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Nitrate contamination of groundwater: A conceptual management framework

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Abstract

In many countries, public concern over the deterioration of groundwater quality from nitrate contamination has grown significantly in recent years. This concern has focused increasingly on anthropogenic sources as the potential cause of the problem. Evidence indicates that the nitrate (NO_3) levels routinely exceed the maximum contaminant level (MCL) of 10 mg/l NO_3-N in many aquifer systems that underlie agriculture-dominated watersheds. Degradation of groundwater quality due to nitrate pollution along with the increasing demand for potable water has motivated the adoption of restoration actions of the contaminated aquifers. Restoration efforts have intensified the dire need for developing protection alternatives and management options such that the ultimate nitrate concentrations at the critical receptors are below the MCL. This paper presents a general conceptual framework for the management of groundwater contamination from nitrate. The management framework utilizes models of nitrate fate and transport in the unsaturated and saturated zones to simulate nitrate concentration at the critical receptors. To study the impact of different management options considering both environmental and economic aspects, the proposed framework incorporates a component of a multi-criteria decision analysis. To enhance spatiality in model development along with the management options, the utilization of a land use map is depicted for the allocation and computation of on-ground nitrogen loadings from the different sources.

Keywords: Groundwater; Nitrate; Contamination; Management; Decision analysis; Conceptual; Fertilizer; Modeling

1. Introduction

Nitrogen (N) is an essential input for the sustainability of agriculture (Delgado, 2002; Shrestha and Ladha, 2002; Lake et al., 2003; Schröder et al., 2004). However, nitrate contamination of

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groundwater is a worldwide problem (Goodchild, 1998; Joosten et al., 1998; Birkinshaw and Ewen, 2000; Saãdi and Maslouhi, 2003; Kyllmar et al., 2004; Liu et al., 2005). Nitrate is soluble and negatively charged and thus has a high mobility and potential for loss from the unsaturated zone by leaching (DeSimone and Howes, 1998; Chowdary et al., 2005). Many studies showed high correlation and association between agriculture and nitrate concentration in groundwater (Ling and El-Kadi, 1998; Joosten et al., 1998; Harter et al., 2002; Shrestha and Ladha, 2002; Jordan and Smith, 2005; Dunn et al., 2005; Liu et al., 2005). The extensive use of fertilizers is considered to be a main non-point source of the nitrate that leaches to groundwater (Postma et al., 1991; Baker, 1992; Hubbard and Sheridan, 1994; Chowdary et al., 2005). In addition, point sources of nitrogen such as septic systems are shown to contribute to nitrate pollution of groundwater (MacQuarrie et al., 2001). Elevated nitrate concentrations in drinking water can cause *methemoglobinemia* in infants and stomach cancer in adults (Lee, 1992; Wolfe and Patz, 2002). As such, the US Environmental Protection Agency (US EPA) has established a maximum contaminant level (MCL) of 10 mg/l NO_3-N (US EPA, 2000).

Nitrate leaching from the unsaturated zone is a complex interaction of many factors (see Fig. 1) such as land use practices, on-ground nitrogen loading, groundwater recharge, soil nitrogen dynamics, soil characteristics, and the depth to water table (Birkinshaw and Ewen, 2000; Vinten and Dunn, 2001; Almasri, 2003; Almasri and Kaluarachchi, 2004a, 2005a). In addition, the

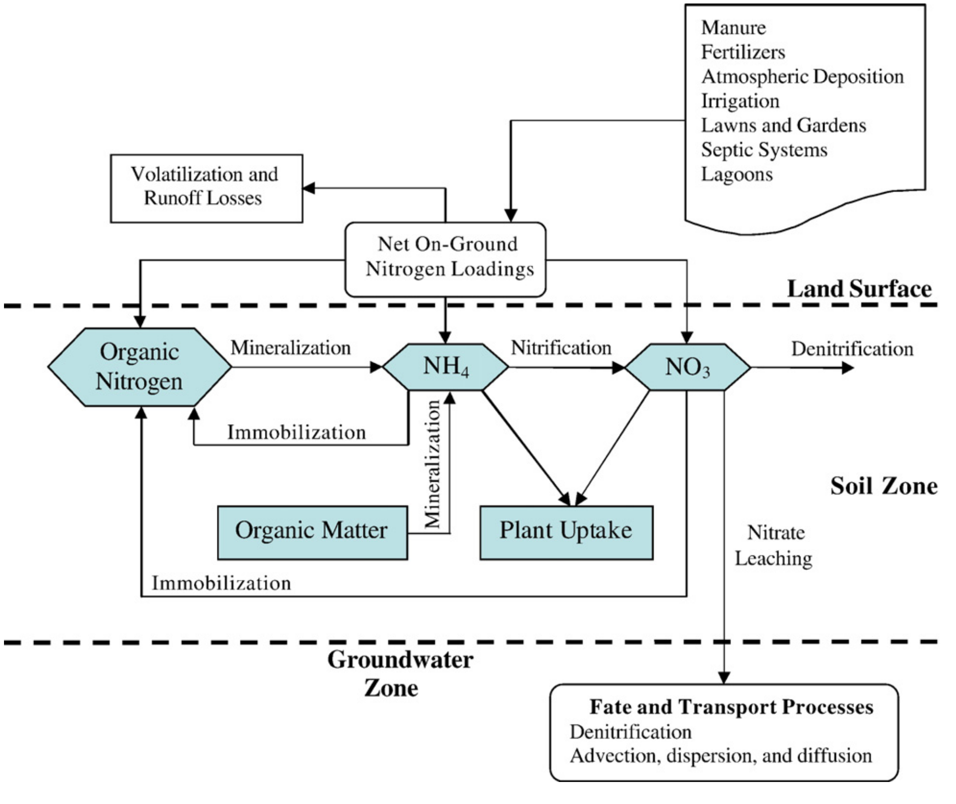

Fig. 1. A schematic representation of the integrated three-zone approach to conceptualize the interacting processes that govern nitrate occurrences in groundwater. Note that nitrate concentration in groundwater is ultimately a function of on-ground nitrogen loading.

temporal variation of nitrate leaching is largely affected by these parameters where many of them vary with time as well. Once reaches the groundwater, nitrate migrates in the aquifer via advection and dispersion. Nitrate, most likely, undergoes denitrification in this zone depending on the properties and prevailing conditions (Frind et al., 1990; Postma et al., 1991; Korom, 1992; Tesoriero and Voss, 1997; Tesoriero et al., 2000; Shamrukh et al., 2001). A nitrate fate and transport model in groundwater (NFTM) can be developed and used, in conjunction with a soil nitrogen model, to simulate the effectiveness of current and future agricultural practices and/or other management options to control nitrate occurrences in groundwater (Mercado, 1976; Ling and El-Kadi, 1998; Kyllmar et al., 2004). This is apparent from Fig. 1 where the spatial distribution of nitrate concentration in groundwater is a function of on-ground nitrogen loading from the dissimilar sources present in the area of concern, the soil dynamics and the groundwater conditions.

Since the management options entail conflicting objectives which in turn yield different economic consequences and diverse prioritization schemes based on the practicality and the applicability of each proposed option, a multi-criteria decision analysis is needed to prioritize the proposed management options and to balance between competing economic and environmental goals. This complex interaction of the different influencing factors motivates the development of a conclusive management framework that entails the assessment and evaluation of nitrate contamination extent, the utilization of mathematical simulation models, and the employment of a multi-criteria decision analysis.

This paper demonstrates a general conceptual management framework for controlling nitrate contamination in groundwater at the critical receptors with nitrate contamination exceeding the MCL. The different components of the management framework are elucidated and discussed.

2. General background

Agricultural practices result in non-point source pollution of groundwater and the effects of these practices accumulate with time (Baker, 1992; Livingston and Cory, 1998; DeSimone and Howes, 1998; Schilling and Wolter, 2001; Liu et al., 2005). Non-point sources of nitrogen from agricultural practices include fertilizer and manure application (Styczen and Storm, 1993; Hubbard and Sheridan, 1994; Paul and Zebarth, 1997; Shamrukh et al., 2001; Harter et al., 2002; Almasri and Kaluarachchi, 2004b; Chowdary et al., 2005). In addition to agricultural practices, non-point sources of nitrogen include dissolved nitrogen in precipitation, irrigation return flows, and dry deposition (Almasri and Kaluarachchi, 2004a, 2005a). Point sources of nitrogen such as septic systems and dairy lagoons are shown to contribute to nitrate pollution of groundwater. In high-density residential areas with no sewer systems, septic systems produce substantial amounts of nitrogen to the subsurface especially in the form of ammonium and organic-N (MacQuarrie et al., 2001). Similar to septic tanks, seepage from dairy lagoons was found to be the source of elevated nitrate in groundwater in many dairy farms and proximity areas (Erickson, 1992).

Identification of areas with heavy on-ground nitrogen loadings from point and non-point sources is important for land use planners, environmental regulators and decision makers. Once such high-risk areas have been identified, preventative measures can be implemented to minimize the risk of nitrate leaching to groundwater (Tesoriero and Voss, 1997). Accurate quantification of nitrate leaching is difficult as there are complex interactions of land use, on-ground nitrogen loading, recharge, soil nitrogen dynamics, soil physical and chemical characteristics, and depth of soil as indicated in Fig. 1 (Ling and El-Kadi, 1998; Vinten and Dunn, 2001; Dunn et al., 2005). When conducting analysis and modeling of nitrate contamination, it is essential to understand the interaction of the aforementioned factors to account for the transient and spatially variable nitrate leaching to groundwater (Hallberg and Keeney, 1993; Refsgaard et al., 1999; Almasri, 2003) as well as the temporal and spatial distribution of nitrate concentration in groundwater.

Soil nitrogen dynamics affect nitrate leaching to groundwater (Ling and El-Kadi, 1998; Refsgaard et al., 1999; Almasri and Kaluarachchi, 2004a). Several models have been developed for simulating the fate and transport of nitrate in soils and details of available models can be found in Ma and Shaffer (2001) and McGechan and Wu (2001). Management of groundwater pollution due to nitrogen in large watersheds requires information from simple soil nitrogen models as these models require less data (Ling and El-Kadi, 1998; Hall et al., 2001). Nevertheless, a realistic balance between simplicity and complexity should be maintained when selecting a soil nitrogen model. Too complex models are highly demanding in terms of data while very simple models do not capture the spatial and temporal variability in nitrate leaching (Dunn et al., 2005). Nitrogen transformation models are essential in designing management options to minimize groundwater pollution that aim at reducing nitrate leaching (Ling and El-Kadi, 1998; Refsgaard et al., 1999; Birkinshaw and Ewen, 2000; Almasri and Kaluarachchi, 2004a). The models are able to predict the soil nitrogen buildup for a given on-ground nitrogen loading pattern and can predict nitrate leaching to groundwater for different management alternatives (Kyllmar et al., 2004). Meisinger and Delgado (2002) provide a good and comprehensive discussion on the principles of managing nitrogen leaching. However, it is insufficient to merely quantify nitrate leaching when developing management frameworks and indeed it is indispensable to account for the saturated zone since nitrate undergoes fate and transport processes herein. Such processes largely alter nitrate mass and concentration in groundwater.

Once nitrate leaches to groundwater from the unsaturated zone, it will be advected and dispersed with the groundwater flow and will likely undergo denitrification (see Fig. 1). A successful management framework of nitrate contamination of aquifers integrates the unsaturated and saturated zones through model coupling. This coupling is practically achieved through the utilization of the output of the nitrate soil model (mass of nitrate leaching) as an input to the NFTM. These NFTM models are essential and the keystone in a successful assessment of the efficiency of the proposed management options and for characterization of the impact of the current practices (Mercado, 1976; Ling and El-Kadi, 1998; Birkinshaw and Ewen, 2000; Shamrukh et al., 2001; Almasri and Kaluarachchi, 2005b).

Regional assessment of groundwater quality is complicated by the fact that nitrogen sources are highly spatial (Goderya et al., 1996; Tesoriero and Voss, 1997). The knowledge of the spatial distribution of nitrogen loading can identify the areas where groundwater needs to be protected (Vinten and Dunn, 2001). This assessment is also of great importance in designating areas that can benefit from pollution prevention and monitoring programs (Goodchild, 1998; Schilling and Wolter, 2001; Vinten and Dunn, 2001; Dunn et al., 2005). Management options to protect groundwater quality are improvements to agricultural practices, land use changes, and designation of protection areas (Goodchild, 1998; Latinopoulos, 2000; Jordan and Smith, 2005). For instance, manure application rates on lands should be in accordance with agronomic requirements and nitrogen mass present in the soil (Livingston and Cory, 1998). Land use changes include redistribution of land use classes and changes in the land use in areas that are vulnerable to contamination. However, this option may not be feasible in many instances due to competing stakeholder concerns arising from economic constraints and these constraints may, sometimes, override the environmental concerns. The management actions in protection areas consist of limiting the application of organic manure and chemical fertilizers (Latinopoulos, 2000). Apparently, a successful management framework should account for the different management options by realistically being able to simulate their impacts.

Spatial analysis techniques are needed in assessing and managing nitrate leaching to groundwater in large watersheds and indeed to persuasively visualize the nitrate concentration in groundwater spatially and temporarily. A geographic Information System (GIS) is a sound approach to evaluate the outcomes of various management alternatives. GIS greatly enhances data collection and processing, the assessment of nitrate leaching risk indices, designating areas of dissimilar vulnerability, model development, and management scenarios (Wylie et al., 1995; Nolan et al., 1997; Refsgaard et al., 1999; Lasserre et al., 1999; Shaffer et al., 2001; Hall et al., 2001; Almasri and Kaluarachchi, 2003, 2004a; Lake et al., 2003; Jordan and Smith, 2005; Dunn et al., 2005). GIS technology is also helpful in facilitating model input and output processing especially in watersheds where field data are regularly updated from frequent monitoring and hence GIS allows rapid visualization of raw data.

3. The management framework

Fig. 2 depicts the general proposed management framework for nitrate contamination of groundwater. The framework consists of two major parts; i.e., the nitrate contamination analysis and decision making. The framework starts off by data collection for the area of concern and corresponding analysis to delineate areas of nitrate contamination and to better characterize the problem extent. Thereafter, the possible sources of contamination are identified in order to efficiently set up the management options that target these sources while reducing nitrate concentration below the MCL. Decision analysis utilizes and employs the developed management options to choose the best option with the aid of the soil and groundwater models. A detailed description of the management framework components is given below in the following subsections.

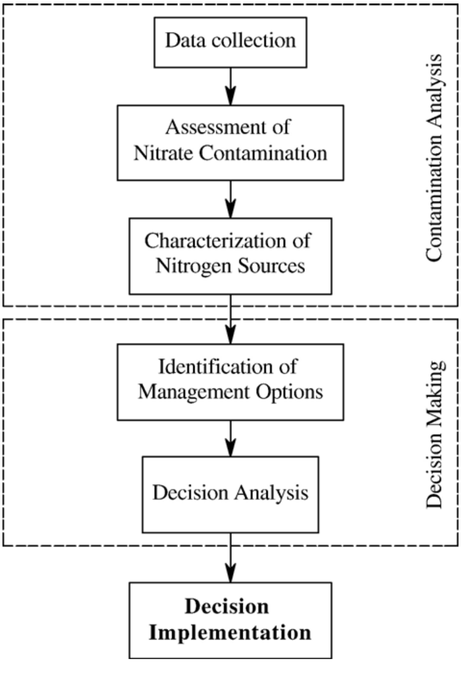

Fig. 2. A flow chart depicting the general proposed management framework for nitrate contamination control.

3.1. Data collection

This is an established step in the development of frameworks for the management of contamination in groundwater. The very objective of data collection is to gather and compile the required information that will be later utilized in the development and implementation of the framework. Fig. 3 depicts the main relevant data that need to be collected. As can be seen from Fig. 3, the data were classified into three main components for the ease of presentation. Sources of data are diverse and include chiefly local reports, scientific publications, field visits, interviews, and personal communication. However, the internet offers in general a great deal of helpful information though quality assurance and reliability should be observed. The data depicted in Fig. 3 are intended to be used throughout the course of the development and implementation of the management framework. Much of the data summarized in Fig. 3 are generally utilized in developing the soil and groundwater models as depicted in Fig. 1 and later illustrated in this paper. For instance, part of the data that will be utilized in model development is for model calibration and verification while another part will be employed in carrying out the multi-criteria decision analysis.

After data collection is accomplished, it is important to have the data in a manageable and processable format such as a GIS-based shapefile and/or as a spreadsheet. This is of great importance in order to be able to expedite the process of framework development and indeed for ease of visualization and assessment. Many obtained maps will require digitization while other data such as climate-related data may require the conversion into a spreadsheet.

3.2. Assessment of nitrate contamination

This is an important step in the sense that it compels the motivation for the development and implementation of the management framework and the corresponding options for a particular

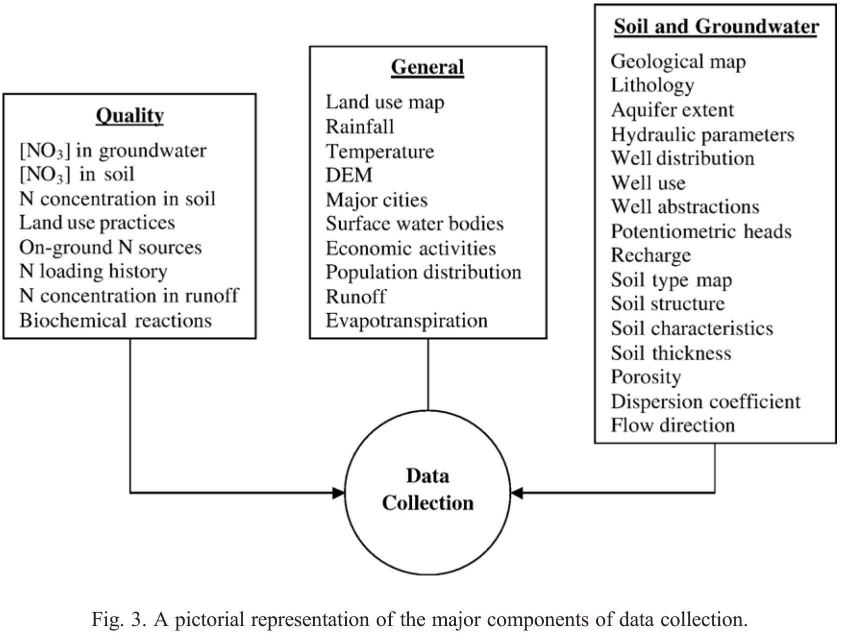

Fig. 3. A pictorial representation of the major components of data collection.

area. In addition, this assessment highlights the areas that encounter elevated nitrate occurrences and will indeed draw attention to the critical receptors. Such receptors will be considered when developing the management options such that nitrate concentrations at these receptors ought to be below the MCL. Apparently, the spatial implementation of the management options will be considered in the vicinity of these receptors or at least in areas that can largely affect the concentration at these receptors. Also, model calibration and verification rely largely on the determination of the nitrate concentration in groundwater at the sampled locations. As an example of nitrate contamination assessment, Fig. 4 depicts the spatial distribution of nitrate concentrations for Sumas-Blaine Aquifer, Whatcom County, Washington State, US. The aquifer encounters high applications of fertilizers and manures since it is overlain by dense agricultural activities (refer to Almasri, 2003 for more details regarding Sumas-Blaine Aquifer). As can be noticed from the figure, many areas witness nitrate concentration beyond the MCL and the areas that possibly encounter high concentrations were highlighted using the Thiessen Polygon approach. Apparently, the elevated nitrate concentrations in the aquifer necessitate the development and implementation of management options to control the encountered elevated contamination. In addition, statistical analysis of the nitrate concentrations across the study area of interest ought to be considered.

3.3. Characterization of contamination sources

After the collection of the data and the assessment of the nitrate occurrences in groundwater, much information becomes available regarding the potential contributing sources. The main outcome of this component of source characterization is to quantitatively determine for each source the corresponding loading, spatially and temporarily (see Fig. 1 for a list of these sources).

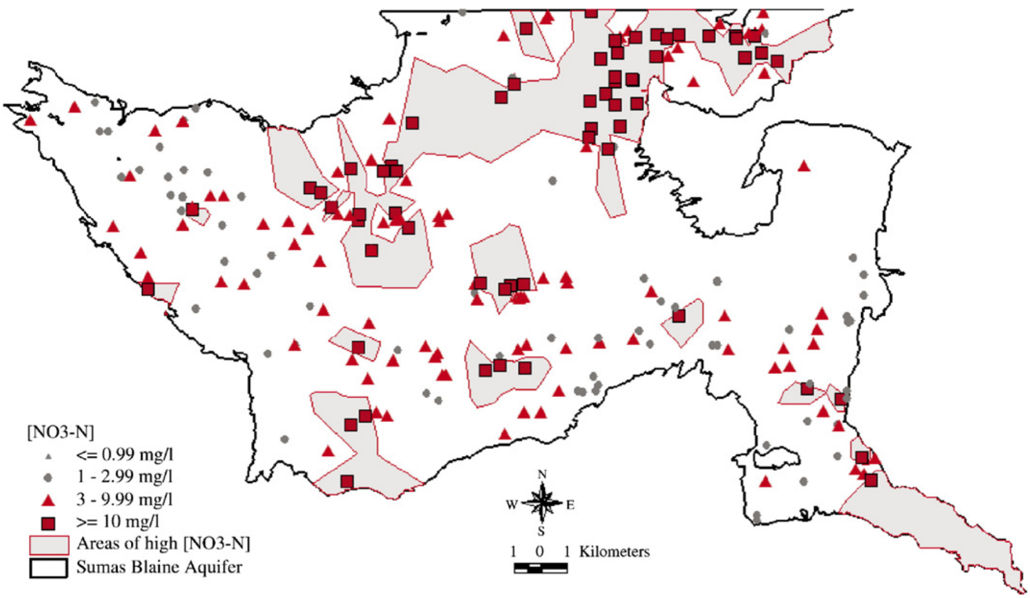

Fig. 4. Nitrate concentration data for 1997 for Sumas-Blaine Aquifer. Shaded polygons represent the extent of the areas of elevated nitrate concentrations beyond the MCL.

For instance, Fig. 5 depicts the breakdown of the percentages of on-ground nitrogen loadings from the different sources present in Sumas-Blaine Aquifer. This at least provides a preliminary understanding and early appreciation to the influential nitrogen sources within the study area of concern and may indeed direct the efforts into the data that need to be collected. Besides, one can easily expect what nitrogen sources the management options should address when being developed and implemented.

For instance, Fig. 5 suggests that dairy manure is the largest nitrogen source in Sumas-Blaine Aquifer and that dairy manure and fertilizers account for 88% of all nitrogen generated in the area. This indeed indicates that successful management options should target these sources. Although these sources very likely to have a more distributed impact yet other sources should not be overlooked since these have local impacts and thus could be responsible for the elevated nitrate concentrations at certain receptors (Erickson, 1992). Detailed explanation regarding nitrogen source allocation and computation is provided in the next sub-section since this step overlaps with model development.

3.4. Mathematical model development

To facilitate the efficient development of management options and to identify nitrate concentration distribution due to the current land use practices, mathematical models ought to be developed and employed such that the outcomes of the management options are known beforehand (see Fig. 1 for the relationship between on-ground nitrogen sources and nitrate concentration in groundwater). In other words, aquifer response, in terms of nitrate concentrations at the critical receptors, due to the proposed management options will be simulated such that the necessary actions are adopted with the minimum strictness.

Fig. 6 illustrates the flow chart for the development of the mathematical models of on-ground nitrogen loading and nitrate dynamics in soil and groundwater. The mathematical models utilize the land use classes to assign the spatial distribution of the on-ground nitrogen loadings from the different sources present in the area of concern as well as to aid in the development of the management options such that these options are implemented at the correct locations.

An example of land use map and its classes is the grid of the National Land Cover Database (NLCD) of the US Geological Survey. The NLCD grid provides the land cover distribution across 21 land use classes for the entire US and can be utilized with the aid of GIS to designate the spatial

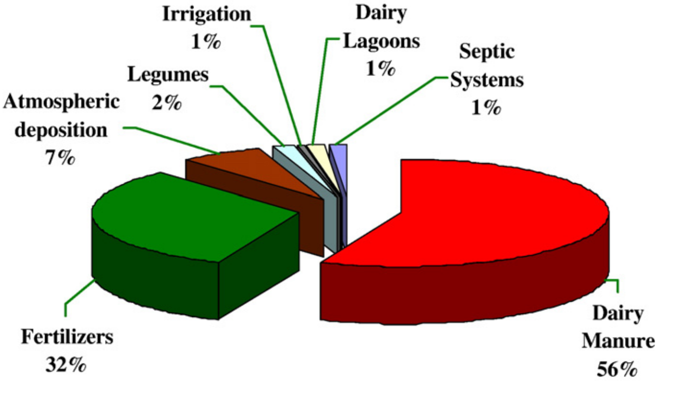

Fig. 5. Percentages of annual on-ground nitrogen loadings from the different sources present in Sumas-Blaine Aquifer.

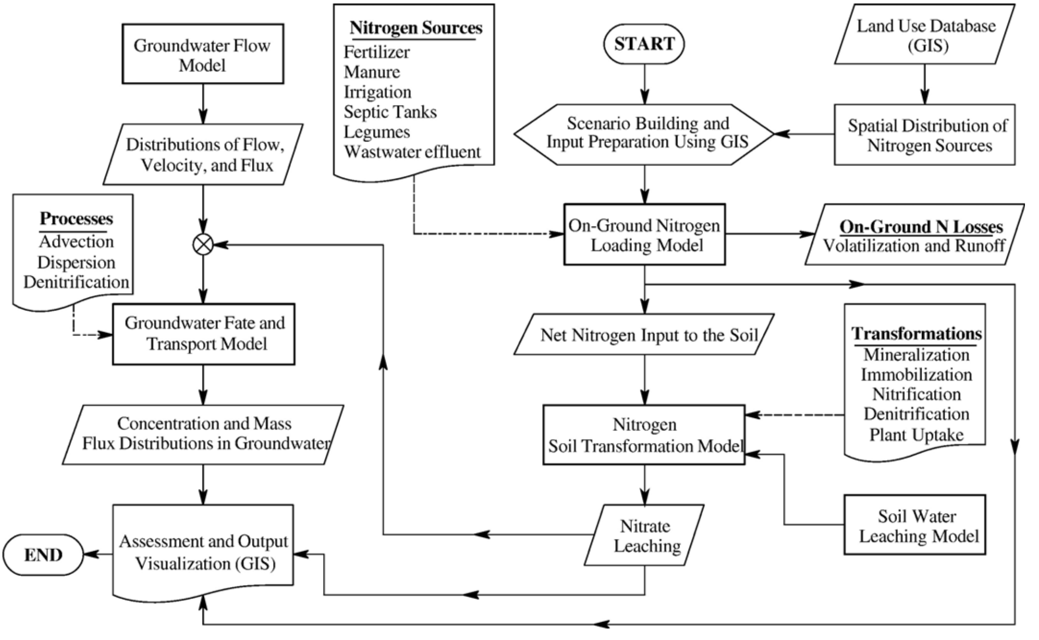

Fig. 6. A flow chart describing the development of the mathematical models for the ultimate assessment of nitrate concentrations at the critical receptors in response to the implementation of the management options.

distribution of on-ground nitrogen loadings (Nolan et al., 2002; Almasri and Kaluarachchi, 2003, 2004a, 2005a; Almasri, 2003). The general sources of nitrogen in agriculture-dominated watersheds and in general (see Fig. 1) are N-rich fertilizers, dairy and poultry manure, septic systems, dairy lagoons, wet and dry deposition, lawns and gardens, irrigation recharge, and legumes. The spatial distribution of the total on-ground nitrogen loadings is due to the different nitrogen sources that contribute concurrently to a given land use class. For example, the land use class of dairy farm receives nitrogen from manure application, wet deposition, regional and dairy dry deposition, and nitrate-contaminated water used in irrigation. Table 1 shows the different nitrogen sources that concurrently contribute to each NLCD class. In addition to the spatial distribution of the on-ground nitrogen loadings, there is also a temporal variability due to the time variability in the applications of the different nitrogen sources such as inorganic fertilizer application, time of animal grazing, precipitation, irrigation, and dairy farm lagoon operations. It is worth mentioning that this temporal behavior in practices dictates nitrate leaching variability with time.

Unlike the nitrogen sources that are dispersed in nature, septic systems and dairy farm lagoons are concentrated sources and thus must be treated as point sources. They are processed independently from the NLCD grid by using GIS point shapefiles that provide the respective locations and the associated data. Such sources can be significant in terms of nitrogen loading both at local and regional levels as depicted in Fig. 7 which shows the spatial distribution of the septic tanks in Sumas-Blaine Aquifer. A detailed illustration of the computations of the on-ground nitrogen loadings are provided in Almasri and Kaluarachchi (2003), Almasri (2003), and Almasri and Kaluarachchi (2004a, 2005a).

Once the distribution of net on-ground nitrogen loading is computed after accounting for the surface losses of nitrogen due to volatilization and runoff (similar computations are furnished in

Table 1
The allocation of the different nitrogen sources to the dissimilar land cover classes of the NLCD

NLCD class	Dairy manure	Wet deposition	Dry deposition (regional)	Dry deposition (dairy)	Irrigation	Fertilizer	Lawns	Legumes
Open water								
Perennial ice/snow								
Low intensity residential		✓	✓				✓	
High intensity residential		✓	✓				✓	
Commercial/Industrial/Transportation								
Bare rock/sand/clay		✓	✓					
Quarries/Strip mines/Gravel pits		✓	✓					
Transitional		✓	✓		✓	✓		
Deciduous forest		✓	✓					
Evergreen forest		✓	✓					
Mixed forest		✓	✓					
Shrubland		✓	✓					
Orchards/Vineyards/Other		✓	✓		✓	✓		
Grasslands/Herbaceous		✓	✓		✓	✓		
Pasture/Hay		✓	✓		✓	✓		✓
Row Crops		✓	✓		✓	✓		
Small grains		✓	✓		✓	✓		
Fallow		✓	✓		✓	✓		
Urban/Recreational/Grasses		✓	✓				✓	
Dairy farms	✓	✓	✓	✓	✓			
Woody wetlands		✓	✓					
Emergent herbaceous wetlands		✓	✓					

Almasri and Kaluarachchi, 2004a), soil nitrogen dynamics can be consequently modeled including the basic soil transformations as depicted in Figs. 1 and 6. Many studies in the literature did highlight the importance of using mathematical models for the simulation of nitrogen soil dynamics and the corresponding nitrate leaching (see for instance Shaffer et al., 1991; Goderya et al., 1996; Ling and El-Kadi, 1998; Birkinshaw and Ewen, 2000; Ma and Shaffer, 2001; McGechan and Wu, 2001; Dunn et al., 2005).

The major soil transformation processes that greatly affect nitrate leaching are mineralization-immobilization, nitrification, denitrification, and plant uptake (see Fig. 1 for the different pathways of nitrogen dynamics). Mineralization is the formation of ammonium during the microbial decomposition of organic-N and immobilization is the assimilation of inorganic nitrogen by microorganisms to form organic-N. Nitrification is the microbial oxidation of ammonium to nitrite and further to nitrate. Denitrification is an anaerobic process by which bacteria converts nitrate to N_2 and N_2O gases. These reactions depend on pH, temperature, soil water content, and soil biological characteristics. Soil characteristics dictate the nitrogen kinetics. In well-drained soils, infiltration is high, therefore, the rate of nitrification is high and denitrification may be insignificant. Soil depth controls the time lag between the on-ground application of nitrogen and nitrate leaching and it influences the time span of soil nitrogen transformations. This indeed affects

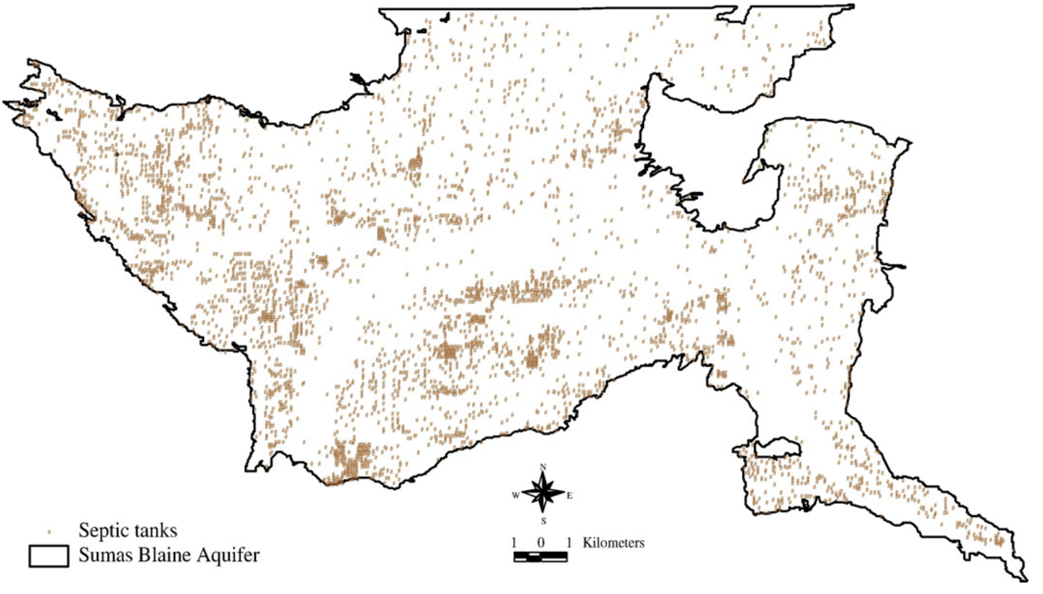

Fig. 7. The spatial distribution of septic tanks in Sumas-Blaine Aquifer.

the amount of nitrate leaching to groundwater. As such, it is important to utilize a model that considers the nitrogen loadings, plant uptake, and nitrogen transformation processes while maintaining simplicity. An example of such models is the one developed by Ling and El-Kadi (1998). They developed a lumped parameter model that utilizes an analytical solution derived for the simulation of nitrate leaching from the unsaturated zone.

As depicted earlier in Figs. 1 and 6, the ultimate output from the soil model is nitrate leaching to groundwater which is utilized in the development of the NFTM. The NFTM requires as an input the specification of nitrate concentration distribution in all water sources including areal recharge and nitrate concentration in any lateral inflow to the aquifer. The NFTM solves the advection–dispersion equation with first rate decay reaction and thus requires the determination of the velocity field. This compels the development of a groundwater flow model to provide this required information to the NFTM (Shamrukh et al., 2001; Chowdary et al., 2005). Examples of groundwater flow and fate and transport models include MODFLOW (Harbaugh and McDonald, 1996) and MT3D (Zheng and Wang, 1999); respectively.

It is worth mentioning that the groundwater flow model can be developed assuming steady-state conditions while for the NFTM it is developed considering transient conditions perhaps with monthly input data. A major step in model development (mainly for groundwater flow and nitrate fate and transport) is the calibration after going through the established steps and the customary protocol in model development (not depicted in Fig. 6). Practically, the most influential parameters to be considered in calibration can be designated through a prior sensitivity analysis. Nevertheless, hydraulic conductivity and denitrification coefficient are the major parameters that dictate to a large extent the groundwater flow and nitrate fate and transport in groundwater. In many instances, nitrate leaching from soil to groundwater ought to be altered to bring in more nitrates into the aquifer (or the contrary). This can be attained through altering the reactions that take place in the soil zone through the coefficients that dictate these dynamics. Thereafter, model verification must be undertaken to build the trust in the model simulation output (Almasri, 2003).

It is important to incorporate the uncertainty and variability of decision criteria in the methodology to ensure an informed decision-making process. Many sources of uncertainty surround and encase the proposed modeling framework. The specific and most important is the uncertainty in the nitrate concentration in groundwater at the critical receptors. Sources of the uncertainty and variability can be attributed in general to the high uncertainty associated with the following: (i) hydrogeologic parameters of the subsurface (for instance hydraulic conductivity); (ii) characteristics of the nitrogen sources and concentrations; (iii) magnitudes of nitrogen losses due to runoff and volatilizations; (iv) soil and groundwater nitrogen transformation rates; (v) nitrate leaching coefficients; and (vi) groundwater recharge rates.

Models can account for the uncertainty and variability in the input parameters by means of probability distributions of such parameters while the models provide the statistical distribution of the nitrate concentrations at the critical receptors. For instance, Table 2 summarizes the probability density functions that describe the uncertainty of the key parameters of the soil nitrogen model (Almasri and Kaluarachchi, 2004a). To model the nitrate concentration values due to the concurrent uncertainty and variability of the input parameters, Monte Carlo method can be used (Khadam and Kaluarachchi, 2003; Almasri and Kaluarachchi, 2004a).

Hydraulic conductivity is the most important hydrogeologic property that is spatially variable in heterogeneous aquifers. Hydraulic conductivity is considered as a spatially correlated random field where many models are available to describe this random stochastic field using a lognormal

Table 2
Probability density functions describing the uncertainty of key parameters of soil nitrogen models (Almasri and Kaluarachchi, 2004a)

Parameter	Unit	Distribution	Range	Most likely
On-ground N loadings				
N in milking cow for manure	lbs/year	Triangle	165–250	210[a]
N in dry cow for manure	lbs/year	Triangle	120–180	150[a]
N in heifer for manure	lbs/year	Triangle	60–90	75[a]
N in calf for manure	lbs/year	Triangle	74–112	93[a]
Lagoon seepage rate	mm/day	Triangle	0.1–5.0	1.0[b]
Fertilizer lawn rate	lbs/acre-year	Uniform	110–260	–
Dairy farm dry deposition	lbs/acre-year	Uniform	5–13	–
Per capita loading from septic systems	lbs/year	Uniform	9–14	–
On-ground N losses				
N losses in runoff	%	Uniform	0–10	–
Ammonia loss from fertilizer	%	Uniform	2–30	–
Ammonia loss from manure	%	Uniform	20–45	–
Soil N transformation rate				
Mineralization rate	day^{-1}	Triangle	0.008–0.05	0.014[c]
Nitrification rate	lbs/acre-day	Uniform	2–30	–
Denitrification constant	%	Uniform	3–9	–
NO_3 leaching coefficients				
Available water for leaching	inches/year	Uniform[d]	1–5 to 36–50	–
Leaching coefficient	–	Uniform	1.0–1.2	–

[a] After Meisinger and Randall (1991).
[b] After Cox and Kahle (1999).
[c] After the analysis of 38 values presented in Stanford and Smith (1972).
[d] Taken as uniform intervals that vary spatially as in Vaccaro et al. (1998).

distribution (Dagan, 1979; Gelhar, 1993). Several methods do exist to generate the random fields of the hydraulic conductivity such as the turning band method of Thompson et al. (1989).

It should be kept in mind that the execution of the simulation models of (i) on-ground nitrogen loadings, (ii) nitrogen transformations in the unsaturated zone, (iii) the groundwater flow, and (iv) the nitrate fate and transport in groundwater (see Fig. 6) necessitates the use of a large number of random fields which indeed is a time-consuming process. In order to reduce this effort, the minimum number of simulations needed to achieve statistical convergence can be estimated using the approach described by Lahkim and Garcia (1999). In this approach, the cumulative mean and variance are evaluated and assessed for convergence based on the number of random fields included in the simulation (Almasri, 2003; Khadam and Kaluarachchi, 2003).

3.5. Development of management options

In order to minimize the elevated nitrate concentrations in groundwater at the critical receptors, management options ought to be developed, introduced, and later implemented. A successful management framework in this regard should promote management options that are amenable to implementation and efficient in lowering the elevated nitrate concentrations. Since management options will eventually affect land use practices, negative economic ramifications would be expected. Careful designation of the management options would be highly required such that the less severe yet efficient options should be recommended and later implemented.

As mentioned earlier, NFTM provides a great deal of aid in spatially designating the sources that have a high impact on nitrate concentration in groundwater. In agriculture-dominated watersheds where many sources dictate nitrate dynamics and occurrences in groundwater, it is necessary to single out the sources associated with each nitrate receptor. That is, on-ground nitrogen sources and corresponding loadings that are located in areas down-gradient from a critical receptor will never affect the concentration in the receptor except through dispersion which is of minimum effect compared to advection or when the groundwater flow is transient and the flow pattern changes drastically with time. In other words, there is a high association between nitrate contamination and groundwater flow directions (Liu et al., 2005). Thus, such down-gradient areas (with the sources and corresponding loadings) should not be targeted when developing the management options. The delineation of influencing areas is persuasively carried out using the NFTM after the development of a groundwater flow model.

It is worthwhile to mention that each receptor may have a different influencing area based on its spatial location. In addition, the boundaries of the influencing area may change temporarily due to the possible variability in the stresses such as pumping rates and recharge from varying rainfall. Fig. 8 depicts the concept of the influencing areas for two different nitrate receptors for a site located in Sumas-Blaine Aquifer (Almasri and Kaluarachchi, 2005a). The procedure of determining the area along with the proper selection of the radii is presented in detail in Almasri and Kaluarachchi (2005a). However, it should be kept in mind that when carrying out regional decision analysis regarding nitrate contamination, the management options are generally implemented for the entire area of concern. It might be difficult to limit the implementation of these management options to certain areas as suggested above due mainly to political constraints.

The management of nitrate contamination of groundwater resources entails in general the control of potential sources that are linked to man-made activities and these in turn can be classified into point and non-point sources as stated earlier. Many of these sources contribute to the problem concurrently. To deal with such situations, different management options can be introduced where each management option can address a specific source that is not addressed by

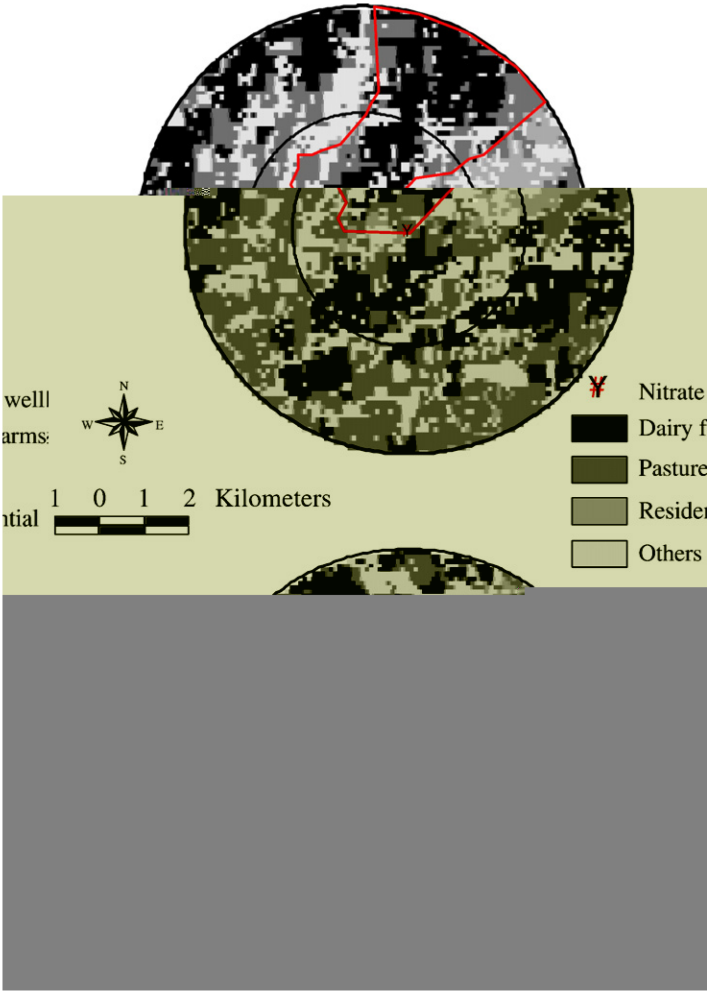

Fig. 8. A schematic for the upgradient contributing areas for two different receptors for different input parameters pertaining to the NLCD classes of dairy farms, pasture, and residential. The land use classes are summarized to four classes for ease of presentation.

other management options. On the other hand, the same polluting source can be controlled through different management options but at different levels of efficiencies and indeed at different cost ramifications and economic consequences.

Table 3 summarizes a set of realistic management options that can be considered when managing nitrate contamination in groundwater resources. In agriculture-dominated areas, the management options should consider the possible minimization of the application of nitrogen-based fertilizers along with other sources as can be inferred from Table 3 (Almasri, 2003) and in concordance with Fig. 1. It should be kept in mind that Table 3 is just an example of an array of possible management options and other options can be considered including for instance the prevention of the use of untreated effluent from wastewater in irrigation, changing land use type from agriculture to industrial, crop rotation, altering the timing of fertilizer application, and the use of nitrification inhibitors.

Table 3
Summary description of suggested management options related to non-point agricultural practices (Almasri, 2003)

ID	Description
Option 1	Do nothing (maintain current loadings)
Option 2	Dairy cattle herd reduction
Option 3	Manure composting/exporting
Option 4	Fertilizer application reduction
Option 5	Adopt a feeding strategy for dairy cattle
Option 6	Adopt a feeding strategy for dairy cattle + fertilizer application reduction
Option 7	Manure composting/exporting + fertilizer application reduction
Option 8	Manure composting/exporting + adopt a feeding strategy for dairy cattle
Option 9	Manure composting/exporting + fertilizer application reduction + adopt a feeding strategy for dairy cattle

To illustrate the issue of the dissimilarity in the effectiveness of the different management options in controlling nitrate concentration in groundwater, two receptors were considered that are located in different locations in Sumas-Blaine Aquifer and results are depicted in Fig. 9 (Almasri, 2003). Fig. 9 demonstrates that options 7 and 9 are the sole management options that meet the MCL constraint at Receptor 1. This receptor is located in a pastured area. Worthwhile noting is that no single management option is capable of meeting the MCL limit for this receptor while; for instance, upon combining management options 3 and 4 to get management option 7, MCL limit was met. Apparently, fertilizer reduction, management option 4, is not decisive, yet this management option has a supporting impact when combined with management option 3. Management options simulated at Receptor 2 (see Fig. 9) located in a dairy farm area show different efficiency levels of management options as compared to those at Receptor 1. Apparently, reducing manure loading, management option 2 or 3, as a single management option is the only single management option that exactly satisfies the MCL constraint where the constraint is tight. As can be concluded from Fig. 9, there is no need for a reduction in fertilizer loading, and management options 4, 6, 7, and 9 are not technically needed. In a broad sense, specific management options may be efficient for some receptors but may not for others. This largely depends on the spatial location of the receptor and the total on-ground

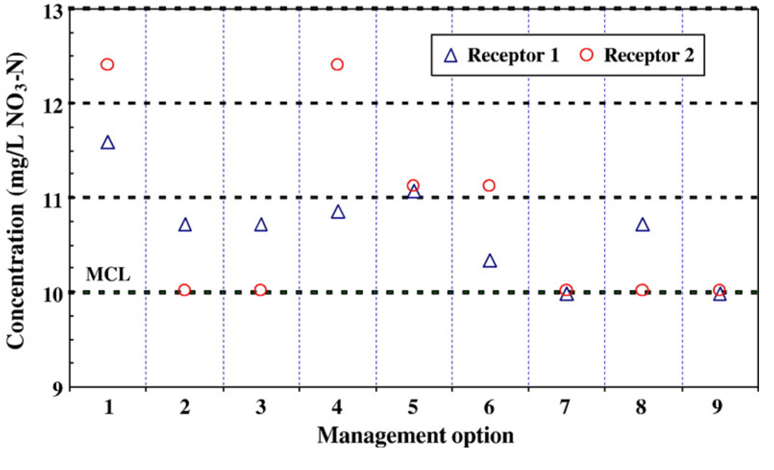

Fig. 9. Nitrate concentrations for two receptors located in Sumas-Blaine Aquifer for the different management options summarized in Table 3. Receptor 1 is located in a pasture area and Receptor 2 is located in a dairy farm area.

nitrogen loading in the surrounding contributing area along with other physical, chemical, and hydrological properties of the subsurface (Almasri, 2003).

It is important to decide on the planning period by the end of which nitrate concentrations at the critical receptors ought to be below the MCL. Long planning periods beyond 10 years are common for reducing nitrate concentration below the MCL (Lee, 1992; Almasri, 2003). With increasing the planning period, less severe management options are needed to bring the nitrate concentrations below the MCL and vice versa.

3.6. Assessment of decision criteria

Decision making to select the best management option requires the identification of the decision objectives, which is decisive to the outcome. The direct objectives herein include the minimization of elevated nitrate occurrences in groundwater at specified critical receptors, reducing the health risk to the exposed population to the extent feasible, and minimizing the cost of the best management option. This set of objectives is not inclusive and may include a variety of other objectives based on the extensiveness of the decision field and the interests and attitudes of the decision maker. The proposed framework introduces two broad sets of decision criteria; environmental and economic to measure the desirability of each management option in accordance with the decision objectives (Almasri, 2003).

The environmental criteria include on-ground nitrogen loading and losses, mass buildup in the subsurface, nitrate leaching to groundwater, and nitrate concentration distribution at the critical receptors. The economic criteria include the total cost associated with the implementation of the management options and the cost per concentration reduction (Almasri, 2003; Almasri and Kaluarachchi, 2005b). Mathematical models can be utilized to evaluate the decision criteria for each proposed management option as can be deduced from Fig. 6. Table 4 summarizes the broad decision criteria to be later used in the multi-criteria decision analysis.

The criteria related to the satisfaction of the concentration constraints include the summation of positive deviations of concentrations from MCL for receptors exceeding the MCL and the total number of receptors that are exceeding MCL. The later criterion accounts for situations in which few receptors have high nitrate concentrations far above the MCL while a large number of

Table 4
Summary of the decision criteria, the corresponding acronyms, and the evaluation methodology (Almasri, 2003)

#	Description of decision criteria	Acronym	Evaluation
1	Summation of concentration deviations above MCL (mg/l)	SCD	NFTM
2	Number of receptors exceeding MCL (−)	EMCL	NFTM
3	Net cost ($)	COST	Economic model
4	Cost per unit concentration reduction ($/mg/l)	CPCR	Economic model
5	Nitrate buildup in the groundwater (lbs)	NBGW	NFTM
6	Cumulative nitrate flux to the surface water (lbs)	NFSW	NFTM
7	Nitrate leaching (lbs)	NL	Soil model
8	Nitrate buildup in the soil (lbs)	NBS	Soil model
9	Ammonium buildup in the soil (lbs)	ABS	Soil model
10	Organic nitrogen buildup in the soil (lbs)	OBS	Soil model
11	Total nitrogen buildup in the soil (lbs)	TNBS	Soil model
12	Total on-ground nitrogen loading (lbs)	OGNL	Loading model
13	On-ground nitrogen runoff losses (lbs)	OGRL	Loading model
14	On-ground nitrogen volatilization losses (lbs)	OGVL	Loading model

receptors may have nitrate concentrations in tiny violations to the MCL. The summation of positive concentration deviations, SCD, is computed for each management option as follows:

$$\mathrm{SCD} = \sum_{k=1}^{z} (C_k - 10) \quad \text{for} \quad C_k > 10 \tag{1}$$

where C_k is the maximum monthly nitrate concentration (mg/l NO_3-N), at the end of the planning period at receptor k and z equals the number of nitrate critical receptors.

The on-ground-related decision criteria consider the on-ground nitrogen loading and nitrogen losses due to runoff and volatilization. Such criteria are important to account for the surface water quality preservation. The soil-related decision criteria consider the nitrate leaching to groundwater and the soil build-up of nitrate, ammonium, organic nitrogen, and total nitrogen. The groundwater-related criteria include nitrate build-up in the groundwater as well as nitrate flux to surface water bodies. This flux accounts for the surface water quality concerns especially in situations where the groundwater baseflow plays an important role in providing streamflow.

Fourteen decision criteria can be proposed for the multi-criteria decision analysis as summarized in Table 4. Each management option has to be appraised for these decision criteria by using the soil and groundwater nitrate fate and transport models and the economic model as summarized in Table 4.

For the economic criteria, two broad cost criteria were considered including net cost and cost per unit concentration reduction. Net cost is defined as the difference between the cost associated with a management option and the benefit. Costs are given a positive sign.

The cost per unit nitrate concentration reduction, CPCR, is defined as follows:

$$\mathrm{CPCR} = \frac{\mathrm{COST}_i - \mathrm{COST}_0}{\mathrm{AC}_i - \mathrm{AC}_0} \tag{2}$$

where COST_0 and COST_i are the net cost incurred from the "do nothing" option and the ith option, respectively; and AC_i and AC_0 are the average concentrations corresponding to the ith and the "do nothing" options, respectively. AC_i is defined as in the following equation:

$$\mathrm{AC}_i = \frac{1}{z} \sum_{k=1}^{z} C_k^i \tag{3}$$

where C_k^i is the simulated nitrate concentration at receptor k corresponding to the ith management option. The reason behind introducing the CPCR parameter is to evaluate the cost effectiveness of a specific management option in the overall reduction of nitrate concentrations (Almasri, 2003).

3.7. Multi-criteria decision analysis

As mentioned earlier, the wide multiplicity in potential management options along with the different levels of nitrate contamination occurrences, especially at a regional scale, entails the employment of a multi-criteria decision analysis to effectively assess the efficacy of each management option. The use of multi-criteria decision analysis enables the relative assessment of the efficacy of the management options after setting up the evaluation criteria against which these management options will be appraised (Yakowitz et al., 1993; Khadam and Kaluarachchi, 2003).

The management options to control nitrate contamination of groundwater entails conflicting objectives. For instance, keeping up the current land use practices (the do nothing option) will not entail acute negative economic ramifications. However; more deterioration of the aquifer water quality would lead to severe environmental consequences. Thus, for this specific option along others, conflict in decision criteria will take place. Multi-criteria decision analysis is a good resort to be considered when dealing with decisions that connote conflicting criteria.

Fig. 10 depicts the general multi-criteria decision analysis process for the Importance Order of Criteria (IOC) method. The IOC is a simple method for ranking the management options aimed at reducing nitrate concentrations in groundwater. As can be concluded from Fig. 10, decision analysis depends mainly on the use of the simulation models to design possible realistic management options that are relevant to the area of concern and apparently to provide the outcome of these management options (Almasri, 2003; Almasri and Kaluarachchi, 2005b). In addition, the simulation models provide the critical areas of high influence on nitrate occurrences in groundwater through conducting sensitivity analysis.

The IOC is one of the simple yet efficient multi-criteria decision analysis methods. The method was successfully applied in many studies in the water and environmental-related decision analysis (see for instance Khadam and Kaluarachchi, 2003; Almasri, 2003). The IOC method (Yakowitz et al., 1993) is conceptually simple and provides the decision maker with clear evidence if one management option is strongly dominant over another. The IOC method is easy to program (for instance using a spreadsheet) and provides rational results (see Almasri, 2003; Almasri and Kaluarachchi, 2005b). The method relies on defining the best and worst total utilities of the management options through the ranking of the decision criteria for each management option. Detailed illustration of the IOC method is provided in Yakowitz et al. (1993).

However, it is worth mentioning that there is a great deal of different multi-criteria decision analysis methods that can be used herein other than the IOC method. For instance, the analytic hierarchy process (AHP) first developed by Saaty in 1971 is a powerful multiple-criteria decision analysis technique for dealing with complex problems. AHP provides an understanding and

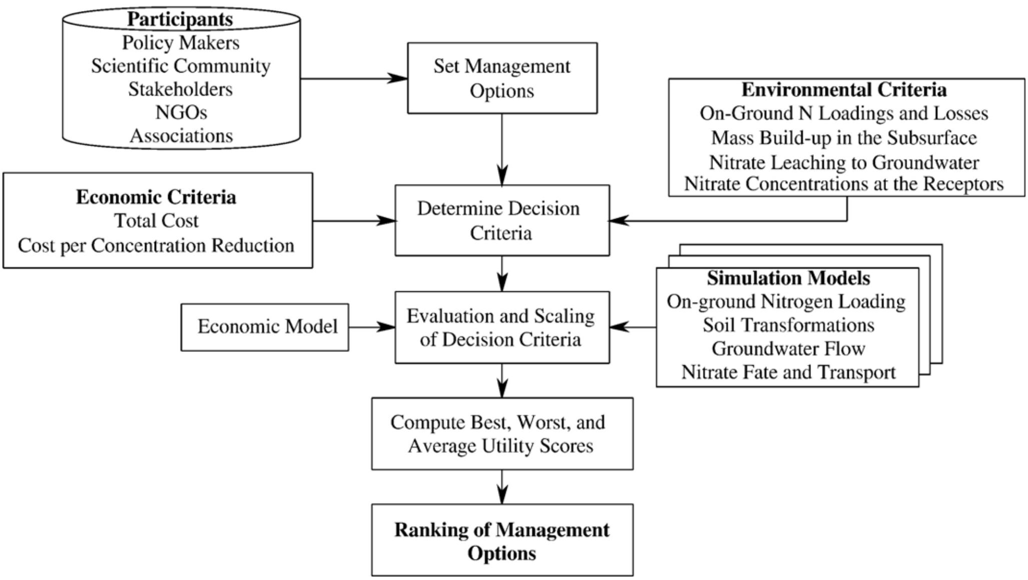

Fig. 10. Pictorial representation of the multi-criteria decision analysis approach for the selection of the best management option for controlling nitrate contamination of groundwater.

awareness of the problem in consideration. The comparison of all pairs of objectives and decision alternatives required to calculate the expected utilities provides the decision maker with a ranking of alternatives. In addition, it provides a consistency index that is used to measure the coherence that decision makers have in their judgments. Recently, Banuelas and Antony (2004) developed a modified version of the method (MAHP). The MAHP is a systematic and stochastic procedure for identifying and representing the element problems hierarchically. It is a stochastic process rather than deterministic, provides means to test difference in the results statistically and incorporates management issues probabilistically. Details on the AHP can be found in Saaty (1980) and Saaty and Alexander (1981).

4. Discussion

The conceptual management framework described herein consists of different components. These components are linked to each other in a sequential manner where for instance the spatial distribution of total on-ground nitrogen loading is assessed and thereafter fed to a soil nitrogen model for the quantification of nitrate leaching to groundwater and so on and so forth (see Fig. 1). Many difficulties may arise when intending to implement the management framework depicted in Fig. 2 along with the modeling flowchart shown in Fig. 6 and the multi-criteria decision analysis as shown in Fig. 10. The fundamental difficulties are summarized in the following:

[1] The variability in on-ground nitrogen sources along with the different properties of the subsurface compels the development of management options that are spatially distributed. For instance, at regional-scale decision analysis for the control of nitrate contamination in groundwater, land use types and applications play an important role in dictating the occurrences of nitrate in the subsurface. It is thus important to utilize GIS capabilities for the processing of land use map such that a meaningful spatial resolution of management options can be arrived at;

[2] Since modeling is a fundament and core component of the management framework (see Fig. 1), it is essential to pay a great deal of attention when developing the soil and groundwater models. This in turn will be a highly demanding process in terms of time and data. Time will be mainly allocated for model conceptualization, development, calibration, verification, and sensitivity analysis. In each of these steps, tiny details need to be considered depending on the complexity of the models;

[3] The implementation of the proposed framework follows an order. However, for an efficient use of this framework especially at a regional level with fine decision resolution, an automated execution of the framework components ought to be considered. In this automated process, a programming effort is needed wherein the output from one component is automatically extracted, processed, and prepared in a proper format, and ultimately fed to the subsequent component. In addition, a code is required to summarize the outcome of the framework in a meaningful way for a quick assessment and in a format that can be further processed. This is an important feature especially for regional-scale analysis when the output is of immense size;

[4] The very premise of this framework is the minimization of the nitrate concentration at specified (critical) receptors where MCL limit must not be violated upon the adoption of the management options. In many situations these nitrate receptors are largely dispersed across the model domain. As such, different areas will end up affecting different receptors while specific receptors will be affected in different magnitudes by different areas. This will result

in considering different management options or in other words different decision variables. These decision variables would imply, for instance, different suggested fertilizer loadings at different areas. Apparently, with the increase in the total number of receptors more decision variables will transpire. This will lead to consider the implementation of this framework within an optimization framework especially when different reduction fractions are to be considered for the source loadings; and

[5] In general, management options influence the agricultural practices in agriculture-dominated watersheds and may entail at some level changes to established land use activities ending up at the delineation of protection areas. However, these changes and alterations may not be feasible in many instances due to competing stakeholder concerns arising from economic constraints and these constraints may, sometimes, override the environmental concerns. Thus, it is quite essential to follow a sort of "bottom-up" management approach that concentrates on the involvement of participants from different interested agencies and entities including policy makers, scientific community, stake-holders, and NGO personnel. On top of that and most importantly is the participation of ordinary citizens such as villagers and farmers. Their participation is of great importance in order to come up with realistic and sound management options that can compromise the competing objectives of both the maximization of economic return and the minimization of the negative ramifications to the environment. Not even that but also more commitment is warranted since a wide spectrum of participants will take a role and thus more people will abide by since these management options were suggested, promoted, and agreed upon by them at the outset. Nevertheless, the key role of this participation and involvement should focus on selecting management options and corresponding decision criteria necessary to carry out a multi-criteria decision analysis.

5. Summary and conclusions

Elevated nitrate concentration in groundwater is a widespread problem worldwide. A conceptual management framework to control and minimize nitrate contamination in groundwater is exemplified. The framework integrates two major steps; nitrate contamination assessment and simulation and decision analysis. Nitrate contamination simulation entails the assessment of on-ground nitrogen loadings from different sources and the utilization of soil and groundwater fate and transport models. Since the decision criteria connote conflicting objectives, a multi-criteria decision analysis is proposed. Once this proposed management framework is adopted and implemented, it is expected that nitrate elevated concentrations at the critical receptors will drop subject to the proper selection of the management options. Overall, the management framework is straightforward and represents the important processes that dictate, to a great deal of extent, the nitrate occurrences in groundwater. The following are the main conclusions:

[1] The major advantage of the management framework is that it tracks properly nitrogen pathways since the introduction at the ground surface until it reaches the critical receptors in the groundwater;

[2] The utilization of a land use map makes it possible to account for the spatiality in the on-ground nitrogen loadings and allows the realistic allocations of the different nitrogen sources present in the area of concern;

[3] Modeling is an important component in the proposed management framework for the control of nitrate contamination in groundwater at the critical receptors. This importance

comes out from the ability of the models to simulate and predict the nitrate concentration distribution due to the implementation of the management options and thus to assess the efficacy of these options; and

[4] Since the management options generally entail competing criteria (for instance, economic versus environmental), the utilization of a multi-criteria decision analysis helps in ranking the management options and facilitates the determination of the best one.

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