precipitation or snowmelt, *A* and *B* are EI coefficients used in AnnAGNPS for different cumulative rainfall distributions. The total potential erosion within each cell is calculated using

$$A = R * LS * K * C * P \qquad (3.12)$$

where A is total potential erosion, R is the annual rainfall-runoff erosivity factor, LS, K, C and P are RUSLE coefficients. Potential erosion is multiplied by the sediment delivery ratio to determine the amount of sediment delivered to the edge of the field. The Hydro-geomorphic Universal Soil Loss Equation (HUSLE) is used to determine the delivery ratio for total sediment. The sediment delivered into the edge of the cell is broken into five particle size classes: clay, silt, sand, large aggregate and small aggregate. All

sediment routing in the concentrated flow channels used within AnnAGNPS are performed using five particle size classes and for each increment of the hydrograph.

g. <u>Chemical Routing</u>

AnnAGNPS recognizes three nutrients: nitrogen, phosphorous and organic carbon. Nitrogen and phosphorous are considered to be able to exist in both the dissolved and attached forms. chemical forms as shown in equation 3.13

$$Ms = \frac{Mc}{1 + Kd} \qquad (3.13)$$

where M_s is total mass of chemical in solution (Mg), M_e is total mass of chemical both attached and in solution (Mg) and K<j is partition coefficient of chemical (Mg).

CHAPTER 4

MODELDEVELOPMENT

4.1 Al-KFAIR WATERSHED - AN OVERVIEW

The study area lies to the North – East of Amman and covers an area of about 50 km^2 as shown in Figure 4.1. According to Palestine Grid, Al Kfair catchment area lies between 234 to 241 East and 177 to 184 North. The study area extends from king Talal dam to Balama east and considers as a part of the Eastern Plateau of Jordan (Bender, 1968).

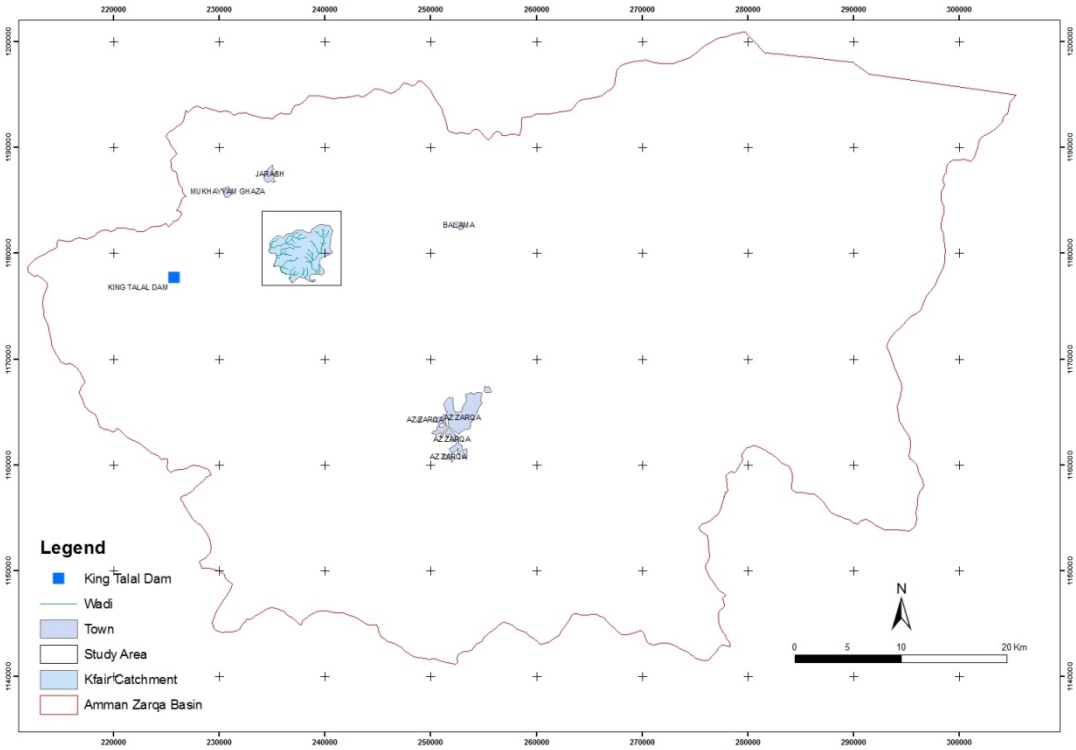

Figure 4.1 : Study area Vicinity map.

From Hydrogeological view point it considered as a part from Amman – Zarqa Basin. The topography of the study area has irregular elevation. However, the elevation values range from 228-842 m above mean sea level in the study area. Figure 4.2, shows the topographic map and drainage system of the study area.

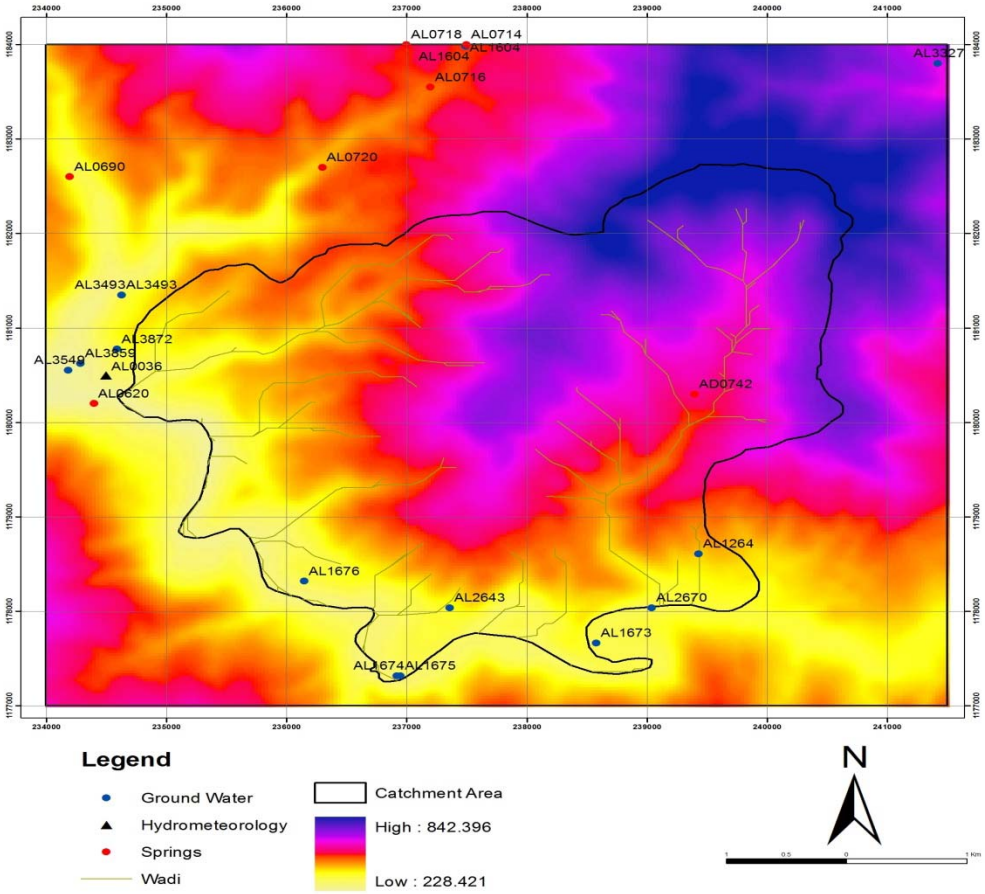

Figure 4.2: Topography and the Drainage System of the study area.

4.1.1 Land use

Al-kfair watershed is predominantly with Natural vegetation which constitutes approximately 64.2 %. Urban and rural residential land uses are about 7% of the watershed area that consist of Former Township of Al-kfair. Forest areas constitute approximately 1.5% of the watershed area and are mostly located at the center of Al-kfair area. Agriculture constitutes around 2% of the watershed area and the predominant types of crops produced include Cereals and Vegetables**.** The rainfed agricultural area constitutes approximately 25.3% from the watershed area and distributed as follow:

- ✓ Non Deciduous Trees (Olives/Bananas/Citrus) (4.1%)
- ✓ Annual Crops (Cereals/Vegetables) (10.5%)
- ✓ Deciduous Trees (Fruit Trees) (10.8%)

4.1.2 Climate

According to the international weather classification, the climate of AZB has semi-arid condition, the temperature in the basin varies in an east-west direction, the average annual minimum and maximum temperatures are 11.1 "C and 23.5 "C, respectively. The total annual average precipitation volume in Amman-Zarqa Basin is calculated to be about 892 MCM/Year (1937-2009) and the average annual precipitation in the western part of the basin is about 400 mm, while the average annual precipitation in the eastern part is about 150 mm. The annual precipitation ranges from less than 50 mm in the South and Southeast Region more than 550 over mountainous region in the North and Northwest of the basin.

4.1.3 Soil Characteristics

Detained soil information was obtained from the Ministry of Agriculture (MoA) Database. MoA provides most of soil parameters needed for AnnAGNPS simulation, such as soil texture, erosive factor, hydraulic properties, and organic matter. Information on soil nutrient contents was estimated based on soil organic matter. Geographical Information System (GIS) soil maps were used in conjunction with the sub-watershed maps to determine the predominant soil assigned to each AnnAGNPS cell. Soil parameters were formatted using the AnnAGNPS Input Editor.

4.2 OVERVIEW OF THE METHODOLOGY

4.2.1 Introduction

AnnAGNPS requires about 400 parameters in 34 different data categories to describe a watershed including topographic data, soil and land-use related data, and climate data. These data are prepared and organized using the component modules provided with the AnnAGNPS program. The modules TopAGNPS, a subset of the Topographic Parameterization (TOPAZ) and Agricultural watershed FLOWnet generation program (AGFLOW) are used to generate spatially varying drainage densities and subcatchment areas. AnnAGNPS cells are hydrologically determined by varying critical source area and minimum source channel length values. Physical cell parameters such as area, length, slopes and LS parameters are determined by AGFLOW module. AnnAGNPS cells are homogeneous in soil and land use type. Dominant soil and land use type for cells are assigned by superimposing the soil and land use shape files over the delineated sub-watershed shape file.

AnnAGNPS Input Editor and AnnAGNPS-Arcview interface programs are used to develop the input file "AnnAGNPS.inp", required for AnnAGNPS pollutant loading model. Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997) has been implemented in AnnAGNPS to predict the sheet and rill erosion resulting from raindrop splash and surface runoff. Soil-Plant-Air-Water (SPAW) (Saxton and Wiley, 2005) computer model that simulate the daily hydrologic water budgets of agricultural lands is used to derive soil water relationships such as wilting point, field capacity and hydraulic conductivity using soil characteristics such as soil texture and organic matter.

Climate data is of great importance for AnnAGNPS simulation and results are largely dependent on the quality of the climate data used in the simulation.

4.2.2 Modules for model development

a. TOpographic PArameteriZation program (TOPAZ)

TOpographic PArameteriZation program (TOPAZ) (Garbrecht and Martz, 1999) is used to generate spatially varying drainage densities and subcatchment areas from digital elevation models (DEMs). DEMs are numerical representations of the elevations of a surface that has been divided into regularly spaced grids (Martz and Garbrecht, 1993). TOPAZ generates raster output files of the drainage network, subcatchment areas and a variety of drainage-related topographic variables that can be easily imported to Geographic Information Systems (GIS) for display.

TOPAZ consists of three programs: Digital Elevation Drainage Network Model (DEDNM), RASter PROperties (RASPRO) and RASter FORmatting (RASFOR) and each program requires, input files provided by the user and input files that are generated internally. For hydrographic landscape discretization and channel network generation, two important parameters: the critical source area (CSA) and the minimum source channel length (MSCL) are defined. These two parameters control the topology and properties of the network and sub-catchments generated by TOPAZ. For example, a higher CSA parameter value results in a drainage network with a lower drainage density, and a higher MSCL parameter value results in a drainage network without short, first order source channels (Garbrecht and Martz, 1999).

b. AGricultural watershed FLOWnet generation program (AGFLOW)

The AGricultural watershed FLOWnet generation program (AGFLOW) (Bingner et al., 1997) is used to determine the topographic related input parameters for AnnAGNPS and to format the TOPAGNPS output for importing into the form needed by AnnAGNPS. This program uses FLOWGEN.INP as input data file. The input file

consists of data such as: AnnAGNPS input and output path, AGFlow output path, values for CSA, MSCL and watershed outlet location.

c. Input Editor

AnnAGNPS Pollutant Loading (PL) model requires two input files: AnnAGNPS.inp and DayClim.inp. AnnAGNPS input editor is used to import data created from TOPAGNPS and AGFLOW, in order to develop AnnAGNPS.inp input file. DayClim.inp, input file created with historically recorded data are imported into the daily climate data section of the input editor program. Also Input Editor is used to compile other input parameters required by PL model. Input Editor Program facilitates exporting data as text, comma separated or as database files.

d. AnnAGNPS-Arcview interface

The AnnAGNPS-Arcview interface is used to simplify the use of TOPAGNPS and AGFLOW modules and to derive cell and reach data required by the AnnAGNPS Input Editor. The interface uses information extracted from the DEM to identify and quantify topographic features and land surface processes based on values of hydrology, drainage characteristics and elevation data and user defined outlet row and column values to generate the TOPAGNPS files (USDA-ARS, 2005).

e. Revised Universal Soil Loss Equation (RUSLE)

AnnAGNPS uses RUSLE (Renard et al., 1997) module to estimate soil loss resulting from sheet and rill erosion in farm fields. RUSLE is the modified version of USLE developed by Wischmeier and Smith in 1978. This equation is very robust and has been used in many research studies related to sedimentation in watersheds (Montgomery et al., 1997). RUSLE computes the average annual erosion by using a functional relationship of six factors, as is given by equation 3.12.

f.Hydro-geomorphic Universal Soil Loss Equation (HUSLE)

Whenever a runoff event occurs, RUSLE calculates the amount of sheet and rill erosion in a field but does not calculate field deposition. In AnnAGNPS, Hydro-geomorphic Universal Soil Loss Equation (HUSLE) is used to determine the amount of sediment delivered to the stream that is generated from sheet and rill erosion. HUSLE calculates the total sediment yield at a point, for a given storm event using average

RUSLE parameters, upstream drainage area, peak discharge and volume of surface runoff.

4.3 DEVELOPMENT OF INPUT DATABASE

To model runoff, sediment erosion and nutrient transport at the watershed scale, a large amount of input data had to be acquired, organized, and stored. This is achieved using the computer modules described in section 4.2.2 and assembling many sources of information such as soil data, landuse details, field management practices and weather information. Topographic information is crucial in determining the watershed and sub-watershed boundaries, channel locations, channel slopes, flow routing, field slopes, flow travel times, the RUSLE LS-factor, aspect and elevation of fields. Most of the required parameters are obtained as described above, while some data such as channel geometry parameters were to be measured in the field in order to represent them realistically. Data related to agricultural activities such as crop growth parameters, crop rotations, operation management data and fertilizer application data are obtained in consultation with the farmers in the study area.

4.3.1 Digital Elevation Models (DEMs)

The use of DEMs provides a convenient source of topographic information. The DEM for the watershed having a resolution of 30 m x 30 m. Stream locations are important in generating AnnAGNPS components. During the field visits to Kufier watershed, it was observed that some modifications to the GIS layers were necessary with regard to the stream network.

4.3.2 Drainage boundary and sub-drainage areas

TOP AGNPS program is used to generate drainage boundary and watershed outlet from the DEM for the watershed. AnnAGNPS-Arcview interface accesses the TOP AGNPS and AGFLOW files, and the DEM in generating the required Arcview shape files from which the necessary data can be extracted. The sub-drainage areas of the

watershed were discretized into AnnAGNPS cells based on the spatial variation of landuse and the location of the digitized stream network. Delineation process started with an assumption of the critical source area (CSA) and minimum source channel length (MSCL) required with the use of TOP AGNPS. The CSA parameter represents the threshold drainage area below which a channel is assumed to form (Garbrecht and Martz, 1999). Larger CSA values produce sparse drainage network and smaller values produce dense drainage network. Also, larger CSA and MCSL values produce large sub-catchment areas and smaller values produce small sub-catchment areas. various combinations of CSA and MSCL were used for watershed delineation are shown in Table 4.1, and numbers of cells and reaches generated from each combination of CSA and MSCL values are also listed in Table 4.1.

Table 4.1: Cell and reach numbers within the study area using different CSA and MSCL values.

Type of delineation	**CSA parameter (ha)**	**MSCL parameter (meters)**	**Number of cells**	**Number of reaches**
1	8	130	344	90
2	20	200	194	46
3	30	300	147	34
4	50	500	64	16

For Al- Kfair watershed, an initial 8.0 hectare CSA and 130 meter MSCL values were selected to produce AnnAGNPS cells. This initial subdivision produced 344 AnnAGNPS cells distributed throughout the watershed. Since watershed was not being adequately divided to capture the spatial variability of land use, some of the AnnAGNPS cells were selected for further subdivision using different CSA and MSCL values. The process of starting with larger cell sizes and working to subdivide only those areas needed to capture landuse features, provides the simplest approach in deriving AnnAGNPS cells. Several combinations of CSA and MCSL values were tested and final subdivision of Al- Kfair watershed produced 147 AnnAGNPS cells and 34 stream segments or reaches.

4.3.3 Land use data

Soil erosion from agricultural lands is heavily dependent on the landuse type and hence it is critical to define the landuse with a greater accuracy. For the Al- Kfair watershed historical landuse data were not readily available. Arcview shape files obtained from MoA for urban, Natural vegetation and Trees areas were superimposed in developing a composite landuse GIS layer.

4.3.5 Soil data

Within the Al- Kfair watershed, seven different soil types are identified from the soil GIS layer. The predominant soil type is limestone. In AnnAGNPS, for the purposes of runoff generation and soil water storage, the soil profile is divided into two layers. The top 200 mm is used as a tillage layer whose properties such as, bulk density, can change. The remaining soil profile comprises the second layer whose properties remain static. For each layer in the soil profile, clay, silt, sand, rock and very fine sand ratio was input. In addition, saturated hydraulic conductivity, field capacity, wilting point, base saturation, pH and organic matter were input. Most of these data were obtained from the available databases and a program developed by Agricultural Research Service of United States Department of Agriculture (Saxton and Wiley, 2005). Soil organic matter, which is an important component of the nutrient cycle, holding soil moisture and soil structure, was also obtained from Richards et al. (1949) for surface soil layers. The above parameters for sub-soil layers were obtained assuming decreasing organic matter contents for sublayers.

Default values provided in the model were used for initial nutrient content (organic and inorganic nitrogen, organic and inorganic phosphorus). Hydrologic soil group for each soil was obtained from the soil GIS layer.

4.3.6 Weather data

AnnAGNPS model has a supporting module GEM for synthetic weather generation based on historical values for nearby weather stations. For this simulation, historical climate

data obtained from Um-Jemal weather station, which is the nearest station with complete data record to Al-Kfair watershed, is used. Daily climate data required for the model such as minimum and maximum temperature and precipitation, dew point temperature, wind speed and sky data are obtained from MoWI. The available hourly data on these parameters was converted to daily data as required for the model input. Daily precipitation is the prime driver of the hydrologic cycle; temperatures are used to define frozen conditions and remaining climate parameters are used in the model to compute potential evapotranspiration using Penman equation.

To represent the current weather conditions in the simulation, all the above climate parameters were obtained for the 5-year period from 2003 to 2007.

4.3.7 RUSLE parameters

a. <u>Rainfall-runoff erosivity factor (R)</u>

The value "R" quantifies the effect of raindrop impact and reflects the amount and rate of runoff likely to be associated with a given rainfall event. It represents two most important characteristics of storm erosivity: amount of rainfall and peak intensity sustained over an extended period of time. The greater the intensity and duration of the rain storm, the higher the erosion potential. "R" is the average annual total of the storm Energy Intensity (EI) values for a given area. The value of EI for a given rainstorm is equal to the product of total storm energy (E) times the maximum 30-min intensity **(I30).** The storm energy indicates the volume of rainfall and runoff and **I30** component reflects the prolonged peak rates of detachment and runoff. The product EI is a statistical interaction term that reflects how total energy and peak intensity are combined in each particular storm (Renard et al., 1997).

In AnnAGNPS, EI value for a given rainfall distribution type and the rainfall amount is determined using the equation 3.11. The annual rainfall-runoff erosivity factor R is the sum of the energy intensity values for all the storms in a given year. R factor is derived based on rainfall intensity data over extended periods (Renard et al., 1997) and is expressed by the equation:

$$R = \sum (EI30)/N \quad (4,1)$$

where R **is** rainfall-runoff erosivity factor in MJ.rnm.ha"^{1}.h"^{1}. yr*^{1}, EI30 for **1th** storm and j is the number of storms in an N year period. The distribution of erosive rains differs significantly with the geographical locations.

b. <u>Soil erodibility factor (K)</u>

Soil erodibility may be thought as the ease with which soil is detached by splash during rainfall or by the surface flow. The K factor represents the average long-term soil and soil profile response to the erosive powers of rainstorms (Renard et al., 1997). The physical, chemical and mineralogical soil properties and their interactions affect K value. K factor is affected by antecedent soil-water and soil surface conditions and seasonal variations of soil properties (Renard et al., 1997). When the soil surface contains rock fragments, it reduces soil detachment by rainfall and reduces soil erosion. When rock fragments present in a coarse textured soil profile (having sand and loamy sand textures), it reduces infiltration and increases soil erosion. In RUSLE, rock fragments in surface soil is accounted in C factor. Subsurface component is accounted in K factor through adjustments of the permeability of soil (Renard et al., 1997).

Coarse textured soils such as sandy soils, have low K values, about 0.05 to 0.2, since they have low runoff even though they can get detached easily. Clayey soils are resistant to detachment and they also have low K values, about 0.05 to 0.15. Medium textured soils, such as silt loam soils are moderately susceptible to detachment and they produce moderate runoff and have K values, about 0.25 to 0.4. Soils with high silt contents are easily detached and are most erodible of all soils. They produce high rates of runoff and K values for these soils tend to be greater than 0.4. Soils with high organic matter content are less erodible because it increases infiltration and reduces the susceptibility of the soil to detachment by reducing overland flow and thus erosion. Soils that are most prone to sheet and rill erosion are those with relatively high sand content, low in organic matter and clay.

AnnAGNPS uses soil nomograph equations to calculate K factor for each soil in the watershed or optionally user can input K factor values for each soil (Bingner and

Theurer, 2003). Stone and Hillborn (2002) suggested soil erodibility factors for soils with different soil textural classes and having average organic matter content. Novotny (2003) suggested values of K depending on soil texture and having organic matter contents of 0.5, 2 and 4 percent. Table 4.2 lists general magnitudes of K values for different soil textural classes and for average organic matter content.

c. Topographic factor *(LS)*

Erosion increases as slope length increases and is accounted in slope length factor (L). As the slope steepness increases, soil loss increases and considered in slope steepness factor (S). In erosion prediction calculations the factors L and S are usually evaluated together and the effect of topography on soil erosion is accounted for by the combined topographic factor (LS). The LS factor represents a ratio of soil loss under given conditions to that at a site with the standard slope steepness of 9% and slope length of 72.6 feet.

Table 4.2: Soil erodibility factor -K for different soil textures

Soil textural class	Stone and Hillborn (2002)	Novotny (2003)
Clay	0.22	0.17
Silty Clay	0.26	0.23
Clay Loam	0.30	0.25
Silty Clay Loam	0.32	0.32
Sandy Loam	0.13	0.24
Sandy Clay Loam	0.20	0.25
Fine Sandy Loam	0.18	0.30
Sand	0.02	0.03
Loamy Sand	0.04	0.10
Fine Sand	0.08	0.14

Very Fine Sand	0.43	0.36
Silt Loam	0.38	0.42
Very Fine Sandy Loam	0.35	0.41
Silt	—-	0.52

Renard et al. (1997) showed, for average erosion, slope length factor (L) varies with slope length *X* as:

$$\mathbf{L} = (X/72.6)^{m} \qquad (4\text{-}2)$$

where 72.6 is the RUSLE unit plot length in feet, m is a variable slope length exponent. The slope length % is the horizontal projection of the slope in feet.

Procedures have been developed to calculate the LS factor for multiple cells using the slope geometry from DEMs for the watershed. Moore (1992), based on erosion theory, developed a relationship to determine the LS factor using the sub-watershed area (A) and average slope (5). Moore's equation to calculate the LS factor is:

$$\mathbf{LS = (A/22.13)^{\circ\text{-}4} * (Sin5/0.0896)^{L3}} \qquad \mathbf{(4.3)}$$

where A is the subwatershed area and 5 is the average slope angle. During input database development, LS values generated for each cell from the AnnAGNPS-ArcView interface procedure were extracted into AnnAGNPS input editor.

d. <u>Cover management factor (C)</u>

Cover management factor reflects the effect of cropping and management practices on erosion rates and is the factor mostly used to compare the impacts of management options on conservation plans. "C" represents the ratio of soil loss from an area with specific cover and management to soil loss from a standard plot. In this case, standard plot is considered as an area under clean-tilled and continuous fallow condition

(Renard et al., 1997). C values range from 1.0 where there is little soil cover to values less than 0.10 where there is dense cover and large amounts of crop residues left on the soil surface.

In evaluating C factor, impact of cropping and management on soil loss is generally divided into series of sub-factors such as, impact of previous cropping and management, impact of vegetative canopy on soil surface, the reduction in soil loss due to surface cover and surface roughness and impact of low soil moisture on reduction of runoff from low intensity rainfall. Each of the subfactors is assigned a value and Soil Loss Ratio (SLR) is obtained by the following equation:

$$SLR = PLU*CC*SC*SR*SM \qquad (4\text{-}4)$$

where PLU is the prior land use subfactor, CC is the canopy cover subfactor, SC is the surface cover subfactor, SR is the surface roughness subfactor and SM is the soil moisture subfactor. Each of the subfactors contains cropping and management variables that affect soil erosion.

` Usually, in croplands, soil and crop parameters vary with time due to either specific management practices or due to climate changes. This makes that the SLR values be calculated frequently enough to capture those variations. Average annual soil loss can be high if cropping and management operations occur during higher rainfall erosivity. RUSLE module in AnnAGNPS calculates SLR values every 15 days throughout the year to incorporate this effect. Once SLR for each time interval is calculated they are multiplied by their corresponding percentage of annual EI values. Cover management factor (C) can be calculated using the following equation:

$$\mathbf{C = [\ SLRi*EIi+SLR_2*EI_2 + . . + SLRj*EIj + . . + SLR_{„}*EI_{„}\ ] / EI_t} \qquad (4.5)$$

where C is the average annual cover management factor, SLRj is the soil loss ratio value for time period j, EI is the percentage annual EI occurring during that time period, n is the

number of periods considered and EI_t is the sum of EI percentages for the entire time period. In the development of input database, canopy cover, surface cover, surface roughness and droplet fall heights for various crops are obtained from Renard etal., (1997).

e. Support practice factor (P)

Support practice factor (P) is the ratio of soil loss with a specific support practice such as contouring, strip-cropping or terracing, to the soil loss with straight-row farming with rows oriented parallel to the slope gradient. Generally support practices affect erosion by modifying the flow pattern, grade or direction of subsurface runoff. Support practice factor reflects the effects of practices that will reduce the amount and rate of the water runoff and thus reduce the amount of erosion. Support practice factor varies from 1.0, when there are no support practices to 0.1 to 0.05 for areas with practices such as terracing.

4.4 SENSITIVITY ANALYSIS

4.4.1 Introduction

In Al-Kfair watershed, **there are no measured values for surface runoff, sediment yield and nutrient loadings and hence no calibration and validation of the model can be performed. In such circumstances, a high uncertainty in model simulations could be expected. To increase the confidence in the model predictions and to improve the understanding of the model behavior, a sensitivity analysis was conducted as part of the model development.** Several parameters, that may have significant influence on the runoff, sediment and nutrient loadings, are selected for sensitivity analysis, based on the model processes and the results of previous studies reported in literature.

4.4.2Method of analysis

The most common method used in sensitivity analysis is to examine percentage change in model output results by changing input parameters one at a time by some constant percentage. The method used in sensitivity analysis reported by Vieux and Needam (1993), was to keep all other variables constant while varying one parameter at a time, by ±25% and ±50% and measuring the change relative to a base value. Ma et al. (2002) performed sensitivity analysis to test eight parameters related to soil and phosphorus sediment, on model output results. In their study they kept all other variables constant while varying one parameter at a time, by ±10%.

Variation by a fixed percentage of the initial parameter value may sometimes cause unrealistic results. If the initial parameter value is located close to the upper or lower bound of the valid parameter range, the variation by a fixed percentage can cause inadmissible values beyond the valid parameter range. Therefore, in place of this conventional method, an alternative approach to define the parameter variation is to be considered. Chaubey et al. (1999) performed a study on uncertainty in the model parameters due to spatial variability of rainfall parameters. In their study, a relative sensitivity index was used to rank the model parameters in terms of their sensitivities in affecting the model outputs. In Lenhart et al. (2002) approach, the parameter values are not varied by a fixed percentage of the initial value, but they are varied by a fixed percentage within the valid domain of the parameter value. Sensitivity is expressed by a dimensionless index I, which is calculated as the ratio between the relative change of model output and the relative change of an input parameter. Mathematically, the partial derivative *dyldx* is used to represent the dependence of a variable y on a parameter x. This expression may be numerically approximated by a finite difference. Let yo be the model output calculated with an initial value xo of the parameter x. Let this initial parameter value be varied by ± Δx, yielding xj = xo - Ax and x_2 = xo + Δx. Let yi and y_2 are the corresponding values for $x_i = x_o - \Delta x$ and $x_2 = x_o + \Delta x$. Then the finite approximation of the partial derivative *dyldx* can be written as, $I = (y_2 - y_1)/ (2\ \Delta x)$.

To get a dimensionless index, F is normalized by dividing with the corresponding initial values. The expression for the sensitivity index I then assumes the form The sign of the sensitivity index I, indicates whether the model output change co –directionally to the

input parameter change, i.e., if an increase in the parameter leads to an increase of the output variable and decrease of the parameter to a decrease of the variable. In order to assess the effect of parameter sensitivity , the calculated sensitivity are ranked into four different classes as shown in table 4-3.

$$I = \frac{(y_2 - y_1)/y_o}{2\Delta x/x_o} \tag{4.6}$$

Table 4-3: Sensitivity classes (Lenhart et.al., 2002)

Class	Index	Sensitivity level
I	$0.00 \leq \lvert I \rvert < 0.05$	Small to negligible
II	$0.05 \leq \lvert I \rvert < 0.20$	Medium
III	$0.20 \leq \lvert I \rvert < 1.00$	High
IV	$\lvert I \rvert \geq 1.00$	Very high

This approach is followed in the present study to determine the sensitivity of the parameters selected in the following section except the cell size. In this approach parameter value, x is varied by $\Delta x = \pm 10\%$ and $\pm 20\%$ as applicable within the entire valid domain of the parameter. The value x_0 is the parameter value assigned for the base case, based on the available data sources.

4.4.3 Parameters selected for the analysis

It has been recognized by several studies that the scale of cell discretization affects the model results significantly with respect to runoff and sediment loading (Vieux and Needam, 1993, Qiu et al., 1997). Therefore, the sensitivity of cell discretization on model results has been investigated to determine the critical cell size that is to be used in this simulation. In AnnAGNPS, each cell or subcatchment is assumed to be homogeneous in landuse, land management and soil type. Predominant landuse and soil type within a cell is considered distributed homogeneously within that cell. Therefore cells sizes should be selected such that they are able to capture the spatial variability of land uses and other features and hence cell size selection should not be done arbitrarily. Several parameters such as RUSLE topographic (LS) factor, channel network and flow path lengths, cell time of concentration are dependent on cell sizes. Different cell discretization would produce different model inputs, thus causing differences in the model output results.

Based on the reviews performed and the reported literature, thirteen AnnAGNPS parameters are selected for sensitivity analysis. They are listed in Table 4.4 with the corresponding run configurations.

Table 4.4: List of parameters used in the sensitivity analysis

No.	Parameter
1	K factor
2	Wilting point
3	Field capacity
4	Initial organic phosphorus in soil
5	Initial inorganic phosphorus in soil
6	Organic matter content
7	Organic N
8	Initial organic nitrogen in soil
9	Initial inorganic nitrogen in soil
10	Plant P uptake
11	Plant N uptake
12	Surface roughness
13	Curve number

4.4.4 Simulation method

In order to observe the effect of cell sizes on model output results and to determine appropriate cell discretization that is to be applied in the simulation study, sensitivity analysis is performed for seven different cell discretizations. Watershed is divided into 33 cells to 700 cells having average cell sizes ranging from 1.0 ha. to 20 ha. for different cell discretizations.

Soil related parameters, K factor, wilting point, field capacity, and organic matter content of soils are varied by $\Delta x = \pm 10\%$ of their base case values. Soil initial organic and inorganic nitrogen ratios were set at model default values in the base case. They were tested with the values 50 ppm for the top layer and 5 ppm for the subsequent layers for organic nitrogen ratio and 5 ppm for the top layer and 0.5 ppm for the subsequent layers for initial inorganic nitrogen ratio.

Crop related parameters, plant N uptake, plant P uptake and the parameter was tested to see the effects on N and P loading. Plant N and P uptake were set to model default values in the base case. N and P uptake values for wheat was set to literature values and tested individually and also tested together to see the effect on N and P loading. Plant uptake values for N and P used in the simulation are shown in the table 4.5.

Table 4.5: Plant N and P uptake values

Crop	N uptake	P uptake
	(kg/kg of harvest)	
Wheat	0.022	0.0025

Surface roughness resulting from roots or any other vegetative effects on the surface is represented by random roughness parameter. This parameter was set to baseline value and tested with $\Delta x = +10\%$ and $+20\%$ to see the effects on erosion.

For the base case, runoff curve numbers for different land use and management and hydrologic soil groups were set to the values shown in Table 4.6. Runoff CNs are varied such that they do not go beyond the valid parameter range for respective hydrologic soil group.

Table 4-6: Runoff curve numbers for hydrologic soil cover (Antecedent moisture condition II, and Ia=0.2S)

Cover			Hydrologic Soil Group			
Land Use	Treatment or Practice	Hydrologic Condition	A	B	C	D
Fallow	Straight Row	---	77	86	91	94
Row Crops	Straight Row	Poor	72	81	88	91
		Good	67	78	85	89
	Contoured	Poor	70	79	84	88
		Good	65	75	82	86
	Terraced	Poor	66	74	80	82
		Good	62	71	78	81
Small Grain	Straight Row	Poor	65	76	84	88
		Good	63	75	83	87
	Contoured	Poor	63	74	82	85
		Good	61	73	81	84
	Terraced	Poor	61	72	79	82
		Good	59	70	78	81
Close-seeded Legumes or Rotation Meadow	Straight Row	Poor	66	77	85	89
		Good	58	72	81	85
	Contoured	Poor	64	75	83	85
		Good	55	69	78	83
	Terraced	Poor	63	73	80	83
		Good	51	67	76	80
Pasture or Range	Natural	Poor	68	79	86	89
		Fair	49	69	79	84
		Good	39	61	74	80
	Contoured	Poor	47	67	81	88
		Fair	25	59	75	83

Cover			Hydrologic Soil Group			
		Good	6	35	70	79
Meadow	Natural	Good	30	58	71	78
Woods	Natural	Poor	45	66	77	83
		Fair	36	60	73	79
		Good	25	55	70	77
Farmsteads	---	---	59	74	82	86
Roads	(dirt)	---	72	82	87	89
	(hard surface)	---	74	84	90	92

4.5 AGRICULTURAL BEST MANAGEMENT PRACTICES

4.5.1 An overview

In general, not all the areas in a watershed, contribute sediment and nutrients to receiving waters. Small areas of the landscape with specific soil characteristics and agricultural practices are often responsible for a majority of the sediment and nutrient loading to surface waters. For best results, an effective Best Management Practices (BMP) or a combination of BMP's must be implemented in these areas that are most critical in exporting NPS pollutants. Therefore, in the selection of one or several BMP's to implement, the ability of BMP's to achieve the water quality goal should be considered. Also, the economic feasibility of implementing such BMP's too should be considered. There are several accepted BMP's that are used frequently in controlling soil erosion and reducing nutrient loadings at receiving waters. In this study, two BMP alternatives that are meant for reducing the sediment erosion within the watershed and that had a reasonable chance of being implemented were considered.

4.5.2 Vegetative filter strips:

Vegetative filter strips (VFS) are permanent grass borders of dense, tall, stiff grass on field boundaries or along stream segments that help in reducing soil input into streams. Filter/buffer strips act as porous dams to temporarily pond surface runoff and allow slowing down and reducing surface runoff from fields. Ponding allows sediments to settle and buffers traps the soil particles from surface runoff and gradually release water to down slope. The effectiveness of buffer strips is dependent on the buffer width, slope of the land, type of vegetation and most importantly on the particle size (Dosskey, 2001).

The AnnAGNPS model does not have a riparian buffer or filter strip component to evaluate the effectiveness of this BMP. Work is under way to develop this capability (Bingner et al., 2005). Though the model cannot model VFS BMP, it can account for the changes in erosion when a cropland is converted to a permanent grass border. In this simulation, streamside cells were converted to buffer strips to evaluate the effectiveness of VFS. Parameters such as, curve number, RUSLE C-factor, overland flow Manning's coefficient and surface condition constant are assigned to represent VFS in streamside cells.

4.5.3 Tile drainage:

Drainage is an important conservation practice. A properly designed drainage system should remove excess water from agricultural fields. Tile drains reduce surface runoff as well as increase the amount of water available for plants by allowing more water to soak into the soil. In this simulation, tile drain BMP was modeled by considering all the cropland in Al-Kfair watershed as tile drained.

CHAPTER 5

RESULTS AND DISCUSSIONS

5.1 OVERVIEW

AnnAGNPS simulation is performed to predict runoff volume, sediment and nutrient loadings at the outlet of Al-Kfair watershed, over the ten years period from 2003 to 2012. In this watershed, there are no measured values for surface runoff, sediment yield and nutrient loadings and hence no calibration and validation of the model is performed. To increase the confidence in model predictions and to improve the understanding of the model behavior, a sensitivity analysis is performed. Results of the sensitivity analysis are presented in section 5.2. Results for runoff volume, sediment and nutrient loadings at the outlet of Al-Kfair watershed are presented in section 5.3.

5.2 SENSITIVITY ANALYSIS RESULTS

5.2.1 Effect of cell discretization

In order to observe the effect of cell sizes on the model output results, sensitivity analysis is performed on seven different cell discretizations. The watershed is divided into 8 cells to 65 cells having average cell sizes ranging from 8 ha. to 30 ha. for different cell discretizations. Simulated watershed runoff, sediment and nutrient loadings for different cell discretizations, obtained at the watershed outlet are summarized in Table 5.1.

Table 5.1: AnnAgnps output result for different Avg. cell size

Input/Output parameter	**Avg. cell size (ha)**			
	8	**15**	**20**	**30**
Number of cells	154	120	90	60
Runoff (mm)	10	11	12	11
Sediment loading at outlet (MCM/yr)	0.014	0.0098	0.00588	0.00294
N conc. at outlet (mg/l)	51.32	53.45	54.01	55.68
P conc. at outlet (mg/l)	9.91	9.86	9.52	9.31

The most significant variation with respect to cell size is the change of sediment loading at the watershed outlet. As the cell size increased from 8 ha. to 30.0 ha., sediment loading reduced approximately by 6 times. Variation of N loading showed an increase as the average cell size increased from 8.0 ha. to 30.0 ha, while P loading did not show such a trend with the change of cell size. It can be concluded that, sediment yield is highly sensitive to the scale of cell discretization and hence estimating the sediment yield without considering the effect of cell discretization, could drastically alter the decisions made concerning non-point source pollution control. Clearly, cell size selection shall not be done arbitrarily and should be based on the scale necessary to capture the spatial variability.

5.2.2 Effects of the soil, crop and other selected parameters

Results of the sensitivity analysis are shown in Table 5.2. Sensitivity index, I in Table 5.2 indicates that the effect of change of the parameter on the simulation results is small and sensitivity index IV indicates that the effect of change of the parameter on the simulation results very high. Indices II and III indicate medium and high sensitivity, respectively.

Table 5.2: Sensitivity indices

No.	Parameter	Runoff	Sediment	N Loading	p Loading
1	K factor	I	III	I	I
2	Wilting point	IV	II	III	IV
3	Field capacity	III	IV	II	III
4	Initial organic phosphorus in soil	I	I	I	II
5	Initial inorganic phosphorus in soil	I	I	I	II
6	Organic matter content	I	I	I	III
7	Organic N	I	I	III	I
8	Initial organic nitrogen in soil	I	I	II	I
9	Initial inorganic nitrogen in soil	I	I	II	I
10	Plant P uptake	I	I	I	I
11	Plant N uptake	I	I	I	I
12	Surface roughness	I	II	I	II
13	Curve number	II	I	II	IV

a. Soil properties

Sensitivity indices indicate that surface runoff is most sensitive to change of wilting point. Wilting point represents the fraction of water volume at wilting point to the soil volume in the soil layer. As the wilting point is increased, less moisture is required by the soil layer to reach to the field capacity resulting more runoff. The same phenomena occur when the field capacity is lowered, resulting in a higher runoff. An increase in field capacity and decrease in wilting point resulted in a lower runoff since more moisture is needed to be absorbed by the soil layer before producing any runoff.

It can be predict that the sediment yield is highly sensitive to the K factor. K factor represents the ease of soil particle to detach from the soil surface and the transportability of the sediment. Wilting point and field capacity showed a moderate effect on the soil erosion. Increase in wilting point and decrease in field capacity has an effect on soil moisture which in turn increase the runoff and hence the sediment detachability and transportability. Other parameters tested showed low to negligible effect on the surface runoff and sediment yield.

Soils with high organic matter content, tends to reduce overland flow by improved non-capillary porosity thus reducing transport of nutrient loading. Soil moisture has a substantial influence on the infiltration capacity of the soil and the surface runoff, thus on the soil erosion. When the soil profile is at or near field capacity, soil moisture is maximum and infiltration capacity is less, thus more runoff and more soil erosion occurs. When the soil profile is near to the wilting point, then the soil moisture is minimum and thus less or no runoff and erosion are expected.

In conclusion, soil related parameters such as; field capacity, wilting point, and K factor are to be selected with a greater accuracy.

b. Crop related parameters

The crop related properties have little to no effect on runoff and sediment yield. P loadings are found to be highly sensitive to initial amount of organic and inorganic N in the soil. As organic N amount was reduced P loading increased. The same pattern was observed, as inorganic N amount was increased by loadings increased.

c. Surface roughness and CN

Effect of change in surface roughness and CN on runoff, sediment yield, N and P loading are noted from the simulations, as surface roughness (SR) is increased by 10% from its base value, variation showed a little decrease in runoff, sediment yield, N and P loadings. As the change is further increased to 20%, as, the effect was considerably large on sediment yield and P loadings. A rough surface with depressions acts as barriers and trap water and sediment, thus reducing detachment by surface runoff and sediment transport. Runoff CN's are varied by 50% and 100% of their base values within the valid range as described in section 4.4. Increase in CN, showed a moderate, about 10%, increase in runoff. As indicated in Table 4.6, increase in CN values mostly occurred in the soil group, which has low runoff potential and a high infiltration potential. The effect of change in CN on sediment yield is minimal. Though the effect of change in CN on P loading is moderate, it showed a high sensitivity on N loading. Changing CN by 50% and by 100% within the range showed an increase in N loading by 25% and 60%, respectively.

5.3 RUNOFF VOLUME, SEDIMENT AND NUTRIENT LOADINGS

The AnnAGNPS simulation is performed to predict runoff volume, sediment, total N and total P loadings at the outlet of Al-Kfair watershed. Rainfall and climate variables are obtained from the closest weather station with complete data set (umm el-Jemal). There are no flow data available to be compared with predicted simulated loadings.

5.3.1 Water loading - Runoff volume

The average annual precipitation for the simulation period is 331 mm and average annual water loading at the outlet is 500,000 m^3. This result in an average annual water yield of 10 mm. Average annual precipitation and runoff are presented in Table 5.3. The average annual runoff is approximately 3% of the total precipitation for the simulation period.

Table 5.3: Average annual precipitation and runoff

Year	Precipitation (mm)	Runoff (mm)	% Runoff
2003	331	10	3.02%
2004	615.04	20	3.25%
2005	409.82	13	3.17%
2006	497.86	16	3.21%
2007	268.77	9	3.35%
2008	241.18	7	2.90%
2009	311.55	9.5	3.05%
2010	672.08	22	3.27%
2011	365.18	12	3.29%

5.3.2 Sediment loading

The total annual sediment load at the watershed outlet is obtained as 119 tons. This results in an annual sediment yield of 0.0238 tons/ha. This sediment load is mainly contributed by sheet and rill erosion and gully erosion. Table 5.4 presents simulated average annual sediment loads at watershed outlet.

Table 5.4: Average annual sediment loading at the watershed outlet

Year	Runoff (mm)	Sediment load (tons)
2003	10	136.97
2004	20	166.67
2005	13	108.33
2006	16	135.93
2007	9	123.27
2008	7	59.40
2009	9.5	163.91
2010	22	99.69
2011	12	78.81

. The highest sediment concentration of nearly 100 mg/1 occurs during the period of low flow following a heavy precipitation in 2004. Average sediment concentration for the simulation period is approximately 25 mg/1.

5.3.3 Nutrient Loading

Simulated average annual total N and P loadings at the watershed outlet are 1228 kg and 240 kg respectively. These numbers result in N and P yield at the watershed outlet as 0.246 kg/ha and 0.048 kg/ha, respectively. These values are somewhat less than the anticipated loading rates. Since the results obtained are based on the existing data and knowledge, there has been no effort made to calibrate the model and hence these results are indicative only. Simulated annual total N and P loadings at watershed outlet are presented in Table 5.5.

Table 5.5: Annual total N and P loading

Year	Nitrogen load (kg)	Phosphorous load (kg)
2003	148.5507	87.64665
2004	679.8757	78.81008
2005	1266.139	94.25935
2006	2393.148	410.2787
2007	1079.35	111.5203
2008	924.2541	88.47737
2009	136.8203	193.5463
2010	2500.617	624.502
2011	470.1493	343.2836
2012	2686.583	370.1737

5.4 AnnAGNPS Model Limitations

The model limitation were summarized as follow:

- ✓ All runoff and associated sediment, nutrient, and pesticide loads for a single day are routed to the watershed outlet before the next day simulation begins (regardless of how many days this may actually take). This means that runoff, sediments, nutrients and pesticides that simulated in a day actually take more than one day to reach the outlet.
- ✓ There is no tracking of nutrients and pesticides attached to sediment deposited in stream reaches from one day to the next.
- ✓ Point sources are limited to constant loading rates (water and nutrients) for entire simulation period. This means that a constant effluent from point-source will be used for the whole simulation period.
- ✓ There is no allowance for spatially variable rainfall.

CHAPTER 6

CONCLUSIONS AND RECOMMENDATIONS

6.1 CONCLUSIONS

This modelling study was conducted to investigate the adaptability of AnnAGNPS model in Al-Kfair watershed, to estimate sediment and nutrient loadings in an effort to prioritize the subwatersheds for treatment/management. The modeled runoff, though within the acceptable limit, is slightly under-predicted compared to the runoff volumes of nearby watersheds. In conclusion, it is evident from this study that, the AnnAGNPS model can be adopted in simulating surface runoff, sediment and nutrient loadings in Al-Kfair watershed that has mostly agricultural land use and can be used in pripritizing watershed management activities.

6.2 RECOMMENDATIONS

From the simulation results it is recommended that at least one experimental, continuously monitoring station be set up to monitor the stream flow, sediment and nutrient concentrations within the watershed. The quantities of water, sediment and nutrient obtained from this simulation may be considered as qualitative indicators. These quantities do indicate the relative quantities of the sediment and nutrient loadings from different cells (pockets of lands) within the watershed. These relative quantities or percentages may be used in prioritizing watershed management activities for soil and water conservation.

REFERENCES

Ambrosio, J. D., T. Lawrence and L.C.Brown, 2001. A basic primer on non-point source pollution and impervious surface. Ohio State University extension fact sheet, Accessed on February 20, 2006 and available at http://www.ohioline.osu.edu/aex-fact/0441.html.

Arnold, J. G, P. M. Allen., and G Bernhardt. 1993. A comprehensive surface-groundwater flow model. Journal of Hydrology, 142: 47-69.

Baginska, B., W. Milne-Home and P. S. Cornish, 2003. Modeling nutrient transport in Currency Creek, NSW with AnnAGNPS and PEST. Environmental modeling and software 18: 801-808.

Bhuyan, S. J., J. K. Koelliker, L. J. Marzen and J. A. Harrington, 2003. An integrated approach for water quality assessment of a Kansas watershed. Environmental modeling and software, 18: 473-484.

Bingner, R. L, J. Garbrecht, J. G. Arnold, and R. Srinivasan. 1997. Effect of watershed subdivision on simulation runoff and fine sediment yield. Transactions of the ASAE40(5): 1329-1335.1997.

Bingner, R. L., and F. D. Theurer. 2003. Topographic factors for RUSLE in the continuous simulation, Watershed model for predicting agricultural, non-point source pollutants (AnnAGNPS), American Society of Agricultural and Biological Engineers, Accessed on Mar 16,2006 and available at http://www.asabe.org

Bingner, R. L., and F. D. Theurer. 2005. AnnAGNPS Technical Processes. National Sedimentation Laboratory, US Department of Agriculture, Oxford, MS.

Bingner, R.L., K. Czajkowski, M. Palmer and J. Coss. 2005. Upper Auglaize watershed modeling project - Final report. Toledo harbor AGNPS project team for US Army Corps of Engineers.

Booty W., D. Lam, G. Bowen, O. Resler and L. Leon. 2005. Modeling changes in stream water quality due to climate change in a Southern Ontario watershed, Canadian Water Resources Journal, 30(3): 211-226.

Borah, D. K. and M. Bera. 2003. Watershed scale hydrologic and non-point source pollution models: Review of mathematical bases, Transactions of American Society of Agricultural Engineers, Vol. 46(6): 1553-1566.

Chaubey, I., C.T. Haan, S. Grunwald and J.M. Salisbury. 1999. Uncertainty in the model parameters due to spatial variability of rainfall, Journal of Hydrology. 220 (1999): 48-61.

Das, S., R. P. Rudra, P. K. Goel, B. Gharabaghi and N. Gupta. 2006. Evaluation of AnnAGNPS in cold and temperate regions, Water science & technology. 53(2): 263 - 270.

Dillaha, T. A., M. L. Wolfe, A. Shrimohammadi, and F.W Byne. 2001. ANSWERS-2000, Environmental and Water Resources Institute, Washington, DC.

Diskin, M.H., (1970). Research approach to watershed modeling, definition of terms, ARS and SCS watershed modeling workshop, Tuscon, Arizona.

Donigian, A.S. and W.C. Huber, 1991. Modeling of non-point source water quality in urban and non-urban areas, Environmental Research Laboratory, US Environmental Protection Agency.

Dosskey, M. G., (2001). Toward quantifying water pollution abatement in response to installing buffers on crop land, Environmental Management. 28 (5) : 577-598.

Fisher, P., R. J. Abrahart and W. Herbinger. 1997. The sensitivity of two distributed non-point source pollution models to the spatial arrangement of the landscape, Hydrological Processes, Vol. 11: 241-252. John Wiley & Sons, Ltd.

Garbrecht, J., and L.W. Martz. 1999. TOPAGNPS, An automated digital landscape analysis tool For topographic evaluation, drainage Identification, watershed segmentation, and subcatchment parameterization for AGNPS 2001 watershed modeling technology, USD A-ARS, Grazinglands Research Laboratory.

Grunwald, S., and L. D. Norton, 1999. An AGNPS-based runoff and sediment yield model for two small watersheds in Germany. Transactions of the ASAE, 42(6): 1723-1731.

Kausman, A., and B. Mitchell. 1997. Soil erosion and sediment yield in forest and agroforestry areas in West Java, Indonesia. Soil and water conservation. 52(4): 376-380.

Leeds, R., L. C. Brown, M. R. Sulc and L. VanLieshout. 1994. Vegetative Filter Strips: Application, Installation and Maintenance, AEX-467-94, Ohio State University Extension Fact Sheet, Accessed on Mar 25, 2006 and available at http://ohioline.osu.edu/aex-fact/0467.html.

Lenhart, T., K. Eckhardt, N. Fohrer and H.G Frede. 2002. Comparison of two different approaches of sensitivity analysis, Physics and Chemistry of the Earth 27: 645654.

Leon, L.F., W. G Booty, G. S. Bowen, and D. C. L. Lam, 2003. Validation of an agricultural non-point source model in a watershed in southern Ontario, Agricultural Water Management 65: 59 - 75.

Lenzi, M. A., and M. D. Luzio, 1997. Suface runoff, soil erosion and water quality modeling in the Alpone watershed using AGNPS integrated with a Geographic Information System. European Journal of Agronomy 6:1-14.

Ma, Y, J. Bartholic, J. Asher, Y. Shi, and D. Ouyang. 2002. NPS Assessment Model: An example of AGNPS application for watershed erosion and phosphorus sedimentation, Journal of Spatial Hydrology, 1(1) 2002.

Martz, L. W, and J. Garbrecht. 1993. Automated extraction of drainage network and watershed data from digital elevation models. Water Resources Bulletin 29 (6): 901-908.

Ministry of the Environment, 2004. White paper on watershed-based source protection planning, Accessed on April 04, 2006 and available at http ://www.ene.gov.on.ca/programs.

Mitchell, J. K., B. A. Engel, R. Sirinivasan, and S. S. Y. Wang. 1993. Validation of AGNPS for small watersheds using integrated AGNPS/GIS system. Water Resources Bulletin, 29(5): 833-842.

Montgomery, J. A., A. J. Busacca, B. E. Frazier, and D. K. McCool. 1997. Evaluating soil movement using C-137 and the Revised Universal Soil Loss Equation. Soil Science Society of America Journal, vol. 61(2): 571-580.

Moore, I.D., 1992. Length-Slope Factors for the Revised Universal Soil Loss Equation, Journal of Soil and Water Conservation, Vol. 47 : 423-428.

Neitsch, S. L., J. G Arnold, J. R. Kiniry, J. R. Williams and K. W. King. 2002. Soil and Water Assessment Tool - Theoretical documentation - Version 2000. Texas Water Resources Institute 2002.

Newham, L. T. H., J. P. Norton, I. P. Prossser, B.F.W. Croke and A. J. Jakeman. 2003. Sensitivity analysis for assessing the behaviour of a landscape-based sediment source and transport model, Envrionmental modeling and software 18(2003) 741751.

Novotny, V, 2003. Water Quality - Diffuse pollution and watershed management, John Wiley and Sons, Inc., New York, NY.

Osterkamp, W.R., P. Heilman, and L.J. Lane, 1998. Economic considerations of a continental sediment- monitoring program. International Journal of Sediment Research, Vol. 13-4 : 12 - 24.

Parsons, J. E., D. L. Thomas, R. L. Huffman, 2001. Agricultural non-point source water quality models: Their use and application, Southern Cooperative Series Bulletin # 398, ISBN: 1-58161-398-9.

Pekarova, P., A. Konicek and P. Miklanek, (1999). Testing of AGNPS model application in Slovak microbasins, Physics, Chemistry and Earth Bulletin, Volume 24(4): 303-305.

Perrone, J. and C. A. Madramootoo, 1997. Use of AGNPS for watershed modeling in Quebec. Transaction of ASCE 40 (5): 1349 - 1354.

Qiu, Z., T. Prato and C. Fulcher, 1997. The Impacts of Data Resolution on Simulation of Water Quality Using the Agricultural Nonpoint Source Pollution Model and a Geographic Information System, Accessed on May 18, 2006 and available at http.7/ www.wca-infonet.org/106015082_WE_WQ_KO51 .pdf

Renard, K. G, G. R. Foster, G. A. Weesies, D. K. McCool and D. C. Yoder. 1997. Predicting soil erosion by water: A guide to conservation planning with the Revised Universal Soil Loss Equation, USDA-ARS Agriculture handbook number 703.

Richards, N.R., A. G Caldwell, F. F. Morwick. 1949. Soil Survey of Essex County, Experimental farm service and Ontario Agriculture College.

Saxton K. E., and P. H. Wiley, 2005. The SPAW model for agricultural field and pond hydrologic simulation. USDA-ARS, Department of Biological Systems Engineering, Washington State University, Accessed on May 22, 2006 and available at http://hydrolab.arsusda.gov/SPAW/index.htm.

Simon, A., R. L. Bingner, E. J. Langendoen and C. V. Alonso, 2002. Actual and reference sediment yields for the James Creek watershed - Mississippi, USDA - ARS, National Sedimentation Laboratory.

Smith, R. L. 2002. Phase 1 watershed assessment final report, Mina Lake / Snake Creek Brown, Edmunds And Mcpherson Counties, South Dakota, South Dakota Water Resource Assistance Program.

Srinivasan, R. and J. G. Arnold. 1994. Integration of a basin scale water quality model with GIS, Water Resources Bulletin, 30 (3): 453-462.

Stone R. P. and D. Hillborn. 2002. Universal Soil Loss Equation, Ontario Ministry of Agriculture and Food, Ontario, Canada. Accessed on March 25,2006 and available at http.V/www.omafra.gov.on.ca/english/engineer/facts/00-001 .htm.

Sugiharto, T., T. H. Mcintosh, R. C. Uhrig, and J. J. Lardinois. 1994. Modeling alternatives to reduce dairy farm and watershed non-point source pollution. Journal of Environment Quality. 23:18-24.

Theurer, F. D., and R. L. Bingner. 2005. Watershed-scale pollutant loading model-AnnAGNPS v3.5, Fact sheet, USDA-ARS-Naturai Resources Conservation Service-2005.

USDA-ARS. 2005. National Sedimentation Laboratory, AnnAGNPS Arcview Interface Documentation. Accessed on Mar 03, 2006 and available at http://www.ars.usda.gov/Research/docs.

U.S. Environmental Protection Agency (USEPA), 2002, National Water Quality Inventory: 2000 Report to Congress, Accessed on June 12, 2006 and available at http://www.epa.gov/305b/2000report/factsheet.

Vennix, S., and W. Northcott, 2004. Prioritizing vegetative buffer strips placement in an agricultural watershed. Journal of Spatial Hydrology, 4 (1): 61-79.

Vieux, B.E., and S. Needam. 1993. Non-point pollution model sensitivity to grid-cell size, Journal of Water Resources Planning and Management, 119 (2) 1993.

Wischmeier, W. H. and D. D. Smith. 1978. Predicting rainfall erosion losses: A guide to conservation planning. Agricultural Handbook. USDA - ARS.

Woolhiser, D. A., and D. L. Brakensiek. 1982, Hydrologic system synthesis, Hydrologic modeling of small watersheds, American Society of Agricultural Engineers, St. Joseph, Montana.

Young, R.A., C. A. Onstad, D. D. Bosch and W.P. Anderson. 1989. AGNPS: A non-point source pollution model for evaluating agricultural watersheds, Journal of Soil and Water Conservation. 44(2): 168-173.

Yuan, Y., R. L. Bingner, R.A. Rebich. 2003. Evaluation of AnnAGNPS nitrogen loading in an agricultural watershed, Journal of American water resources association, Accessed on July 21,2006 and available at http:// www.findarticles.com/articles/mi_qa4038/is_200304/ai_n9206397.

Zhen, X., 2002. Development of Best Management Practice (BMP) placement strategies at the watershed scale, Ph. D. dissertation, University of Virginia, US. Retrieved July 12, 2006 from ProQuest Digital Dissertation database. Article number 3057500.

PART II: Groundwater Model

MODELING THE EFFECTS OF SEPTIC TANKS ON GROUNDWATER QUALITY

1.0 Introduction

There are nearly half a million cesspools (septic tanks) servicing over a million people in Jordan. Septic system effluent discharged to the unsaturated zone reaches ground water, where it impacts water quality. Chemical concentrations in effluent discharge vary with the type of use. For a typical household using the septic tanks for discharge of drinking, shower, toilet, and laundry water, concentrations of nitrogen, sodium, potassium, bicarbonate, chloride, phosphorus, and carbon are greater in septic effluent than in groundwater.

Septic waste discharged to coarse-textured soils proceeds vertically through the unsaturated zone and into ground water. Once it reached the ground water, a septic plume develops and moves with ground water flow. Approximate times for septic effluent to pass through the unsaturated zone to ground water range from a few hours to few years. This time depend on the volume of effluent, soil properties of the unsaturated zone and the distance to groundwater.

Nitrate is the primary chemical of concern in most septic plumes. Nearly all nitrogen passing through the septic tanks converts to nitrate in the aerobic soil zone and eventually leaches to groundwater. Nitrates are conservative in shallow groundwater because oxygen is present and total organic carbon concentrations are too low to sustain intensive microbial activity.

Predicting nitrate concentrations in an unsewered area consists of quantifying the contribution of each system. It is difficult, however, to predict the nitrate concentration in specific locations because plumes may mix and the fate of individual plumes is unknown. In addition, seasonal changes in groundwater flow may occur because of different inputs from the septic system, effects of surface water bodies, and local pumping from wells.

The study was conducted in Al-Kfair Town, located in Amman Zarqa Basin. The Al-Kfair area has a population of approximately 3500 and is undergoing rapid expansion with unsewered developments. The study area is located in an area of SCL soils underlain by an aquitard which is underlined by sandstone aquifer.

Based on a previous modeling study for Al-Kfair area using AnnAGNPS software which proved that there were no significant effects of agricultural practices on groundwater quality based on the usage of fertilizers, Pesticides and manure in the study area. Since most of the farmers did not use any of them in their agricultural practices. So, in this report we will model only the effects of septic systems on groundwater quality using HYDRUS-2D software.

2. Contaminant Transport and Model Description

2.1 Description of wastewater/aquifer interaction

In septic tank effluent, the total nitrogen concentrations consist of about 75% ammonium and 25% organic nitrogen. In general, nitrogen can undergo several transformations including: adsorption of ammonium to negatively charged particles in the soil (clays, however, do not typically remove nitrogen from the system); volatilization of ammonia in alkaline soils; nitrification and subsequent movement of nitrate towards groundwater; biological uptake of both ammonium and nitrate and denitrification (following nitrification if the environmental conditions are appropriate). A schematic illustration of the forms and transformations of nitrogen in effluent is shown in Figure 2.1.

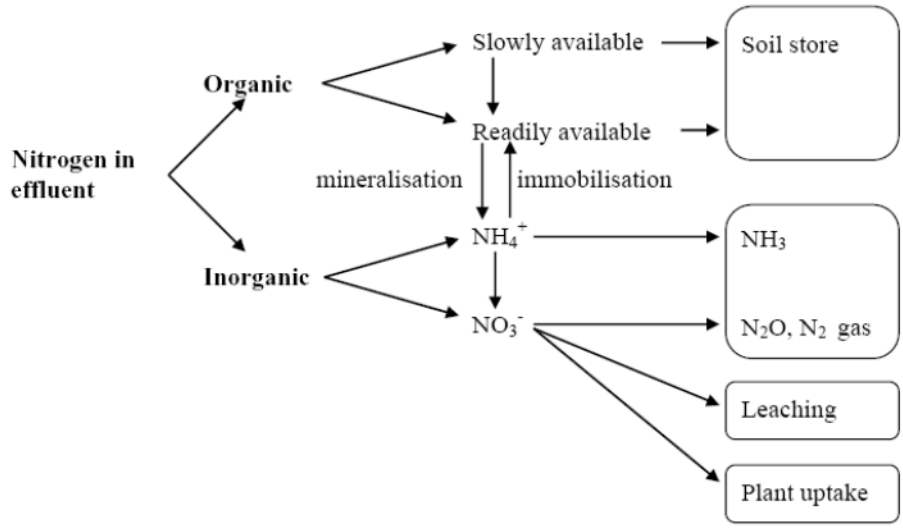

Figure 2.1: Forms of nitrogen in effluent.

Of the biochemical processes which result in the transformation of nitrogen species, most work has been done in relation to nitrification which occurs in the unsaturated zone (Whelan and Barrow, 1984). Particularly in sandy soils, aerobic bacteria in the biomat and upper vadose zone rapidly convert the ammonium almost entirely to nitrite and then nitrate. Nitrification is a bacterially mediated acid-forming process which occurs in two stages summarized in the following equation:

$$\mathbf{NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+} \qquad \mathbf{(2.1)}$$

Effluent from the septic tank also contains a rich supply of labile carbon and, during effluent oxidation in the vadose zone, carbon biodegradation occurs coincidentally according to the following equation (Robertson and Cherry,1992):

$$\mathbf{CH_2O + O_2 \rightarrow CO_2 + H_2O} \qquad \mathbf{(2.2)}$$

2.2 Chemistry of Nitrogen

Nitrogen, which in its pure gaseous form comprises 78% by volume of the earth's atmosphere, can exist in nine various forms in the environment due to seven possible oxidation states as shown in table 2.1 (WEF, 1998).

Table 2.1 : Environmental Nitrogen forms

Nitrogen Compound	Formula	Oxidation State
Organic nitrogen	Organic-N	-3
Ammonia	NH_3	-3
Ammonium ion	NH_4^+	-3
Nitrogen gas	N_2	0
Nitrous oxide	N_2O	+1
Nitric oxide	NO	+2
Nitrite ion	NO_2^-	+3
Nitrogen dioxide	NO_2	+4
Nitrate ion	NO_3^-	+5

The principal forms of nitrogen of concern in onsite wastewater treatment and soil-groundwater interactions are Organic-N, ammonia/ammonium ion (NH_3/ NH_4^+), nitrogen gas (N_2), nitrite (NO_2^-), and nitrate (NO_3^-) (Rittman & McCarty, 2001; Sawyer et al., 1994; US EPA, 1993)

2.2.1 Nitrogen Cycle in Soil-Groundwater Systems

As shown in Figure (2-2), transformation of the principal nitrogen compounds can occur through several key mechanisms in the environment: fixation, ammonification, synthesis, nitrification, and denitrification (US EPA, 1993).

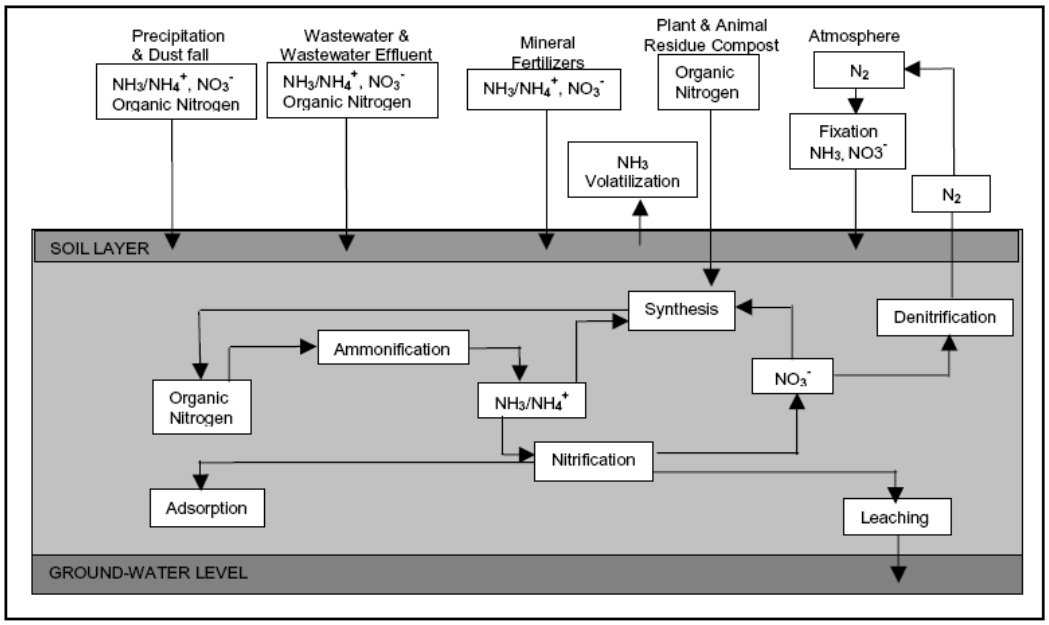

Figure 2.3 :The nitrogen cycle in soil and groundwater (after U.S EPA,1993).

A) Nitrogen Fixation.

Nitrogen fixation is the conversion of nitrogen gas into nitrogen compounds that can be assimilated by plants. Biological fixation is the most common, but fixation can also occur by lightning and through industrial processes:

Biological: Nitrogen gas → Organic Nitrogen

Lightning: Nitrogen gas → Nitrate

Industrial: Nitrogen gas → Nitrate and Ammonia/Ammonium ion

B) **Ammonification.**

Ammonification is the biochemical degradation of organic nitrogen into ammonia or ammonium ion by bacteria that use organic carbon in building cell tissue. These are called heterotrophic bacteria. These bacteria can transform the nitrogen either in the presence of oxygen

(aerobic conditions) or without oxygen (anaerobic conditions). Within an onsite wastewater system, ammonification of organic nitrogen in the human waste stream occurs primarily within the anaerobic conditions of the septic tank. Some of the organic nitrogen, however, is not degraded and becomes part of the humus in the receiving soils

C) Synthesis.

Synthesis is the biochemical mechanism in which ammonium ion or nitrate is converted into plant protein (organic nitrogen):

Nitrogen fixation is a unique form of synthesis that can only be performed by nitrogen fixing bacteria and algae:

D) Nitrification.

Nitrification is the biological oxidation of ammonium ion to nitrate through a two-step process by two species of bacteria called *Nitrosomonas* and *Nitrobacter*. In the first step, ammonium ions are converted to nitrite by *Nitrosomonas sp*. The second step involves the conversion of nitrite to nitrate by *Nitrobacter sp.* Both these species are considered autotrophic bacteria because they use carbon dioxide (CO_2) as the source of carbon for building cell tissue]The two-step reaction is usually very rapid. Because of this it is rare to find nitrite levels higher than 1.0 mg/L in water. The nitrate formed by nitrification is, in the nitrogen cycle, used by plants as a nitrogen source (synthesis) or reduced to N_2 gas through the process of denitrification. Nitrate can, however, contaminate groundwater if it is not used for synthesis or reduced through denitrification.

E) **Denitrification.**

Nitrate can be transformed to nitrogen gas under conditions where dissolved oxygen is absent (called anoxic conditions) by heterotrophic bacteria (those that use organic carbon for building cell tissue).

In order for denitrification to occur, it must happen without dissolved oxygen present. If dissolved oxygen is present, the organisms will use it rather than the nitrate bound oxygen in their metabolism. In this latter case, nitrogen in the form of nitrates would remain to pass into and through the soil, eventually ending up in groundwater

2.3 Governing equations

2.3.1 Flow equation

The governing equation for variably saturated water flow is described by Richards's equation and can be written as

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x}\left[K(\theta)\frac{\partial \psi}{\partial x}\right] + \frac{\partial}{\partial z}\left[K(\theta)\frac{\partial \psi}{\partial z}\right] + \frac{\partial K(\theta)}{\partial z} \qquad (2\text{-}3)$$

where θ is the water content, t is time, $K(\theta)$ is the unsaturated hydraulic conductivity, ψ is the pressure head, x is the horizontal coordinate, and z is the vertical coordinate.

The dependent variables, θ and ψ in Eq. (2-3), are interrelated through the two fundamental relationships of the soil properties: the hydraulic conductivity function $K(\theta)$ and the soil moisture retention curve $\psi(\theta)$. Eq. (2-3) is general in the sense that it is equally valid for both homogeneous and heterogeneous soil, and there are no constraints on the hydraulic functions. By utilizing a defined relationship between θ and ψ, known as the soil water capacity soil, moisture retention curve $\psi(\theta)$ can be eliminated. The soil water capacity can be written as:

$$C = \frac{\partial \theta}{\partial \psi} \qquad (2\text{-}4)$$

and can be obtained as the slope on the soil moisture retention curve. The tension based version of the governing equation becomes:

$$C\frac{\partial \psi}{\partial t} = \frac{\partial}{\partial x}\left[K(\theta)\frac{\partial \psi}{\partial x}\right] + \frac{\partial}{\partial z}\left[K(\theta)\frac{\partial \psi}{\partial z}\right] + \frac{\partial K(\theta)}{\partial z} \qquad (2\text{-}5)$$

This equation is usually referred to as Richards' Equation. It may still apply when ψ becomes positive in which case the equation degenerates to the Laplace's equation.

An alternative version of the governing Eq. (2-3) can be obtained by introducing the soil water diffusivity:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x}\left[D\frac{\partial \theta}{\partial x}\right] + \frac{\partial}{\partial z}\left[D\frac{\partial \theta}{\partial z}\right] + \frac{\partial K(\theta)}{\partial z} \qquad (2\text{-}6)$$

This equation is usually termed the diffusivity form of the flow equation. To apply the diffusivity form, the medium must be uniform in order to provide a continuous variation in θ. Further, it requires that C is different from zero, which is not fulfilled under saturated conditions, where ψ varies and θ remains constant provided the media incompressible. Eq. (2-3) is widely used in the numerical modeling of the variably saturated media.

2.3.1.1 Hydraulic conductivity function

As previously stated, the governing equation for the unsaturated flow requires information about two hydraulic functions: the hydraulic conductivity function $K(\theta)$ and the soil moisture retention curve $\psi(\theta)$. The hydraulic conductivity decreases strongly as the moisture content θ decreases from saturation as shown in Figure 2-3. This is not surprising since the total cross-sectional area for the flow decreases as the pores become filled with air. In addition, when smaller part of the pore system is available to carry the flow, the path of flow becomes more tortuous. Other reasons may explain the strong decrease, e.g. an increase of the viscosity of the water, when the short-range adsorptive forces become dominant in relation to the capillary forces.

The experimental procedure for measuring the $K(\theta)$ function is rather difficult and not very reliable (Tindall and Kunkel, 1999). Alternatively procedures have been suggested to derive the function from more easily measurable characterizing properties of the soil or simply to rely on empirical relationships (Jury *et al*., 1991).

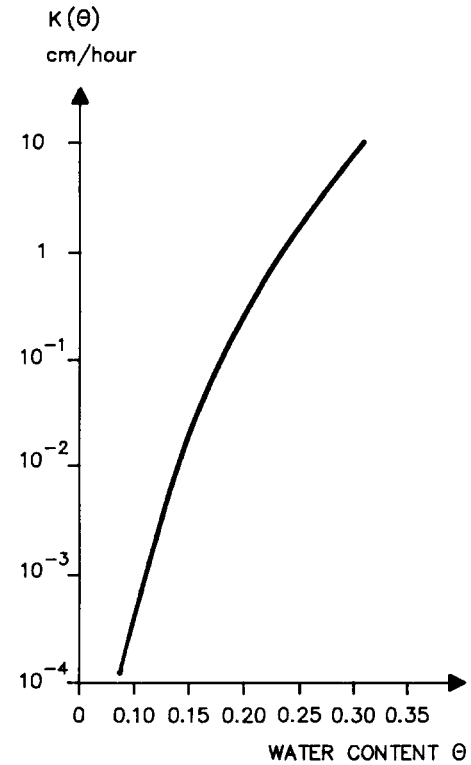

Figure 2-3. Example of the shape of the hydraulic conductivity function.

Since no universal relations are available for unsaturated hydraulic conductivity versus soil suction or water content, several empirical relations have been proposed. Tindall and Kunkel (1999) summarized these relationships as follows:

$$K(\psi) = \frac{a}{\psi}$$ *(Baver, Gardner, and Gardner 1972)*

$$K(\psi) = a\left(b + \psi^{n}\right)^{-1}$$ (Childs and Collis-George 1950)

$$K(\psi) = \frac{K_s}{\left[1 + \left(\frac{\psi}{\psi_c}\right)^{n}\right]}$$ (Gardener 1958)

$$K(\psi_m) = \frac{K_s}{b + \psi_m^{n}}$$ (Childs and Collis-Gorge 1950)

$$K(\theta) = a(\theta)^n \qquad \text{(Marshal and Holmes 1979)}$$

$$K(\theta) = K_s \left(\frac{\theta - \theta_r}{\phi - \theta_r} \right)^n \qquad \text{(Brooks and Corey 1966)}$$

$$K(\theta) = K_s \exp(a \psi_m) \qquad \text{(Mualem 1976)}$$

$$K(\theta) = K_s \sqrt{\frac{\theta - \theta_r}{\phi - \theta_r}} \left[1 - \left(1 - \left(\frac{\theta - \theta_r}{\phi - \theta_r} \right)^{1/m} \right)^m \right]^2 \qquad \text{(Van Genuchten 1980)}$$

Where $m=1-1/n$, $K(\theta)$ is the unsaturated hydraulic conductivity, K_s is the saturated hydraulic conductivity for the same medium, *a, b, m* and *n* are empirical constant (for fine texture media n=1-2 and can be up to 4 or more for coarse media), ψ_c is the matrix potential for which $K=0.5(K_s)$ and θ_r is the residual saturation. The relationship introduced by Van Genuchten (1980) is widely used.

2.3.1.2 Soil moisture retention curve

The relationship between the water content θ and pressure head ψ is termed the soil moisture retention curve or the soil moisture characteristics. The function is basically defined by the textural and the structural composition of the soil. Also the organic matter content may have an influence on the relationship. A characteristic feature of the soil moisture retention curve is that ψ decreases fairly rapidly with moisture contents shown in Figure 2-4. Hysteresis effects may appear, and, instead of being a single valued relationship, the θ-ψ relation consists of a family of curves. The history of wetting and drying of the soil will determine the actual curve.

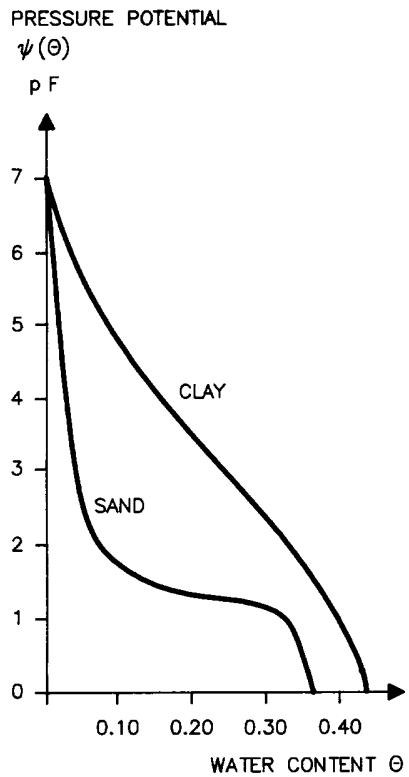

Figure 2-4. Example of soil retention curves.

2.3.2 Solute Transport Equation

The two dimensional advection–dispersion equation for reactive solute transport in variably saturated soil is

$$\frac{\partial \theta C}{\partial t} + \rho \frac{\partial C^*}{\partial t} = \frac{\partial}{\partial x_i}\left[\theta D_{ij} \frac{\partial C}{\partial x_j}\right] - \frac{\partial}{\partial x_i}\left[\theta V_i C\right] \qquad i,j=1,2 \qquad (2\text{-}7)$$

Where C is the concentration of the solute in the fluid phase, C^* is the adsorbed solute concentration, defined as adsorbed solute mass per unit mass of soil, ρ is the bulk density of the soil, V_i is the component of the soil water velocity in direction x_i and D_{ij} is the (i,j)th component of the laboratory –scale dispersion coefficient tensor, given by the constitutive relationship given by Bear (1972)

$$D_{ij} = \alpha_T |V| \delta_{ij} + (\alpha_L - \alpha_T)\frac{V_i V_j}{|V|} + D_d \tau \delta_{ij} \qquad (2\text{-}8)$$

Where α_L and α_T are the longitudinal and transverse laboratory-scale dispersivities, respectively, $|V| = \left(V_x^2 + V_z^2\right)^{1/2}$ is the magnitude of the soil water velocity, D_d is the ionic or molecular diffusion coefficient in free water, τ is the tortuosity factor and δ_{ij} is the kronecker delta function ($\delta_{ij}=0$ *if* $i \neq j$ and $\delta_{ij}=1$ *if* $i=j$). When the equation applied to two-dimensional flow in a vertical cross section, then we take the $x_1=x$ as the horizontal coordinate and $x_2=z$ as the vertical coordinate. The molecular diffusion coefficient in free water is given by the constitutive relationship given by (Weber and Digiano, 1996)

$$D_d = \frac{10^{-9} T}{\mu_v} \left(\frac{\rho_{mo;i}}{W_{g,mo,i}} \right)^{1/3} \tag{2-9}$$

Where T is the temperature (Kelvin), μ_v is the dynamic viscosity (*poise*), $\rho_{mo,i}$ is the density of the molecule (g/cm^3), $W_{g,mo,i}$ is the molecular weight of the molecule (*g/mole*) and D_d is the molecular diffusion in water in(cm^2/s).

2.3.2.1 Sorption

Sorption processes cover a number of geochemical and chemical reactions such as adsorption of solutes to the aquifer material surface by electrostatic forces (called cation exchange). If these processes occur sufficiently fast compared with the water flow velocity, they can be described by an equilibrium sorption isotherm. Different equilibrium sorption isotherms have been identified analyzing results from experiments with different sediment, soil and rock types (Fetter, 1993; Watts, 1998). The most commonly applied isotherms namely the linear, Freundlich and Langmuir equilibrium sorption isotherms.

Sorption processes that do not occur sufficiently fast compared with the water flow velocity have to be described by a kinetic sorption isotherm (Bear, 1972; and Fetter, 1993). Each type of adsorption isotherm has different effect on solute transport in the porous media.

2.3.2.2 Equilibrium sorption isotherms

The linear sorption isotherm is mathematically the simplest isotherm and can be described as a linear relationship between the amount of solute sorbed onto the soil material and the aqueous concentration of the solute:

$$C^* = K_d C \tag{2-10}$$

where C^* is the mass of solute sorbed per dry weight of the solid, C is the concentration of solute in equilibrium with the mass of solute sorbed onto the solid and K_d is known as the distribution coefficient.

The distribution coefficients are often related to the organic matter content of the soil by an experimentally determined parameter (K_{oc} called the distribution coefficient for soil organic carbon), which can be used to calculate the K_d values.

$$K_d = f_{oc} K_{oc} \tag{2-11}$$

Where f_{oc} is the organic carbon content.

The Freundlich sorption isotherm is a more general equilibrium isotherm, which can describe a non-linear relationship between the amount of solute sorbed onto the soil material and the aqueous concentration of the solute:

$$C^* = K_f C^n \tag{2-12}$$

Where K_f and n are constants.

Both the linear and the Freundlich isotherm suffer from the same fundamental problem that there is no upper limit to the amount of solute that can be sorbed. In natural systems one would expect that there is a finite number of sorption sites on the soil material, i.e. there is an upper limit on the amount of matter that can be sorbed.

The Langmuir sorption isotherm takes into account that there is only a limited number of sorption sites on the soil material. When these sites are filled sorption of solutes will no longer occur. The isotherm is often given in the following form:

$$\frac{C}{C^*} = \frac{1}{\alpha\beta} + \frac{C}{\beta} \tag{2-13}$$

or

$$C^* = \frac{C\alpha\beta}{1+\alpha C} \tag{2-14}$$

Where α is a sorption constant related to the binding energy and β is the maximum amount of solute that can be absorbed by the soil material.

2.3.2.3 Kinetic sorption isotherms

All equilibrium models assume that the rate of change in concentration due to sorption is much greater than the change due to any other causes and that the flow rate is low enough that equilibrium can be reached. If this is not the case and equilibrium is not attained, a kinetic model is more appropriate. In a kinetic model the solute transport equation is linked to an appropriate equation to describe the rate that the solute is sorbed onto the solid surface and desorbed from the surface.

The most simple nonequilibrium condition occurs when the rate of sorption is a function of the concentration of the solute remaining in solution and that once sorbed onto the solid, the solute can not be desorbed. This is an irreversible reaction and the processes leads to attenuation of the solute. The irreversible first order kinetic sorption model that described this can be given by the following equation (Fetter, 1993)

$$\frac{\partial C^*}{\partial t} = k_1 C_s$$

(2-15)

Where k_1 a first order decay rate constant.

If the rate of solute sorption is related to the amount that has already been sorbed and the reaction is reversible, then a reversible linear kinetic sorption model can be used. An example of such a model is given by the following equation (Simunek et al., 1999):

$$\frac{\partial C^*}{\partial t} = \gamma (k_a C_s - C^*) \qquad (2\text{-}16)$$

Where γ is a first order rate coefficient and k_a is a constant equivalent to K_d.

2.4 Model Selection

Numerical models provide the most general tool for the quantitative analysis of groundwater application. They are not subject to many of the restrictive assumptions required for analytical solutions. Numerical solution normally involves approximating continuous (defined at every point) partial differential equation with a set of discrete equations in time and space. Thus, the region and the time period of interest are divided in some fashion, resulting in an equation or set of equations for each subregion and time step. These discrete equations are combined to form a system of algebraic equations that must be solved for each time step. Finite difference (FDM) and finite element methods (FEM) are the major numerical techniques used in groundwater application.

The application of the finite element method to groundwater problems is a relatively recent development compared with the finite difference method (Wang and Anderson, 1982). Each method leads to a set of algebraic equations in which the unknowns are the heads at a finite number of nodal points. Figure 2-5 shows the conceptual view of a problem domain as approximated by the Finite element method. The finite difference method is usually implemented with rectangular cells. The finite element method refers to numerical solution whereby a region is divided into subregions called element, whose shapes are determined by a set of points called nodes as shown in Figure 2-6. Faust and Mercer (1980) summarized the important advantages and disadvantages of FEM and FDM as shown Table 2-2.

Table 2-2. Brief summary of important advantages and disadvantages of FDM and FEM

Advantages	***Disadvantages***
FINITE DIFFERENCE METHOD	
Intuitive basis. Easy input data. Efficient matrix techniques. Program changes easy.	Low accuracy for some problems. Regular grids.
FINITE ELEMENT METHOD	
Flexible geometry. High accuracy easily included. Evaluate cross product terms better.	Mathematical basis is advance. Difficult data input. Difficult programming

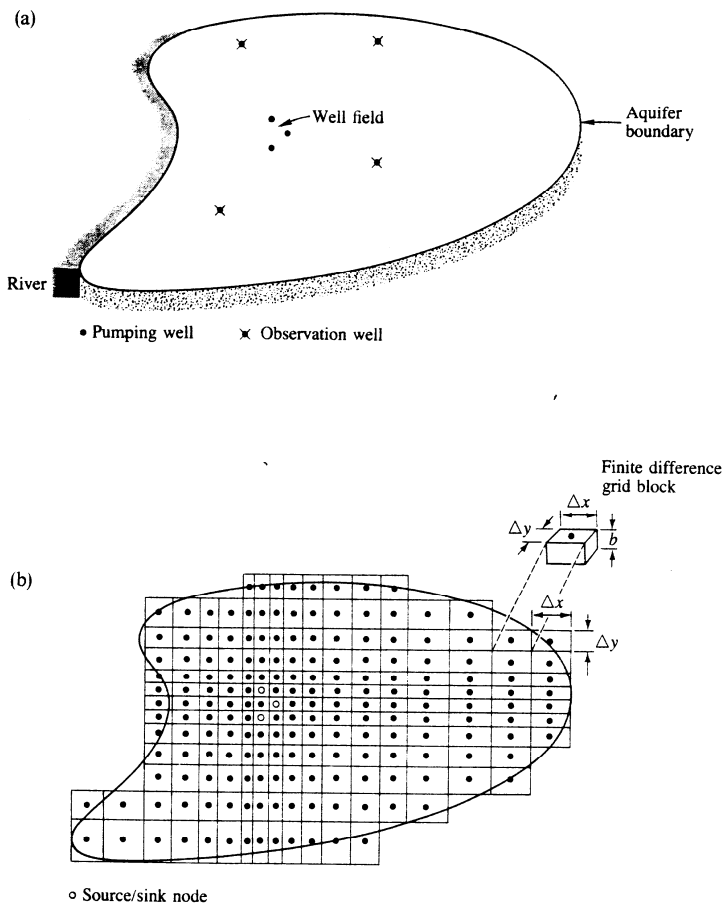

Figure 2-5. Construction of finite difference grid.

(Wang and Anderson, 1982)

(a) Map view of aquifer system

(b) Finite difference grid with block centered nodes.

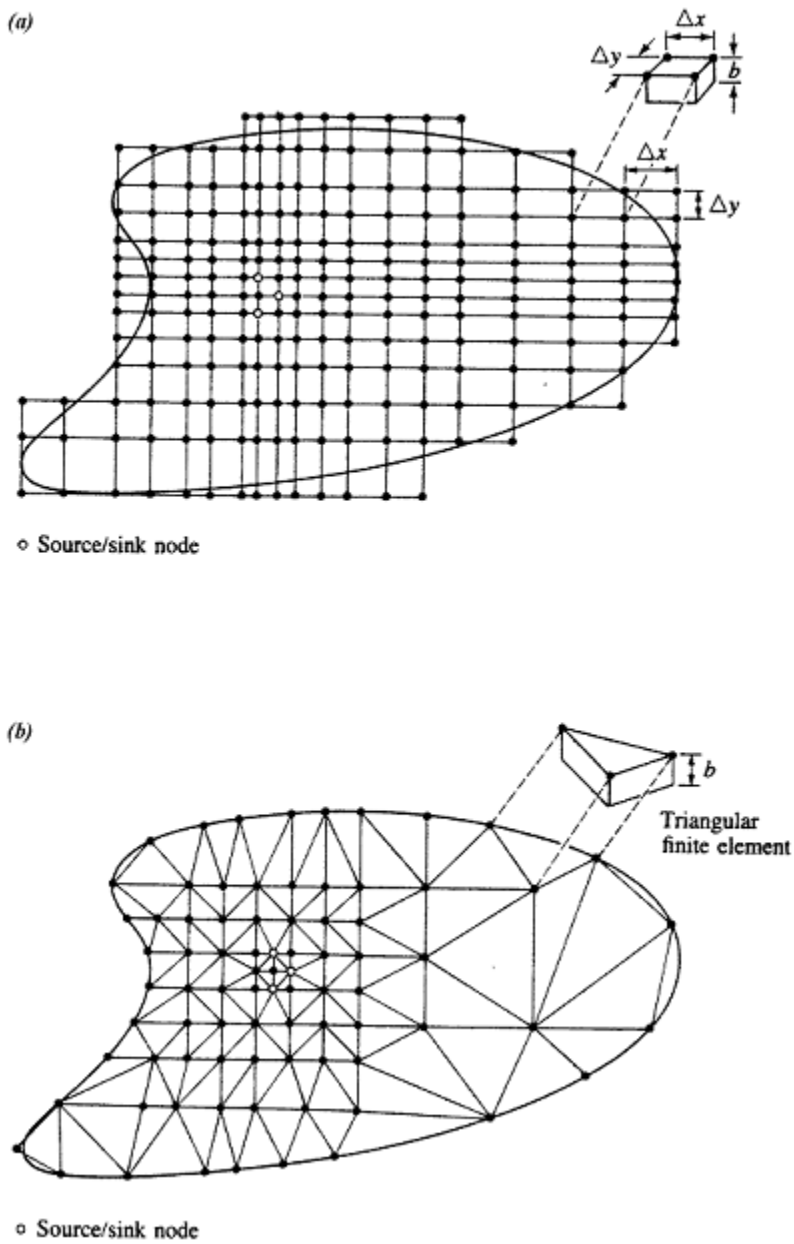

Figure 2-6. Comparison between finite element mesh and finite difference mesh. (Wang and Anderson, 1982).

(a) Finite difference method mesh block.

(b) Finite element method.

In order to simulate the effect of cesspool systems on groundwater quality, it was determined that the code should be able to handle the complex geometry, simulate water and solute transport in variable saturated media, and represent the septic tank flow fluctuating. The HYDRUS-2D software can handle the previous challenging modeling topics and the following section describes it in some details.

2.5 HYDRUS-2D model description

HYDRUS-2D is a Microsoft Windows based modeling environment for analysis of water flow and solute transport in variably saturated porous media developed by (Simunek, et al., 1999). The program numerically solves the Richards' equation for saturated unsaturated water flow and the Fickian-based advection dispersion equations for heat and solute transport The software package includes the two-dimensional finite element model HYDRUS2 for simulating the movement of water, heat, and multiple solutes. The model includes a parameter optimization algorithm for inverse estimation of a variety of soil hydraulic and/or solute transport parameters. The model is supported by an interactive graphics-based interface for data preprocessing, generation of a structured mesh, and graphic presentation of the results. The modeling environment includes a mesh generator for unstructured finite element grids, MESHGEN-2D.

2.5.1 The HYDRUS2 Model

The program numerically solves the Richards' equation for saturated unsaturated water flow and the Fickian-based advection dispersion equations for heat and solute transport. The flow equation incorporates a sink term to account for water uptake by plant roots. The heat transport equation considers conduction as well as convection by flowing water. The solute transport equations consider advective-dispersive transport in the liquid phase, and diffusion in the gaseous phase. The transport equations also include provisions for nonlinear and/or nonequilibrium reactions between the solid and liquid phases, linear equilibrium reactions between the liquid and gaseous phases, zero order production, and two first order degradation reactions: one which is independent of other solutes, and one which provides the coupling between solutes involved in sequential first-order decay reactions. The program may be used to

analyze water and solute movement in unsaturated, partially saturated, or fully saturated porous media.

HYDRUS2 can handle flow regions delineated by irregular boundaries. The flow region itself may be composed of non-uniform soils having an arbitrary degree of local anisotropy. Flow and transport can occur in the vertical plane, the horizontal plane, or in a three-dimensional region exhibiting radial symmetry about the vertical axis. The water flow part of the model can deal with prescribed heads (constant or time-varying) and flux boundaries, as well as boundaries controlled by atmospheric conditions. Soil surface boundary conditions may change during the simulation from prescribed flux to prescribed head type conditions (and vice versa). The code can also handle a seepage face boundary through which water leaves the saturated part of the flow domain, and free drainage boundary conditions.

For solute transport, the code supports both constant and varying prescribed concentrations (Dirichlet or first type) and concentration flux (Cauchy or third type) boundaries. The dispersion tensor includes a term reflecting the effects of molecular diffusion and tortuosity.

The unsaturated soil hydraulic properties are described using van Genuchten (1980), Brooks and Correy (1964) and modified van Genuchten type analytical functions. Modifications were made to improve the description of hydraulic properties near saturation. The HYDRUS2 code incorporates hysteresis by using the empirical model introduced by Scott et al. (1983) and Kool and Parker (1987). This model assumes that drying scanning curves are scaled from the main drying curve, and wetting scanning curves from the main wetting curve. HYDRUS2 also implements a scaling procedure to approximate hydraulic variability in a given soil profile by means of a set of linear scaling transformations which relate the individual soil hydraulic characteristics to those of a reference soil.

The governing equations are solved using a Galerkin type linear finite element method applied to a network of triangular elements. Integration in time is achieved using an implicit (backward) finite difference scheme for both saturated and unsaturated conditions. The resulting equations are solved in an iterative fashion, by linearization and subsequent Gaussian elimination for banded matrices, a conjugate gradient method for symmetric matrices, or the ORTHOMIN method for asymmetric matrices. Additional measures are taken to improve solution efficiency in transient problems, including automatic time step adjustment and checking if the Courant and Peclet numbers do not exceed preset levels. The water content term is evaluated using the mass

conservative method proposed by Celia et al. (1990). To minimize numerical oscillations upstream weighing is included as an option for solving the transport equation.

2.5.2 User Interface

The Microsoft Windows based Graphics User Interface (GUI) manages the input data required to run HYDRUS-2D, as well as grid design and editing, parameter allocation, problem execution, and visualization of results. The program includes a set of controls that allows the user to build a flow and transport model and perform graphical analyses on the fly. Both input and output can be examined using areal or cross-sectional views and line graphs.

2.5.3 Automatic Mesh Generation

Data preprocessing involves specification of the flow region having an arbitrary continuous shape bounded by polylines, arcs and splines, discretization of domain boundaries, and subsequent generation of an unstructured finite element mesh.

3. Approach Methodology

3.1 Numerical Setup

A two-dimensional vertical section designed to represent the groundwater flow field in the vicinity of a Al-Kfair watershed is proposed as the generic test case. Figure 3-1 shows the dimensions and geometry of the numerical model.

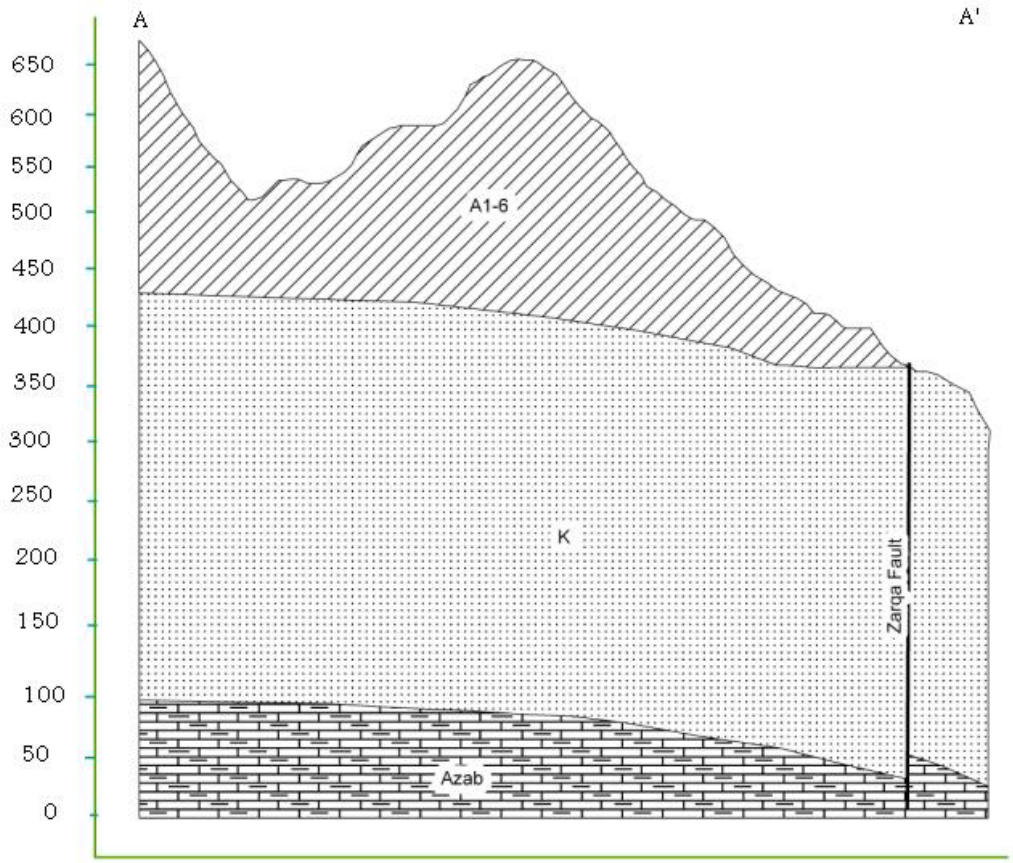

Figure 3-1. Geometry and dimensions of the generic profile.

The porous media is assumed to be homogeneous and isotropic and of uniform porosity.
Based on the geologic and structural features described in the previous reports, two main layers were identified:

Ajlun group (a1/a6) aquitard has low permeability in the study area and the number of well reached the base in this formation are limited. At the eastern part of this formation in the study area the base is 0 m amsl but at the south-west part it reach 100 m amsl.

The Kurnub aquifer represents the lower aquifer system and outcrops only in western and southern parts of study area along the Suweileh anticline. It consists of sandstone, white or varicolored with layers of reddish silt and shale. Because of the presence of clay layers in Kurnub aquifer, there are variations in horizontal and vertical permeability.

Table 3-1, Figure 3-2 and Figure 3-3 show the saturated and unsaturated soil parameters and the soil moisture (θ) and saturated hydraulic conductivity (K_s) parameter obtained from the literature and the soil catalog of HYDRUS-2D V.2.0 (1999) developed by J. Simunek and M. Th. Van Genuchten.

Table 3-1. Soil Parameters.

	Van Genuchten Soil Parameters				
Aquifer layer	θ_r	θ_s	α (cm^{-1})	n	K_s(cm/day)
Ajloun Group A1/6	0.07	0.36	0.005	1.09	0.000006
Kurnub K	0.057	0.41	0.124	2.28	0.004
[2]Parameters obtained from the soil catalog of HYDRUS-2D V.2.0 (1999) developed by J. Simunek and M. Th. Van Genuchten.					

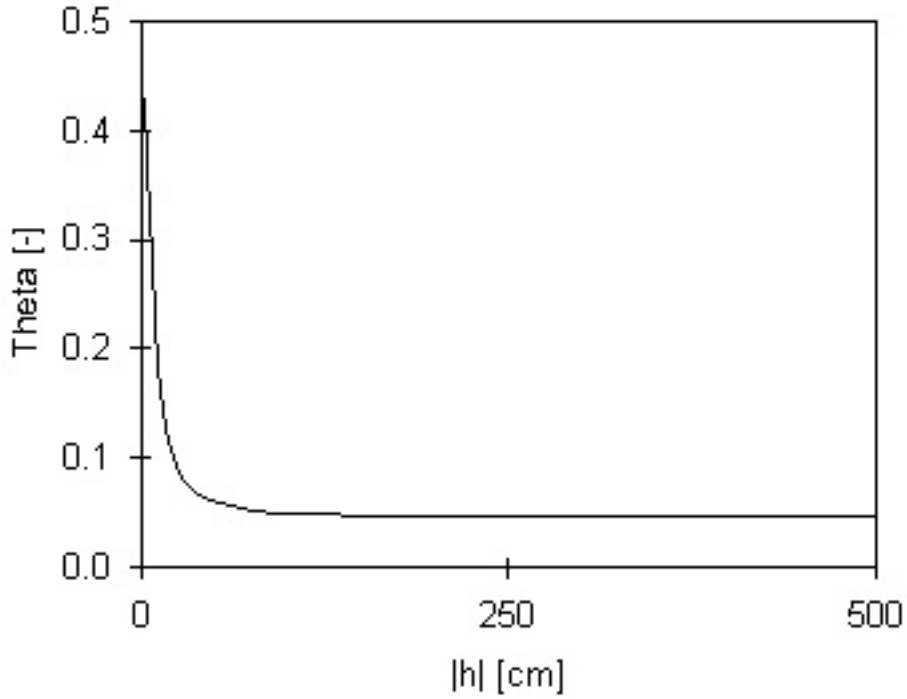

Figure 3-2. Soil moisture curve for limestone soil.

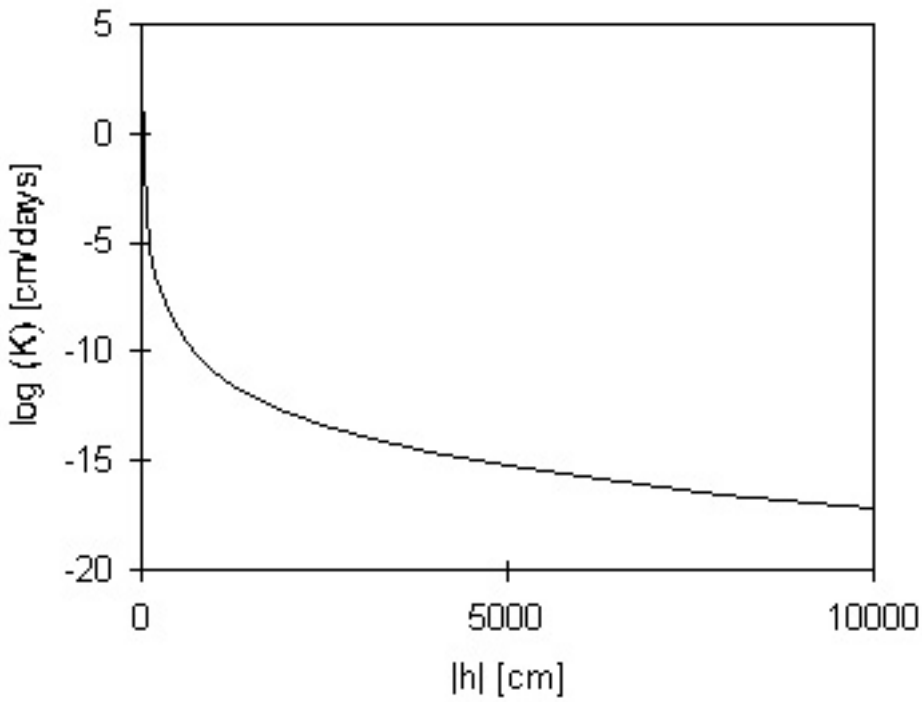

Figure 3-3. Hydraulic conductivity curve for sandystone soil.

3.1 Assessing Grid Error.

The finite element dimensions must be adjusted to the particular application. The resolution should be made relatively small in the regions where large hydraulic gradients are expected. Regions with sharp gradients are usually located in the vicinity of internal sources or sinks, or close to the soil surface where highly variable metrological factors can cause abrupt changes in pressure head. The size of elements can gradually increase with depth to reflect the gradually reduction in pressure head gradient at deeper depths. Therefor the discretization should have different resolution for each soil. For example coarse-textured soils (i.e. silt) require a finer discretizations than fine-texture soil (i.e. sand) (Izatk, 1988).

Numerical solutions of transport equation often exhibit oscillatory behavior and/or excessive numerical dispersion near relatively sharp concentration fronts. These problems can be especially serious for connective dominated transport characterized by small dispersivities. Undesired oscillation can be prevented by selecting appropriate combination of space and time discretizations. Two dimensionless numbers may be used to characterize the space and time discretizations. One of these is Peclet number, P, which defines the predominant type of the solute transport in relation to coarseness of the

finite element grid, which defined as:

$$P_i = \frac{q_i \Delta x_i}{\theta D_{ii}} \tag{3-1}$$

where q_i is component of the darcian fluid flux density, Δx_i is the character length of the finite element, θ is the water content and D_{ij} is the effective dispersion coefficient tensor in the soil matrix. The Peclet number increase when the convection part of the transport equation dominates the dispersive part. To achieve acceptable numerical results, the spatial discretization must be kept relatively fine to maintain low Peclet number. Numerical oscillation can be virtually eliminated when the Peclet numbers do not exceed about 5. The second dimensionless is the Courant number, C_r, which characterizes the relative extents of numerical oscillations. The Courant number is associated with the time discretization as follows:

$$C_{r_i} = \frac{q_i \Delta t}{\theta R \Delta x_i} \tag{3-2}$$

where Δt is the time increment and R is the solute retardation factor.

In this study, different discretizations were used based on the type of the problem and the type of soil texture. To asses the grid error, different finite element mesh size with different time

increment were used, starting with a fine element mesh that was gradually increased in size while monitoring the Peclet and Courant numbers which, should be less than 10.0 and 1.0 respectively.

3.2 Initial HYDRUS-2D model simulations

Water movement and solute transport will be simulated using the HYDRUS-2D model to quantify the effects of septic tank on the water and solute transport from and into the aquifer.

3.2.1 Model construction and inputs

3.2.1.1 Space and time discretizations

To carry out this assessing analysis, an initial model solution was obtained for a coarse finite element mesh. Then by decreasing the element size of the mesh till the solution does not change for the head and concentration at a particular observation point. Peclet and courant numbers will be monitoring for each run to make sure that there are not exceeding specific values.

The size of the model domain was 4351 m in width and 500 m in average depth, including a 100-m deep unsaturated zone, and 325-m deep saturated zone. The finite element grid consisted of a total of 3267 nodes and 9499 elements as shown in Figure 3-3. The grid was spaced approximately 10 cm near the septic systems and around 70 cm far away from the septic systems and in the saturated zone grid was larger.

The maximum simulation period was 3650 days. Time discretizations were as follows: the initial time step was $1*10^{-3}$ day, the minimum time step was $1*10^{-6}$ day, and the maximum time step was 1 day.

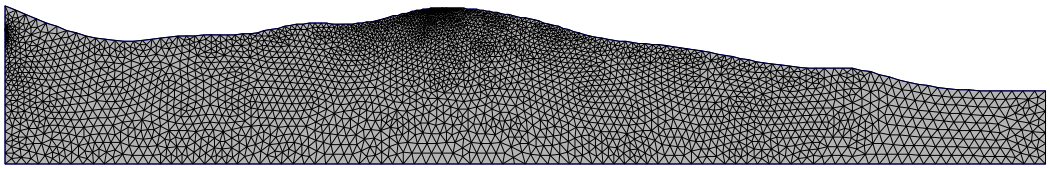

Figure 3-4. Finite element mesh.

3.2.1.2 Boundary and initial conditions

3.2.1.2.1 Water movement

In order to simulate water movement in the model, a Neuman boundary condition (no flow) is applied to the bottom and left and wright edge of the aquifer(in the A1/6 aquitard). An atmospheric boundary condition is chosen for the topsoil. For the septic tank a constant flux boundary was considered.

An initial head distribution conditions is gotten by simulating the aquifer which has been subjected to the regional flow of groundwater for a long period of time without existing of the septic systems in the study area and the initial head condition result is shown in Figure 3-5

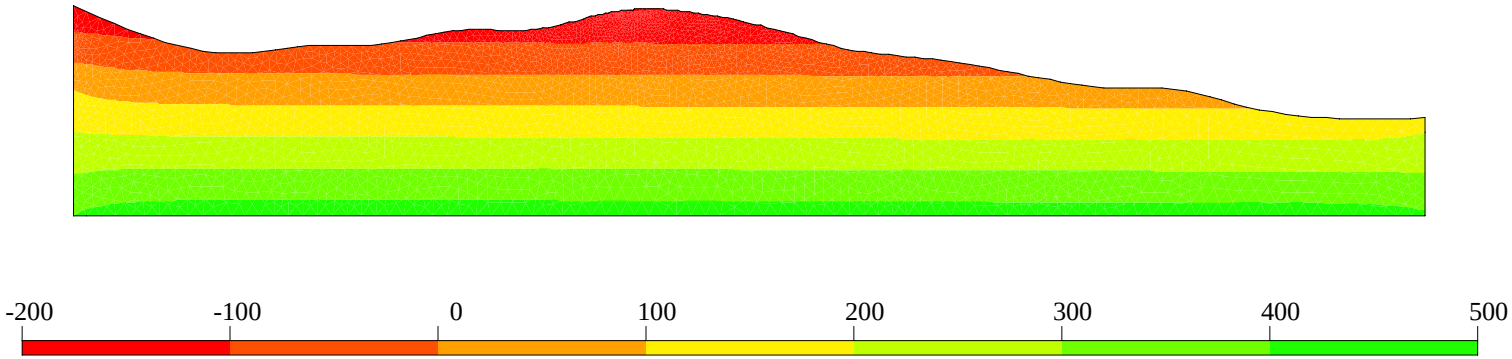

Figure 3-5. Initial head distribution in the model.

3.2.1.2 Solute transport

The left and right solute transport boundary conditions and the bottom boundary treated as the Neuman boundary condition (no flow boundary). The septic tanks at Al-Kfair town boundary are considered as third type of boundary condition. The septic tanks have constant concentration of Ammonium (225 mg/L). Regarding the initial condition, the model domain is assumed to be free of contaminant (i.e. $C_{(t=0,X,Z)}$= 0.0).

***3.3.1.3 Hydraulic and transport parameter*s**

Table 3-2 summarizes the hydraulic parameters used as input for the HYDRUS-2D.

Table 3-2 Hydraulic parameters used in HYDRUS-2D

	θ_r	θ_s	α (cm^{-1})	n	K_s(cm/day)	α_L (cm)	α_T (cm)
Kurnub	0.057	0.41	0.124	2.28	0.000006	52.5	5.25
Ajloun Group	0.07	0.36	0.005	1.09	0.004	7.145	0.7145

For the aquifer material, a longitudinal dispersivity (α_L), a value of 52.5 cm was used in the model (Domenico and Schwartz, 1998). A ratio of 10 was adapted for longitudinal to transverse dispersivity (α_T), based on the commonly used value in the literature. For the Ajloun group layer a value of 7.145 cm was used for the longitudinal dispersivity and the ratio between longitudinal and transverse dispersivity was used as 10 (Domenico and Schwartz, 1998).

Solute transport equations (3.1) and (3.2) for this situation was used:

$$R_1 \frac{\partial c_1}{\partial t} = D \frac{\partial^2 c_1}{\partial x^2} - v \frac{\partial c_1}{\partial x} - \mu_1 R_1 c_1 \qquad 3.3$$

$$R_i \frac{\partial c_i}{\partial t} = D \frac{\partial^2 c_i}{\partial x^2} - v \frac{\partial c_i}{\partial x} + \mu_{i-1} R_{i-1} c_{i-1} - \mu_i R_i c_i \qquad i = 2, 3 \qquad 3.4$$

where μ is a first-order degradation constant, D is the dispersion coefficient, v is the average pore water velocity (qx/θ) in the flow direction, x is the spatial coordinate in the direction of flow, and where it is assumed that 3 solutes participate in the decay chain. The three-species nitrification chain was used to present the real situation in the septic tank systems:

$$NH_4^+ \rightarrow NO_2^- \rightarrow NO_3^-$$

The experiment involves the application of a NH_4^+ solution to an initially solute-free medium (C_i = 0). The input transport parameters for the simulation are listed in Table 3-3.

. Table 3-3: summarizes the transport parameters.

Parameter	Value
v [cm/hour]	1.0
D [cm²/hour]	0.18
μ_1 [hour^{-1}]	0.005
μ_2 [hour^{-1}]	0.1
μ_3 [hour^{-1}]	0.0
R1 [-]	2.0
R2 [-]	1.0
R3 [-]	1.0
C_i *[-]*	0.0
$C_{0,1}$ *[-]*	1.0

The estimated values of physical and chemical parameters from the available literature were applied to the aquifer material. These parameters were then used as fixed inputs for the HYDRUS-2D model. Simple Van Genuchten parameters (Van Genuchten, 1980) will choose in the hydraulic functions.

4. Results and Discussions

In sewage effluent, N is almost entirely in the form of dissolved NH_4^+ (225mg/L) and NO_3^- concentrations are insignificant . Therefore, NH_4^+ was introduced to the model domain through the flux and concentration of the sewage discharge.

Considering the relatively dry climate in Al-Kfair town, a mean effluent discharge rate of 55 L /person/day was used. This assumes that all water used by a household is discharged through the wastewater disposal system. When dissolved NH_4^+ leaves the anaerobic environment of the septic tank, it is transformed to NO_3^- by autotrophic bacteria in the presence of O_2. Ammonium is nitrified to NO_3^- via a three-species nitrification chain reaction ($NH_4^+ \rightarrow NO_2^- \rightarrow NO_3^-$) as follows:

$$NH_4^+ + 1.5\ O_2 \rightarrow NO_2^- + H_2O + 2H^+$$

$$NO_2^- + 0.5\ O_2 \rightarrow NO_3^-$$

Under anoxic conditions and in the presence of an organic C source, the reverse process occurs and NO_3^- is reduced to N_2 gas. The denitrification chain reaction is

$$NO_3^- + 2CH_3OH \rightarrow 6\ NO_2^- + 2CO_2 + 4H_2O$$

$$6\ NO_2^- + 3CH_3OH \rightarrow 3N_2 + 3CO_2 + 3H_2O + 6OH^-$$

These processes can be considered in the HYDRUS model. Although an intermediate product, NO_2^- , is involved in a three-species chain reaction, the conversion between NH_4^+ and NO_2^- is so rapid that the reaction can be simplified to two single steps $NH_4^+ \rightarrow NO_3^-$ or $NO_3^- \rightarrow N_2$ gas

The simulation of the nitrification chain reaction was carried out for ten years.

Result for the first year simulation showed that:

- ✓ The plume of ammonium extended a round 67 m below Al-Kfair town.
- ✓ The Maximum concentration for Nitrite was 0.126 mg/L at 27.52 m below Al-Kfair town and the nitrite plume extended about 55.3 m below Al-Kfair town.
- ✓ The Maximum concentration for Nitrate was $6.39*10^{-4}$mg/L at 26.58 m below Al-Kfair town and the nitrite plume extended about 54.2 m below Al-Kfair town.

Figure 4.1 through 4.3 show the show the Ammonium, Nitrite and Nitrate concentration after one year.

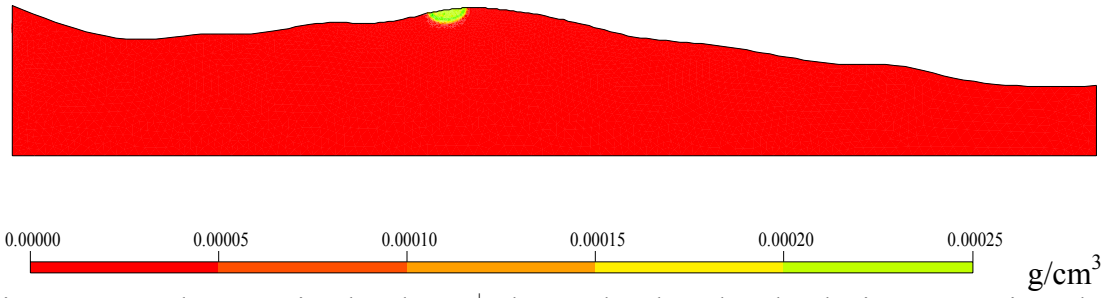

Figure 4.1: Hydrus-2D simulated NH_4^+ plumes developed under the impact septic tank systems of Al-Kfair (results at 1 year).

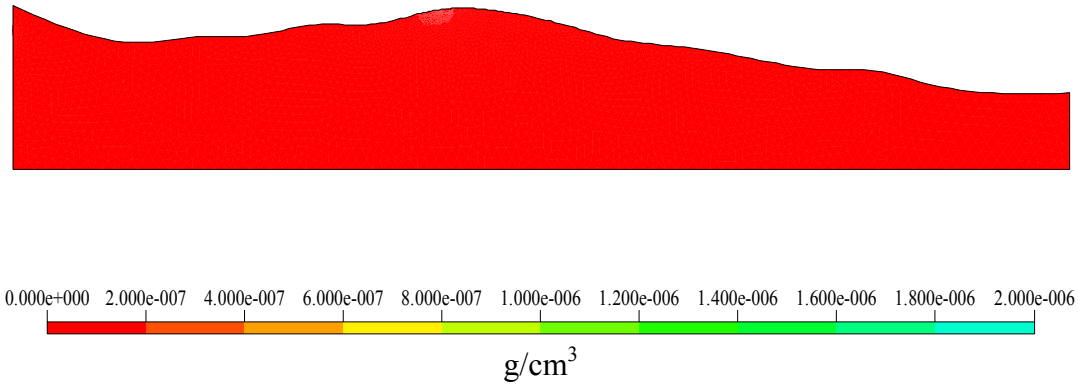

Figure 4.2: Hydrus-2D simulated NO_2^- plumes developed under the impact septic tank systems of Al-Kfair (results at 1 year).

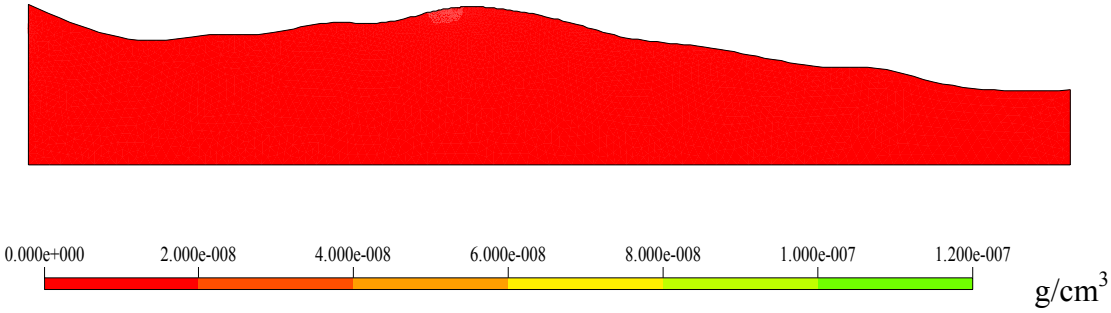

Figure 4.3: Hydrus-2D simulated NO_3^- plumes developed under the impact septic tank systems of Al-Kfair (results at 1 year).

The results of the fifth year simulation showed that:

- ✓ The plume of ammonium extended a round 160 m below Al-Kfair town
- ✓ The Maximum concentration for Nitrite was 0.764 mg/L at 86.51 m below Al-Kfair town and the nitrite plume extended about 146.3 m below Al-Kfair town.
- ✓ The Maximum concentration for Nitrate was $2.16*10^{-2}$ mg/L at 145.6 m below Al-Kfair town and the nitrate plume extended about 146.2 m below Al-Kfair town.

Figure 4.4 through 4.6 show the show the Ammonium, Nitrite and Nitrate concentration after five years.

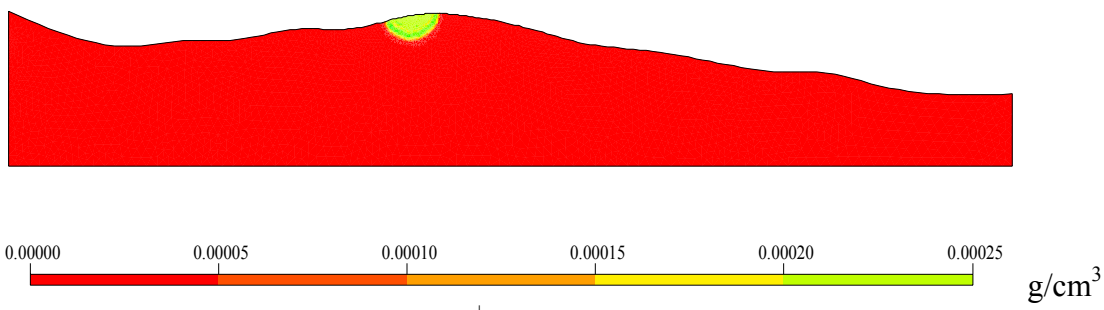

Figure 4.4: Hydrus-2D simulated NH_4^+ plumes developed under the impact septic tank systems of Al-Kfair (results at 5 years).

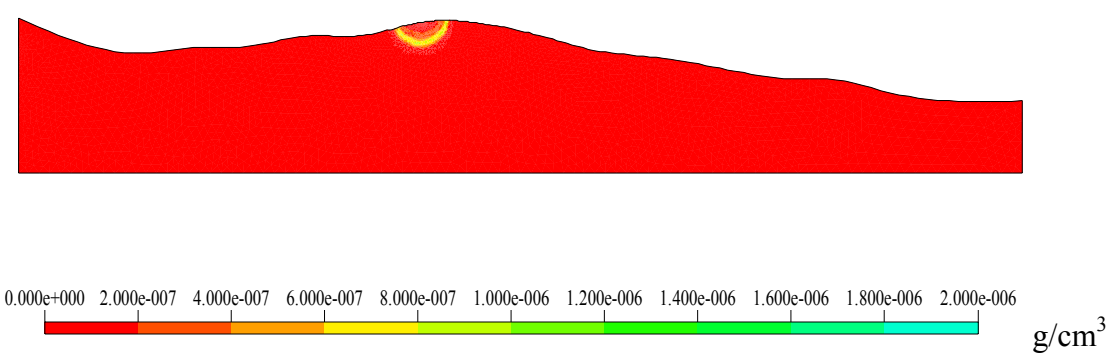

Figure 4.5: Hydrus-2D simulated NO_2^- plumes developed under the impact septic tank systems of Al-Kfair (results at 5 years).

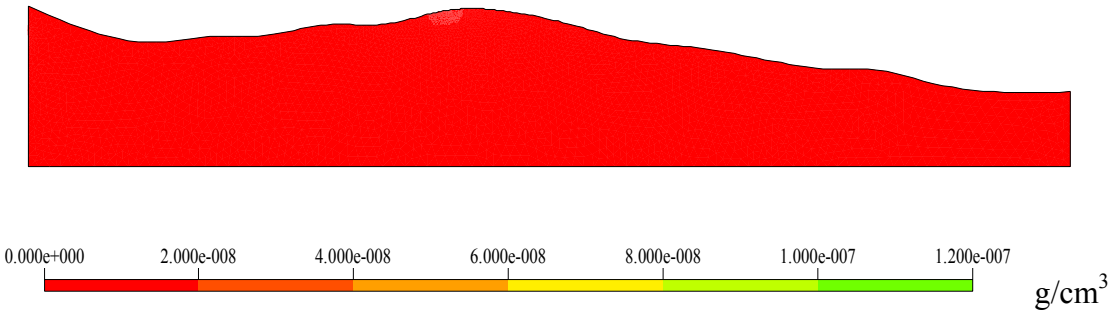

Figure 4.6: Hydrus-2D simulated NO_3^- plumes developed under the impact septic tank systems of Al-Kfair (results at 5years).

The result of the tenth year simulation showed that:

- ✓ The plume of ammonium extended a round 178 m below Al-Kfair town .
- ✓ The Maximum concentration for Nitrite was 1.632 mg/L at 130.68 m below Al-Kfair town and the nitrite plume extended about 170.82 m below Al-Kfair town.
- ✓ The Maximum concentration for Nitrate was $9.68*10^{-2}$ mg/L at 131.48 m below Al-Kfair town and the nitrate plume extended about 130.7m below Al-Kfair town.
- ✓ Figure 4.7 through 4.9 show the show the Ammonium, Nitrite and Nitrate concentration after ten years.

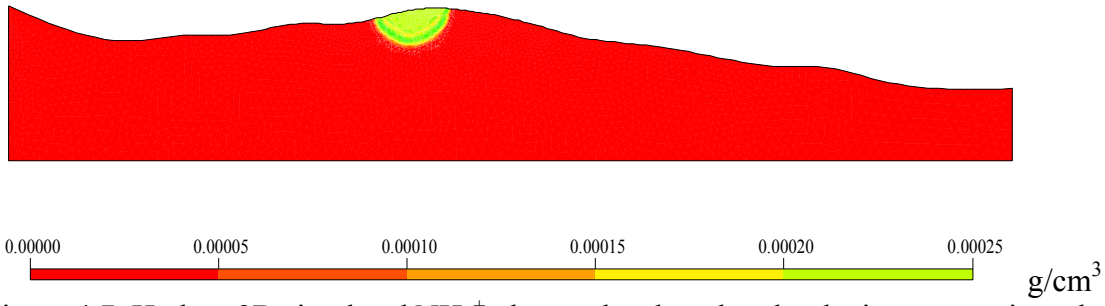

Figure 4.7: Hydrus-2D simulated NH_4^+ plumes developed under the impact septic tank systems of Al-Kfair (results at 10 years).

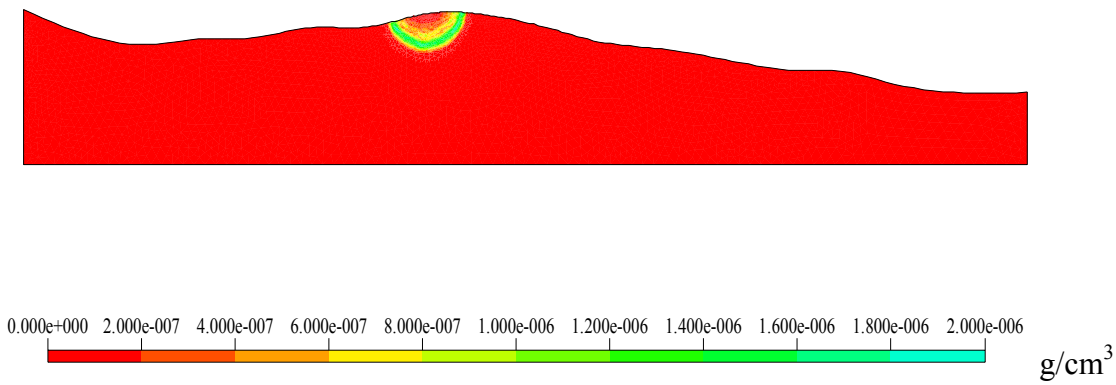

Figure 4.8: Hydrus-2D simulated NO_2^- plumes developed under the impact septic tank systems of Al-Kfair (results at 10 years).

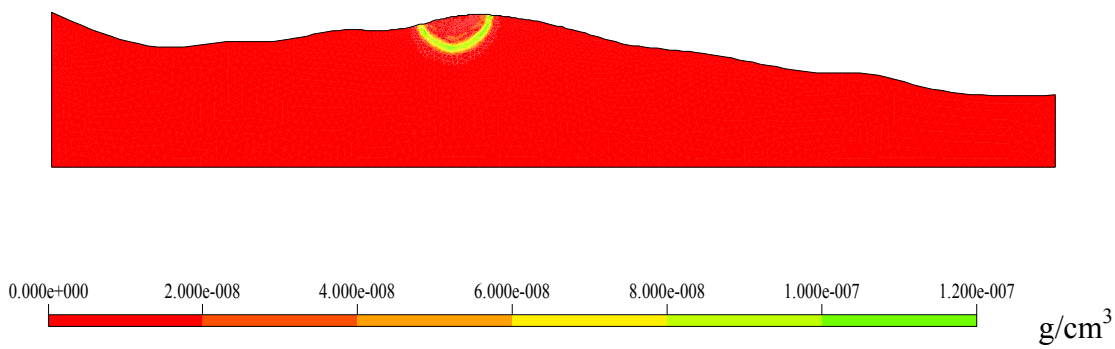

Figure 4.9: Hydrus-2D simulated NO_3^- plumes developed under the impact septic tank systems of Al-Kfair (results at 10 years).

Fortunately, the depth of the unsaturated zone below Al-kfair town is about 245 m, and the nutrient did not reach the saturated zone in Kurnub aquifer during the simulation period which was 10 years.

In addition if the NO_3^- and other contaminant coming from the septic tanks reach the saturated zone, there will be more dilution for that contaminant.

Although the cumulative impact of disposal systems is shown by model-simulated data, the NO_3^- concentrations in groundwater are still below the World Health Organization drinking water guideline of 50 mg /L (WHO, 2004), even where a high input effluent NH_4^+ concentration was used.

References

Bear, J. 1972. Dynamics of Fluids in Porous Media. Elsevier, New York.

Brooks, R.H., and A.T. Corey. 1966. Properties of porous media affecting fluid flow. J. Irrig. Drainage Div., ASCE Proc. (72IR2), 61-88.

Celia, M.A., D.E. Rolston and R.L. Zarba. 1990. A general mass conservative numerical solution for the unsaturated flow equation, Water Resour. Res., 26, 1483-1496.

Domenico, A.P., and F.W., Schwartz, 1998, Physical and chemical hydrogeology, John Wiley & Sons, New York.

Faust, C.R. and J.W. Mercer. 1980. Ground water modeling: Numerical models. Ground Water, 18, 395-409.

Fetter, C.W., 1993. Contaminant Hydrogeology. Macmillan. New York.

Jury, W.A., W.R. Gardener, and W.H. Gardner, 1991, Soil Physics. John Wiely & Sons, Inc., New York.

Kool, J.B., J.C. Parker. 1987. Development and evaluation of closed form expression for hysteretic soil hydraulic properties. Water Resour. Res., 23, 105-114.

Sawyer, C., *et al.*, *Chemistry for Environmental Engineering*, 4th Edition, McGraw-Hill, New York, 1994.

Simunek, J., M. Sejna and M. Th. van Genuchten. 1999. The HYDRUS-2D software package for simulating water flow and solute transport in two-dimensional variably saturated media, Version 2.0, U. S. Salinity Laboratory, USDA, ARS, Riverside, CA.

Scott, P. S., G. J. Farquhar, and N. Kouwen, Hysteresis effects on net infiltration, in *Advances in Infiltration*, *Publ. 11-83*, pp. 163–170, Am. Soc. of Agric. Eng., St. Joseph, Mich., 1983.

Tindall, A.J., and J. R., Kunkel, 1999. Unsaturated Zone Hydrology for scientist and engineers. Prentice Hall, New Jersey.

US EPA, Manual: Nitrogen Control, EPA/625/R-93/010, Office of Water, Washington, D.C., September, 1993.

Van Genuchten, M. Th. 1980. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils, Soil Sci. Soc. Am. J, 44, 892-898.

Wang, H.F. and M.P. Anderson. 1982. Introduction to Groundwater Modeling : Finite Difference and Finite Element Methods. Freeman. New York.

Water Environment Federation, *Natural Systems for Wastewater Treatment*, 2nd Edition, Manual of Practice FD-16, Alexandria, Virginia, 2001.

Watts, R.J. 1998. Hazardous Waste: Sources Pathways Receptors. John Wiley & Sons. New York.

Weber, W.J. and F.A. Digiano. 1996. Process Dynamics in Environmental Systems. John Wiley & Sons. New York.

Whelan, B. R, Barrow, N. J. (1984). The movement of septic tank effluent through sandy soils near Perth. I. Movement of nitrogen. *Australian Journal of Soil Research* **22,** 283-292.

World Health Organization. 2004. Guidelines for drinking-water quality: Vol. 1. Recommendations. 3rd ed. WHO, Geneva, Switzerland.