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Improved methods for aquifer vulnerability assessments and protocols for producing vulnerability maps, taking into account information on soils

**EXECUTIVE SUMMARY**

The overall objective of this project is to improve methods for aquifer vulnerability assessments and protocols for producing vulnerability maps, taking into account soils information. The project is subdivided into several phases and this phase comprises an outline of the approach taken to derive a new GIS algorithm for vulnerability assessments.

There are two main groups of methods for assessing groundwater vulnerability, namely:

1. Index or subjective rating methods,
2. Statistical or process-based methods.

These two groups were addressed, using GIS, in this project. The first is based on a revised “DRASTIC” approach and is called the “ReSiS” approach, and the second, called “UGIf” is based on a deterministic approach. These two methods were chosen for further modifications and adaptations to South African conditions in order to come up with better and improved approaches for assessing groundwater vulnerability.

To reiterate, the main difference between these two models is that ReSiS is an index based approach for assessing mainly intrinsic vulnerability whereas the UGIf model is a process based analytical model that can be used for assessing contaminant specific vulnerability. For a case study assessment for vulnerability the Coastal Park portion of Cape Flats aquifer area was selected.

The ReSiS method takes into account groundwater recharge, the vulnerability of the soil zone, the vulnerability characteristics of the intermediate zone (i.e. the zone beneath the soil zone and above the saturated zone), and the vulnerability characteristics of the saturated zone. It is considered important to consider the saturated zone to determine the spatial and temporal persistence of contamination within the saturated zone. For the rating of each of these three zones the intrinsic vulnerability and the specific vulnerability can be calculated. The weightings or importance of each of these layers is taken into account by considering the thickness and thus importance of each of these zones. This method also takes into account the influence of preferential flow. ReSiS has been tested on the Coastal Park aquifer.

The existing land use data of the Coastal Park region were examined for testing UGIf and accordingly the UGIf model was revised in order to make it applicable and testable for recharge and pollutant assessment for South African conditions. Three screening level models viz. the attenuation factor model, the leaching potential index model and the ranking index model for specific groundwater vulnerability assessment (for example BTEX) and a simple approach for intrinsic vulnerability of conservative contaminants were selected and incorporated in the UGIf model. The preliminary results of the UGIf approach for the Coastal Park area, using literature based input data, shows a consistent pattern of vulnerability of groundwater to organic pollution. A comparison of results from all the four methods reveals that the areas having lesser recharge rates (areas of influent type surface water bodies, and lined pools around the sewage treatment plants) have lower values of vulnerability whereas areas having higher recharge rates (open ground/grass land) have higher vulnerability.
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## Acronyms and Abbreviations

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<tr>
<td>A</td>
<td>Aquifer media</td>
</tr>
<tr>
<td>AF</td>
<td>Attenuation Factor</td>
</tr>
<tr>
<td>BTEX</td>
<td>Benzene, Toluene, Ethylbenzene, and Xylene.</td>
</tr>
<tr>
<td>C</td>
<td>Conductivity (hydraulic)</td>
</tr>
<tr>
<td>CSIR</td>
<td>Council for Scientific and Industrial Research</td>
</tr>
<tr>
<td>CVR</td>
<td>Chemical Vulnerability Rating</td>
</tr>
<tr>
<td>D</td>
<td>Depth to groundwater</td>
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<tr>
<td>DRASTIC</td>
<td>Vulnerability Assessment Method</td>
</tr>
<tr>
<td>EMC</td>
<td>Event Mean Concentration</td>
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<tr>
<td>GEOSS</td>
<td>Geohydrological and Spatial Solutions (Pty) Ltd</td>
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<tr>
<td>GIS</td>
<td>Geographical Information Systems</td>
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<tr>
<td>GIS</td>
<td>Geographical Information Systems</td>
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<tr>
<td>HVR</td>
<td>Hydraulic Vulnerability Rating</td>
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<tr>
<td>I</td>
<td>Impact of the vadose zone</td>
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<tr>
<td>ISCW</td>
<td>Institute for Soil, Climate and Water</td>
</tr>
<tr>
<td>LPI</td>
<td>Leaching Potential Index</td>
</tr>
<tr>
<td>MORECS</td>
<td>Meteorological Office &amp; Rainfall Evapotranspiration Calculation System</td>
</tr>
<tr>
<td>PFM</td>
<td>Preferential flow multiplier</td>
</tr>
<tr>
<td>R</td>
<td>Recharge</td>
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<td>ReSIS</td>
<td>Recharge/Soil/Intermediate/Saturated zone index vulnerability method</td>
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<tr>
<td>RI</td>
<td>Ranking Index</td>
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<td>S</td>
<td>Soil media</td>
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<td>T</td>
<td>Topography</td>
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<tr>
<td>UGF</td>
<td>Deterministic vulnerability method developed by A. Thomas.</td>
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<tr>
<td>US</td>
<td>University of Stellenbosch</td>
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<td>V-RATE</td>
<td>Vulnerability Rating</td>
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<td>Vulnerability Weighting</td>
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<td>Water Research Commission</td>
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<td>ZVM</td>
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1 INTRODUCTION

The Water Research Commission (WRC) awarded a research project to the CSIR and collaborating organizations, entitled “Improved methods for aquifer vulnerability assessments and protocols for producing vulnerability maps, taking into account information on soils”. The project addresses the unsaturated zone (soil and intermediate zones) and the saturated zone. The two cross-cutting components within this project are:

- Geographical Information System (GIS) algorithms for vulnerability assessment and guideline development, and
- A Decision Support Framework.

This report is primarily concerned with the development of GIS based algorithms to predict groundwater vulnerability.

1.1 Background

From a literature survey carried out (Conrad, 2004), two main groups of methods for assessing groundwater vulnerability emerge, namely:

1. Index or subjective rating methods and
2. Statistical and process-based methods.

The first group produces categories of vulnerability (usually high, medium and low), for water resource decision-makers, typically using overlay techniques, with the most widely used index method being the DRASTIC method (Aller et al., 1987). Some subjective hybrid methods do not rely on preconceived scoring systems (such as DRASTIC) but instead project specific categorizations.

The second group produces final products developed by scientists (such as probabilities of exceeding target concentrations), and additional work/research is required to make the assessments useful to decision-makers. These statistical and process-based methods range from statistical to deterministic or process-based approaches that do not produce subjective categorizations but endeavor to simulate or account for physical processes of water movement incorporating several predictor variables to assess the fate and transport of contaminants in the environment.

Uncertainties in groundwater vulnerability assessments are unavoidable and are derived from model or data-related errors (National Research Council, 1993), as a result of inaccuracies in data or input parameters (or model coefficients). Uncertainties in model output can result from assumptions in the model itself, the wrong choice of models, the input data and numerical approximations. The ‘total’ or ‘simulation’ error of a groundwater vulnerability assessment can be regarded as the sum of model, data, and parameter errors (Loague et al. 1996). The challenge of meeting specific scientific objectives of groundwater vulnerability assessments while minimizing uncertainty with limited resources requires careful considerations in design of the approach and realistic expectations of data and model performance. Quantitative estimates and/or other indications of uncertainty can be incorporated in process-based and statistical

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approaches to groundwater vulnerability assessment, but becomes more difficult to quantify when using a subjective approach.

1.2 Objectives

This report addresses the “GIS’Guideline development” component. The first deliverable of this component (1.1) was a workshop for identification of key study areas for comparing vulnerability assessment methods. The second deliverable (1.2), a Literature Review addressed:

(i) GIS-based vulnerability assessment methods used to date, and
(ii) Related data uncertainty and error propagation.

Following on from the literature survey carried out, the third deliverable (1.3) was to derive GIS-based algorithms to predict groundwater vulnerability, based on selected case studies. The focus of this deliverable is on developing new methods of using GIS to aid aquifer vulnerability assessment.

These new methods must:

- Incorporate the results of the unsaturated zone (soil and intermediate zones) and saturated zone in determining aquifer vulnerability.
- Take into account the fact that data is always likely to be scarce in the practical application of vulnerability assessment in South Africa.
- Incorporate a consideration of uncertainty and error propagation, such that the methods should provide an indication of the confidence level associated with a determination of groundwater vulnerability.
- Be analyzed for sensitivity to changes in all variables that are incorporated.

In this deliverable there is no requirement to generate maps of any particular region, except for the purposes of aiding the development of methods and demonstrating their application.
2 GIS-BASED ALGORITHMS AND APPROACHES TO VULNERABILITY ASSESSMENT

2.1 Main characteristics of GIS-based algorithms

Most groundwater originates as excess rainfall locally infiltrating the land surface, and thus activities at the land surface threaten groundwater quality. GIS-based algorithms (subjective or deterministic) have been used to model the spatial variability in land surface and sub-surface at various scales and data densities (Conrad, 2004). Simple and robust matrices need to be established, which indicate what activities are possible at an acceptable risk to groundwater. Instead of applying universal controls over land or soil use and effluent discharge to the ground, modeling the natural contaminant attenuation capacity of the strata overlying the saturated aquifer in GIS can be used in determining aquifer pollution vulnerability (Foster, 1998).

Index (subjective) methods tend to be the most simple to apply and can be used over a range of spatial scales, but include subjective categorizations with uncertainties that cannot be quantified. Properly designed hybrid methods that combine statistical and deterministic or process-based components and exclude subjective categorizations can provide insights on important processes controlling vulnerability over a range of spatial scales while maintaining objectivity and hence scientific defensibility. On the other hand, a subjective hybrid method that combines results of an objective model with a subjective categorization scheme to produce indexes of vulnerability would lose objectivity and ultimately may not be scientifically defensible.

The soil zone can play a significant role in attenuating chemical concentrations, particularly through processes such as filtration; solution and precipitation; biochemical transformations and volatilization. The soil zone thus needs to be included in groundwater vulnerability assessments due to the likely presence of clay, organic contents and microbial populations, however the soil characteristics also influence the scale of nutrient and pesticide leaching from a given agricultural practice. However it must be noted that in many point sources of contamination the sub-surface contaminant load is applied below the soil zone at the base of excavations, such as pits, trenches, leaking underground tanks and quarries, and the attenuation capacity of the soil zone does not contribute to reducing the overall vulnerability. Detailed studies on the characteristics of the soil zone are being carried out by the University of Stellenbosch and a revised soil classification is being produced according the attenuation capacity of South African soils.

The intermediate zone is very important in assessing groundwater vulnerability. Not only is it strategically placed, between the ground surface and the saturated zone, it is most effective in pollution attenuation and even elimination as vadose zone water movement is typically slow and restricted to the smaller pores with larger specific surface. In the intermediate zone there is significant potential for:

- Interception, sorption and elimination of pathogenic bacteria and viruses;
- Attenuation of heavy metals and other inorganic chemicals, through precipitation (as carbonates or hydroxides), sorption or ion exchange;
- Sorption and biodegradation of many hydrocarbon and synthetic organic compounds.
However, water movement in the intermediate zone is complex and its ability to attenuate pollution difficult to predict. Marked changes in the behavior of some pollutants can occur if the polluting activity has sufficient organic or acidic loading to bring about significant change in the Eh or pH of the zone. In the case of persistent mobile pollutants the vadose zone merely introduces a large time delay before arrival at the saturated zone, without any significant attenuation occurring.

There are opposing schools of thought concerning the saturated zone and its role in vulnerability assessments. Some consider the natural mobility and persistence of pollutants in the saturated zone should be considered. Others state that vulnerability assessments should only consider the zone between the ground surface and the saturated zone as important, the objective being to provide planners with the information that will ideally try and prevent pollution from even reaching the saturated zone. If viewed from a risk perspective, it is considered in this project, the view is that the attenuation capacity of the saturated zone should be taken into account, as it will provide an indication of the persistence of a pollutant in time and space and this is valuable information in assessing groundwater vulnerability and associated protection measures. The CSIR is carrying out detailed studies on the saturated zone in the context of groundwater vulnerability.

In most situations the degree of contaminant attenuation will be largely dependent on the intermediate zone pollutant pathways and residence times. While natural flow rates for most porous formations do not exceed 0.2 m/d, when averaged over long time periods, in the presence of preferential pathways and fractured formations, flow rates may be more than an order of magnitude higher. Thus accelerated flow rates due to preferential flow paths need to be taken into account, particularly where microbial, biodegradable and readily retarded contaminants are concerned. This preferential flow factor is considered significant for all three zones and is being taken into account in this project.

The GIS-based method to be developed must be scaleable with respect to data availability and data scarcity. National data sets at spatial scales of 1:1M can be used for the three zone method in the absence of field data, however the recommended approach is to assess vulnerability at a site specific scale. The levels of uncertainty increase with the coarsening scale of input data and the relationship between data scale and data certainty is shown in Figure 1. If finer resolution data sets are available for the area being studied then improved levels of data certainty will be achieved. For detailed site assessments the scale resolution will be 1:10k or finer and this will result in a relatively high level of certainty.
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Figure 1. The relationship between data scale and certainty.

Scientists can provide water-resource decision makers scientifically defensible information for the assessment of groundwater vulnerability. Uncertainties in the assessment can be elucidated either quantitatively or qualitatively, and the scientific defensibility and ultimate usefulness of the product will increase. Science objectives should be clearly distinguished from water-resource management objectives. Ultimately, successful groundwater vulnerability assessments blend scientifically defensible analyses used to meet science objectives with additional interpretations by water-resource decision makers to meet management or policy objectives (Conrad, 2004).

In the South African context, groundwater occurs predominantly in fractured aquifer settings, with a low percentage of groundwater occurring in primary porosity type settings. Thus it is important to take into account the degree of fracturing in assessing groundwater vulnerability. This is a difficult factor to quantify in an indexed GIS approach, due to the large number of associated variables and it can be accommodated for in such a method as a qualitative estimation, based on specialist opinion. Deterministic GIS-based algorithms may be better suited to calculate the effect of fracturing, should the extent of fracturing be quantifiable.

2.2 Selected GIS-based approaches to vulnerability assessment

GIS systems have become widely used in many areas (Conrad, 2004). A GIS system is well suited to carrying out analysis of spatially referenced layers of information. The ability to convert data layers into cell-based data sets facilitates the rapid achievement of a final result of a groundwater vulnerability assessment. It is relatively straightforward to compile vulnerability maps combined with other infrastructural information so as to be easily referenced and understood by planners and decision makers.

In the introduction the two main different approaches to vulnerability assessment were briefly discussed. The “index or subjective rating method” is relatively easily addressed within a GIS framework. The cell-based layer approach facilitates the assignment of ratings and weights and rapid achievement of a final result of groundwater vulnerability. This approach also means that the algorithm can easily be repeated as new or more detailed data sets are obtained or if ratings and weightings need to be adjusted as a result of a sensitivity analysis for example. The most well known “index
Improved methods for aquifer vulnerability assessments and protocols for producing vulnerability maps, taking into account information on soils or subjective rating method" is the DRASTIC method. This method is discussed in some detail below and then it is examined how this approach has been revised for the assessment of groundwater vulnerability within the South African context of hydrogeological conditions. GIS has developed to be able to accommodate successful integration with process-based models that simulate physical processes in the environment. Thomas (2001) has developed a desktop GIS (ArcView 3.2) based model for assessing urban groundwater recharge pollutant flux model called ‘UGf’. A comparison of DRASTIC and UGf reveals that the DRASTIC is an index model capable of dealing with intrinsic vulnerability whereas UGf model is a process based model based on analytical approaches which can deal with contaminant specific vulnerability assessments.

Assessments of pollution from organic compounds in urban environments were identified as one of the key research areas for this project. UGf is meant for urban recharge pollutant flux estimation, dealing with volatile organic compounds (BTEX) in urban environments (Thomas, 2001), so it was chosen as one of the GIS based algorithms for modeling pollutant concentrations and travel times of organic contaminants in urban environments. To make it suitable for South African conditions it is being adapted to include various vulnerability assessment methods. It is being tested on the Cape Flats Aquifer, which is a primary unconfined aquifer.

2.2.1 Brief overview of DRASTIC, its capabilities, assumptions, applicability, limitations, and accuracy

The DRASTIC method of Aller et al. (1987) uses the typical overlay technique often applied in subjective rating methods. The DRASTIC approach is based on four major assumptions (Aller et al., 1987):

- The contaminant is introduced at ground surface
- The contaminant is flushed into the groundwater by precipitation
- The contaminant has the mobility of water
- The area evaluated using DRASTIC is 40.5 ha or larger.

The implication of these assumptions is that DRASTIC should not be used for contaminants that do not have the mobility of water, or for point assessment (such as storage tanks). In addition, groundwater conditions in South Africa are dominated by secondary and fracture controlled flow conditions. DRASTIC does not take into account preferential flow paths or fractured systems particularly well. The DRASTIC method takes into account the following factors:

\[
\begin{align*}
D &= \text{depth to groundwater} \\
R &= \text{recharge} \\
A &= \text{aquifer media} \\
S &= \text{soil type} \\
T &= \text{topography} \\
I &= \text{impact of the vadose zone} \\
C &= \text{conductivity (hydraulic)}
\end{align*}
\]

\[ (1) \quad (2) \quad (3) \quad (4) \quad (5) \]

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The number indicated in parenthesis at the end of each factor description is the weighting or relative importance at that factor.

Groundwater vulnerability maps developed using the DRASTIC method have been produced in many parts of the world. In spite of the widespread use of DRASTIC, the effectiveness of the method has met with mixed success. Several comparisons of maps produced with the DRASTIC method to actual groundwater quality data have shown that the DRASTIC maps rarely predict the predisposition of areas to groundwater contamination (Rupert, 2001). Even in instances where positive correlations between DRASTIC maps and groundwater contamination were observed, researchers stated that the point ratings needed to be modified (Rosen, 1994) or that finer resolution GIS data were required (Barrett and Williams, 1989). DRASTIC has also been criticized as only providing relative results. The large number of parameters included is a problem, implying that critical parameters for a specific application are undervalued (IAH, 1994). It also under-estimates the vulnerability of fractured aquifers compared to unconsolidated aquifers (Rosén, 1994).

Some of the shortcomings of the DRASTIC method can be addressed and these have been described in literature. It is well documented that a sound and detailed hydrogeological understanding of the area being studied is necessary. Different vulnerability assessment methods or adaptation of methods are necessary depending on the hydrogeological setting. In a karstic environment, Biondić, (1998); Mádl-Szönyi and Füle (1998); Doerfliger et al., (1999) and Witkowski et al., (2003) all document adaptations made to the DRASTIC method for vulnerability assessment in a karstic setting, based on dominant hydrogeological processes and controls within their study areas. Witkowski et al., (2003) used MODFLOW to quantify some of the parameters (recharge, hydraulic conductivity and groundwater flow velocities). Differing approaches have been followed in primary aquifer conditions (Secunda et al., 1998).

An additional concern is whether an intrinsic or specific vulnerability assessment is to be carried out. The approaches followed differ significantly depending on whether the “internal” geological factors are to be considered, or whether vulnerability assessments are to be carried out for a specific type of contaminant. Another consideration is whether non-point source or point source pollution assessments are to be carried out. From the assumptions outlined by Aller et al., (1987), DRASTIC can only be applied for the non-point source pollution, as DRASTIC is inaccurate in point source assessments.

Bekesi and McConchie (2000) write that the deeper part of the unsaturated zone (between the base of the soil and phreatic surface) has an important role on the fate of contaminants and that this zone is often ignored or oversimplified. They found four factors to be of major significance in controlling vulnerability, namely: the soil, the unsaturated zone, rainfall recharge and the aquifer medium.
2.2.2 Brief overview of UGI{f}, its capabilities, assumptions, applicability, limitations, and accuracy

UGI{f} is a GIS based urban recharge pollutant flux model developed from a PhD research in United Kingdom (Thomas, 2001). The model is written in the Avenue programming language within ArcView GIS (version 3.x) and is primarily meant for the estimation of groundwater recharge pollutant fluxes of specific pollutants viz. BTEX, nitrate and chloride to an urban unconfined, primary aquifer. The following processes (with their calculation method indicated in brackets) are accommodated for by the model:

- Infiltration and runoff (NRCS curve number method);
- Evapotranspiration (UK Meteorological Office and Rainfall Evapotranspiration Calculation System (MORECS) based on Penman-Grindley model);
- Interflow (empirical index approach);
- Volatilization (Henry’s law);
- Sorption (distribution coefficient); and
- Degradation (first order decay).

Details of its implementation in ArcView GIS 3.2 and the applications of this model to the Birmingham (U.K) unconfined aquifer can be found in Thomas et al. (2001; 2006); and Thomas and Tellam (2004; 2005). Figure 2 shows the interface of the UGI{f} model for the assessment of Non-Point Source (NPS) pollutant fluxes of BTEX, in this case for the Coastal Park landfill site in the Cape Flats aquifer.

![Image: Interface of the UGI{f} model for assessing NPS BTEX pollutant fluxes.](image-url)
**Model structure:** The basic structure of the UGIf model is shown in Figure 3. A land use / land cover classification allows the production of a land cover map. To each land cover class attributes are assigned which relate to permeability, runoff and water quality. With meteorological data and the land cover related runoff characteristics, an estimate of ‘potential recharge’ for each of the land use classes can be made, where potential recharge is defined here as ‘actual recharge’ (i.e. the water reaching the water table) plus interflow. An estimate of the potential mass flux (i.e. flux before interflow, evapotranspiration, and reaction) at the water table can also be made using runoff water quality data associated with the land cover classes. Interflow is estimated from geological maps and used to convert the potential recharge estimates into ‘actual recharge’.

The time taken to pass through the unsaturated zone can be estimated using the actual recharge estimates, hydraulic properties related to the geological units, (reversible) sorption properties related to each geological unit, and unsaturated zone thickness as calculated from land surface and water table maps. Using pollutant-related reaction properties and the estimates of time taken to pass through the unsaturated zone, the decay of degrading pollutants can be calculated, and hence the pollutant mass flux at the water table. Solute concentrations in recharge waters, corrected for evapotranspiration where necessary, are also calculated. A simplified flow chart of estimating pollutant mass fluxes reaching the water table is given in Figure 4.

![Figure 3. Structure of UGIf model (drift = superficial deposits).](image)

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**Capabilities and Applicability:** With respect to vulnerability assessments, the UGIf model can be categorized as a process based model using analytical equations and empirical approaches. Its main capabilities are prediction of pollutant concentration and mass fluxes exiting the vadose zone, pollutant transport velocity and travel time in the vadose zone for specific pollutants viz. BTEX. UGIf shows promise for use in providing input for regional groundwater solute transport models; in identifying gaps in knowledge and data; in determining which processes are the most important regarding urban groundwater quantity and quality; in evaluating existing recharge models; in planning, for example in investigating the effects of land use or climate change; and in assessing groundwater vulnerability.

**Limitations:** The model takes into account the principal processes involved, and, as it is incorporated in a GIS, it allows the complexities of spatial heterogeneity to be investigated. A major limitation is the way time is being dealt with. It is assumed that land use and land use-related properties do not vary within the ‘time-slice’ or period being considered by the model. Daily recharge estimations are summed over the user-specified period. Within this period, steady-state conditions are assumed for the movement of water and solutes through the unsaturated zone. Thus, individual recharge pulses are not tracked: residence time in the unsaturated zone is calculated on the basis of the averaged recharge rate, but it is only used, with a delay arising from any sorption, to estimate degradation/decay of the pollutant concentration. Without incurring considerable computation times, it would be difficult to track individual recharge pulses simultaneously.

**Figure 4** Simplified flow chart for estimating pollutant mass fluxes at the water table.
In Non-Point Source pollution assessment, all units of the same land use type are assumed to have the same Event Mean Concentration (EMC) value regardless of their spatial location within the urban area. However, in reality the concentration of pollutants in recharging water will vary depending on the soil type and vadose zone chemistry. Pollution is often related to past human activities, e.g. accumulation of waste material. Sometimes the background concentration in the subsurface may significantly contribute to the pollution. At present, this contribution is not accounted for in the model due to the often lacking information on the background chemistry and heterogeneity of the urban hydrogeological system. Another weakness is the lacking relationship between the water table elevation and recharge.

**Prediction Accuracy:** The different sub-models in UGIf make use of many input parameters (both spatial and non-spatial data) and the accuracy of their predictions is dependent on the assumptions made in each of the sub-models and the accuracy of the input data used.
3 AVAP-GIS based approaches

3.1 The ReSIS layer method

3.1.1 Main Adaptations

From a review of the DRASTIC method, it is considered that there is significant “double accounting” and that the 7 layers can justifiably be reduced to 3 significant layers corresponding to the three zones discussed previously, i.e. the soil zone (S), the intermediate zone (I) (beneath the soil zone and above the saturated zone) and the saturated zone (S). A fourth component must be taken into account and that is the amount of rainfall occurring, and particularly that component of rainfall that becomes groundwater recharge and adds to the saturated zone, termed natural groundwater recharge (Re). An acronym for the revised DRASTIC method can therefore be ReSIS\(^1\).

The model is still based on the same rated and weighted approach as the DRASTIC method, but provision is made for scalability of the data. For each of the input layers above, the recommended input is the site specific data collected by specialists. In the absence of sufficient field data, or in preparation for field visits in determining sampling sites, the model makes provision for the inclusion of coarser resolution data sets. This has the effect of increasing the uncertainty of the model to the level of the coarsest data set. National scale assessments are not recommended. Each of the input layers is described in Appendix A.

3.1.2 Elements of ReSIS layer model

Figure 5 describes the process in assessing groundwater vulnerability using the ReSIS model.

- The Vulnerability Rating (V RATE) is calculated for the Hydraulic and Chemical attenuation per soil, intermediate and saturated zone layer
- The Vulnerability Weighting (V WEIGHT) is then varied according to role played by layer e.g. for underground tanks, soils weight will be low. The weighting is based on the relative importance of each of the three zones.
- A preliminary vulnerability rating is then obtained by applying a recharge factor to each layer (VR PRELIM = (V RATE \* V WEIGHT))
- The final vulnerability rating is obtained by applying a preferential flow rate multiplier (VR FINAL = VR PRELIM \* PFM). This preferential flow multiplier depends on the degree of fracturing and preferential flow paths within the study area.

---

\(^1\) A new term defined by GEOSS, 2005.
Figure 5. Flowchart of ReSIS layer model
3.1.3 Data (input) requirements

The ReSiS model requires the following rated layers:

- Natural recharge (Re)
- Soil zone (S)
- Intermediate zone (I)
- Saturated zone (S).

The data input requirements for each layer are specified in Appendix A. At the core of the ReSiS layer method is the Zone Vulnerability Matrix (ZVM). In the Zone Vulnerability Matrix, a Hydraulic Vulnerability Rating (HVR) and a Chemical Vulnerability Rating (CVR) are assigned per zone, regardless of the scale of the input data.

Specialist input is required to create the HVR based on hydraulic attenuation capabilities of the particular layer (soil, intermediate or saturated zone). The HVR appears to be subjective, but is based on physical characteristics associated with the hydraulic attenuation of the particular zone under investigation. The HVR gives an indication of the intrinsic vulnerability of the particular zone.

In a similar manner, the CVR must be assigned based on the particular contaminant investigated. This rating is based on the chemical attenuation qualities of the zone in question and is strongly related to sorption. By combining the HVR and CVR in proportions described in Table 1, the Vulnerability Rating (V RATE) for the zone can be derived. Appendix B gives an example of the Zone Vulnerability Matrix for the soil zone.

**TABLE 1. COMBINING HVR AND CVR TO CREATE V RATE**

```
<table>
<thead>
<tr>
<th>HVR</th>
<th>Max</th>
<th>Min</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydraulic attenuation</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>(CVR)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Chemical attenuation</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Min</td>
<td>Max</td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>5</td>
</tr>
</tbody>
</table>
```

3.1.4 Quantitative assessment of uncertainty

Due to the fact that the ratings and weights assigned for this model remain subjective, it is not possible to calculate the uncertainty of data and accuracy of the model quantitatively. A qualitative assessment can however be made based on the scale of the data and the experience of the specialists assigning HVR and CVR ratings. Qualitatively, the level uncertainty and accuracy will be associated with the coarsest dataset. A cumulative system can be devised to address the compounded effect of adding the four layers together.
3.2 Revised UGlf Model

3.2.1 Main Adaptations

The UGlf model was revised to make it more suitable for South African conditions. Three new screening level models for vulnerability assessment and a simple approach to assessing intrinsic vulnerability of conservative contaminants were selected and incorporated in the model. Another modification included the editing of the script of land use grid map preparation in order to accommodate for different land use / land cover types which occur in South Africa. Currently this algorithm can write class names or descriptions of 45 types of possible land use units.

3.2.1.1 Selection of suitable screening level vulnerability assessment models

A literature review on vulnerability assessment was carried out to find suitable screening level vulnerability assessment models which could be easily developed using ArcView GIS. Screening level models are relatively simple, easy-to-use, require very little input data and provide a management decision support. Their major areas of application are: 1) management of water resources (regional planning as related to groundwater control); 2) formulation and implementation of regulatory policies (zoning, land use alterations and practices that protect groundwater quality); 3) identification of “hot-spots” and selection of pollution abatement strategies; and 4) design and management of groundwater monitoring programs (Tim et al., 1996). From the literature three widely used screening level models for organic compounds were selected for development in ArcView GIS. These models are: 1) the Attenuation Factor model of Rao et al. (1985), 2) the Leaching Potential Index Model of Meaks and Dean (1990) and 3) the Ranking Index Model of Britt et al. (1992). A brief description of these models with their equations is given below.

The Attenuation Factor Model

In order to facilitate the classification of pesticides according to their groundwater pollution potential, Rao et al (1985) developed the attenuation factor methodology that includes intrinsic properties of pesticides, hydrologic conditions and soil properties. Rao et al. (1985) expressed an Attenuation Factor (AF) index which denotes mass emission of a chemical from the unsaturated zone to groundwater as:

\[ AF = \frac{M_2}{M_1} = \exp \left[ -0.693R_e \frac{Z \lambda}{T_{1/2}} \right] \]

Where

- \( M_1 \) = initial mass of chemical applied at the ground surface;
- \( M_2 \) = mass of chemical exiting the vadose zone;
- \( T_{1/2} \) = the half life period of the chemical; \( \lambda \) = first order degradation rate coefficient for the chemical;
- \( R_e \) = Retardation factor;
- \( Z \) = vadose zone depth;
- \( q \) = net recharge rate obtained from a water balance calculation.

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The retardation factor for volatile organic compound/pesticides can be calculated as:

\[ R_t = 1 + \left( \rho_b K_d + \left( \theta_s - \theta \right) K_d \right) / \theta \]

Where
- \( \rho_b \) = bulk density of the soil;
- \( \varepsilon \) = air-filled porosity;
- \( \theta \) = the volumetric water content of the soil;
- \( \theta_s \) = the saturated water content of the soil on a volume basis;
- \( K_d \) = the partition coefficient for the pollutant in the soil; \( K_d = K_{oc} f_{oc} \) where \( K_{oc} \) is the organic carbon partition coefficient, \( f_{oc} \) fraction of organic carbon content;
- \( K_d \) = the dimensionless value of Henry’s law constant.

Here \( R_t \) includes the effects of soluble-vapor phase distribution, therefore this method is applicable to volatile organic compounds. For non-volatile organic compounds the normal retardation factor equation \( R_t = 1 + \left( \rho_b K_d \right) / \theta \) can be used.

**Leaching Potential Index (LPI) Model**

LPI is a methodology for ranking sites on the basis of their susceptibility to groundwater contamination. This method is a simplification of the one dimensional mass balance equation for the convective transport-dispersion-reaction process of solutes in a homogeneous porous medium. Assuming steady state conditions and negligible dispersion, Meeks and Dean (1990) simplified the mass balance equation to:

\[ \frac{C_2}{C_1} = \frac{M_2}{M_1} = \exp \left[ -\frac{0.693 R_t Z}{q T_{1/2}} \right] \]

The leaching potential index LPI can be calculated as:

\[ LPI = 1000 \left[ \frac{\theta}{q} \right] \left( \frac{0.693}{T_{1/2}} \right) R_t Z \]

Where
- 1000 is a constant that converts the LPI into a practical range. The term within the parenthesis is an indication of the vulnerability of a site. High values indicate greater susceptibility to contamination.

**Ranking Index Model**

The Ranking Index model is a methodology developed by Britt et al. (1992) for streamlining the pesticide registration and approval program of the Florida Department of Agricultural and Consumer Service in the USA. The ranking index (RI) for a chemical denotes the vulnerability to groundwater contamination by that compound. RI is expressed as:

\[ RI = \left[ \frac{0.693 R_t Z}{q T_{1/2}} \right] \]

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This model requires the setting up of a threshold value for RI (e.g. 500); thus a chemical with an RI of 500 or higher for a particular site was considered for registration. If the RI is less than 500, then a complete analysis involving studies on leaching, adsorption/desorption, hydrolysis, soil dissipation, and groundwater monitoring was required for registration.

3.2.1.2 Vulnerability of Conservative Contaminants
Assessment of the intrinsic vulnerability of conservative contaminants can be carried out based on the evaluation of vertical travel time from the land surface to the aquifer. The travel time through the vadose zone can be calculated using the simple formula:

$$T_{time} = \frac{z \theta}{V_d}$$

Where
- $T_{time}$ = travel time (years);
- $z$ = vadose zone depth (m);
- $\theta$ = average moisture content or volumetric water content; and
- $V_d$ = average recharge rate (m/day).

Witkowski and Kowalczyk (2004) carried out an assessment of groundwater vulnerability for conservative contaminants in Poland using a similar approach (one of their equations matches this approach).

3.2.1.3 Implementation of Vulnerability Assessment Models within ArcView GIS
All of these four vulnerability assessment models were programmed within ArcView GIS using the Avenue programming language. In order to implement these models within UGIf, two Avenue scripts were written; the first one implemented the three screening level models in one model run. The script has to be run separately for each compound of interest. The second Avenue script implemented the vulnerability assessment model for a conservative contaminant, for example, chloride.

3.2.2 Data Requirements and Outputs
The input required for modeling direct recharge and NPS pollutant fluxes in recharge using UGIf is the following:

- Meteorological data (rainfall, evapotranspiration and soil moisture deficits);
- Land use/land cover map with event mean concentration attributes;
- Hydrologic soil group map;
- Geological map with hydraulic and geochemical attributes; and
- Topographic and water table depth data in grid form.

The three screening level algorithms of UGIf require a combined grid containing attributes of average recharge rate (m/day), soil moisture or volumetric water content, vadose zone depths (m), and the retardation factor values. While running the programme the user has to input the half life period of a given organic compound. The input required for a conservative contaminant is a combined grid containing attributes of average recharge rate (m/day), soil moisture or volumetric water content and vadose zone depths (m).
Standard model outputs include:
- Distribution of surface runoff;
- Cumulative infiltration;
- Potential recharge;
- Ground level slope;
- Interflow;
- Actual recharge;
- Pollutant fluxes in surface runoff;
- Travel times of each pollutant through the unsaturated zone; and
- Pollutant fluxes and concentrations at the water table.

Testing the recharge estimation model of UGIf under South African conditions requires meteorological data in a format shown in Table 2 (similar to the output from UK MORECS). As this sort of data is not readily available in South Africa, this model needs further modification to estimate evapotranspiration and soil moisture deficit values. The alternative is to make use of other, existing, models for evapotranspiration estimation or programming in Excel. Apart from the model input data generation task, the use of UGIf for vulnerability assessments also required other modifications (e.g. inclusion of analytical models).

**Table 2. Sample meteorological input file needed for recharge estimation using UGIf.**

<table>
<thead>
<tr>
<th>DAY</th>
<th>RAINFALL</th>
<th>POT_EVAP</th>
<th>ACT_EVAP</th>
<th>SMD</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0</td>
<td>1.5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>0</td>
<td>0.5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>2.4</td>
<td>2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>0</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>19.4</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>0</td>
<td>2.5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>6.8</td>
<td>1</td>
<td>1</td>
<td>&gt;1</td>
</tr>
<tr>
<td>8</td>
<td>19.2</td>
<td>1</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>9</td>
<td>14.8</td>
<td>1</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>10</td>
<td>29.7</td>
<td>0.5</td>
<td>0.5</td>
<td>0</td>
</tr>
<tr>
<td>11</td>
<td>9.2</td>
<td>1</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>12</td>
<td>1.4</td>
<td>1.6</td>
<td>1.6</td>
<td>0.2</td>
</tr>
</tbody>
</table>

### 3.2.3 Reliability of Assessment

Testing the UGIf model for a particular region in South Africa requires a variety of spatial and non-spatial inputs. Testing the UGIf model for a particular site, for example a portion of the Cape Flats area (e.g. Coastal Park area), requires the availability of local data. Scarcity of input data (e.g. land use information, evapotranspiration and soil moisture deficit data, hydraulic properties and geochemical parameters like fraction of organic carbon, etc.) limits testing and applicability of the model. Environmental models are simplified representations of real systems, and uncertainty is always associated with their representations. In many cases the systems, especially urban groundwater systems, are heterogeneous, where a wide range of parameters with a wide range of possible values for them control the complex behavior of the system. In the case of recharge and solute transport simulations of urban environments, the
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Hydraulic and transport parameters are generally not known in sufficient detail. Most of the input parameters in the present GIS based urban pollutant flux models are derived from literature values and therefore predictive runs and the results obtained from them are subject to much uncertainty in relation to the complex heterogeneity of the urban system being modelled. There may also be additional uncertainty relating to whether the conceptual model with simplified analytical equations is fully applicable to the field situation in an urban area (Thomas, 2001).
4 DEMONSTRATING THE AVAP-GIS BASED APPROACHES AT COASTAL PARK.

4.1 Physiography of Coastal Park

Since the focus of this project is on urban catchment areas, Coastal Park on the Cape Flats has been selected as one of the study sites. This is an important urban industrial area ranked as a highly contaminated area that will provide a high concentration of pollution sources from industry, urban settlements as well as agriculture. These pollution sources have occurred in the Cape Flats already for some time.

It is very important that the project covers the diversity of South African conditions and Coastal Park provides a site with a diversity of soils, which will allow model calibration for a variety of conditions in order to increase the representativity of the study. Unfortunately Coastal Park will not provide a range of groundwater vulnerability conditions. The sandy geology and shallow water tables make this site very vulnerable to pollution. This site can therefore represent the extreme case of vulnerability. Logistically Coastal Park is ideally situated, easily accessible from the University of the Western Cape and the University of Stellenbosch. It is envisaged that the research in the Coastal Park will facilitate and improve management of pollution problems in the area.

4.2 Preliminary results of the ReSIS layered approach

The ReSIS layered model was tested on available Coastal Park data. The inherent uncertainty of the model was addressed through the spatial resolution of the input data. From Prof Martin Fey's work, a Zone Vulnerability Matrix was developed for the soil zone (see Appendix B) and applied to Coastal Park. The other parameters required for the layered approach were extracted from GRAII data (intermediate zone, saturated zone and natural recharge). The requirements of these parameters are described in more detail in Appendix A.

4.2.1 Deriving Zone Vulnerability Matrix for the Soil zone

In the absence of site specific data, the 1:250 000 Land Type data set was obtained from the Institute for Soil, Climate and Weather (ISCW). Coastal Park fell predominantly in Land type class Ha7 as can be seen in Figure 6. The memoirs accompanying the Land type data, describe the predominant soil types associated with particular terrain units for Land type Ha7 (Figure 7). Using a 20m DEM, terrain units were derived from topographical data using a method described by Biasi (2001). These are portrayed in Figure 8. The probability of particular soil types occurring in particular spatial units in accordance with the terrain unit were assigned in the proportions described by the Land Type memoirs as seen in Table 3. This allowed assignment of a V RATE for each terrain unit which could be included in the ReSIS layer model. The derived V RATE for each terrain unit is given in Table 3.
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**Figure 6.** Predominant Land Types and DEM derived terrain units in the area of Coastal Park

**Figure 7.** Land type Ha7 terrain unit profile

**Table 3.** Soil types assigned to terrain units for Coastal Park translated into vulnerability rating

<table>
<thead>
<tr>
<th>Terrain Unit</th>
<th>Percentage</th>
<th>Soil Type</th>
<th>HVR</th>
<th>CVR</th>
<th>V Rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>3</td>
<td>3</td>
<td>Imperfectly drained soils, often shallow and often with a plinthic horizon</td>
<td>4</td>
<td>3</td>
<td>4.82</td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>Wetlands</td>
<td>5</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>92</td>
<td>Excessively drained sandy soils</td>
<td>5</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td></td>
<td>7</td>
<td>Excessively drained sandy soils</td>
<td>5</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>8</td>
<td>Lithosols (shallow soils on hard of weathering rock)</td>
<td>2</td>
<td>3</td>
<td>3.38</td>
</tr>
<tr>
<td></td>
<td>20</td>
<td>Imperfectly drained soils, often shallow and often with a plinthic horizon</td>
<td>4</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>40</td>
<td>Wetlands</td>
<td>5</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>25</td>
<td>Excessively drained sandy soils</td>
<td>5</td>
<td>5</td>
<td></td>
</tr>
</tbody>
</table>
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**Figure 8.** Terrain units derived from 20m DEM for Coastal Park

### 4.2.2 Vulnerability assessed using the ReSIS method

In the same manner a V RATE was derived for the intermediate zone and the saturated zone. Natural recharge was rated based on the coefficient of variation of annual precipitation (Schulze, 1997). Table 4 describes ratings and weights applied for Coastal Park. A confidence rating has been designed based on the scale of the data. In this particular exercise, preferential flow paths were not considered since the study area is greatly modified. Figure 9 shows the groundwater quality measured at the study site against the backdrop of the derived vulnerability ratings.

**Table 4. ReSIS vulnerability rating for Coastal Park**

<table>
<thead>
<tr>
<th>Layer</th>
<th>V RATE</th>
<th>V WEIGHT</th>
<th>V</th>
<th>Scale/ Confidence</th>
</tr>
</thead>
<tbody>
<tr>
<td>R – Natural recharge</td>
<td>12.5</td>
<td>-</td>
<td>12.5</td>
<td>1km²</td>
</tr>
<tr>
<td>S – Soil zone</td>
<td>4.0</td>
<td>5.0</td>
<td>20.0</td>
<td>1.250K</td>
</tr>
<tr>
<td>I – Intermediate zone</td>
<td>4.5</td>
<td>5.0</td>
<td>22.5</td>
<td>1km²</td>
</tr>
<tr>
<td>S – Saturated zone</td>
<td>4.5</td>
<td>4.0</td>
<td>18.0</td>
<td>1km²</td>
</tr>
<tr>
<td>TOTAL</td>
<td></td>
<td></td>
<td>73.0</td>
<td>Low</td>
</tr>
</tbody>
</table>

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Figure 9. Groundwater quality (EC) against ReSIS vulnerability

From a visual inspection of Figure 9, there is no clear correlation between the groundwater quality, as indicated by EC and the soil zone vulnerability. This is attributable to the large degree of modification on the site due to the presence of the landfill i.e. the presence of landfill liners, drainage channels, roads, slopes etc. Due to this large degree of site modification it is not possible to directly correlate intrinsic vulnerability to groundwater chemistry as represented by EC.
4.3 Preliminary results of the revised UGIf approach

The revised UGIf vulnerability algorithms were tested on Coastal Park data and preliminary results obtained for benzene are shown in Figures 10, 11 and 12. The predicted values are not the true representation as some of the model inputs, especially hydraulic property values are assumed values based on literature. Comparison of the attenuation factor values and leaching potential index values shows that the results look more or less similar in their spatial distribution. The pools around the sewage treatment plant (influent type having a small leakage rate because of lining) have lower values in both maps indicating lesser vulnerability values or zones mainly because of lesser amount of recharge from these areas. The open ground/grass land areas have higher attenuation factor values and leaching potential index values indicating higher vulnerability in those areas because of higher recharge in these areas. The residential areas have medium vulnerability in both maps, which also corresponds to the medium recharge rates in these areas. A comparison of the ranking index map with that of the attenuation factor map and the leaching potential index also reveals the same behavior. Comparison of the travel time of a conservative contaminant (nitrate; Figure 13) with the previous maps also indicates a more or less similar distribution of vulnerability. Preliminary results thus suggest that for all four predictions there is a consistent pattern of vulnerability of groundwater to organic pollution at the Coastal Park landfill site.

Figure 10. Attenuation Factor Values for Benzene in Recharge Waters of Coastal Park.
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Figure 11. Leaching index potential values for Benzene in Recharge Waters of Coastal Park.

Figure 12. Ranking Index for Benzene in Coastal Park.
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The UGlf model requires a substantial amount of input data: slope, soil texture, geological map with lithological descriptions and hydraulic property values (porosity, bulk density, specific retention, presence of clay (clay index), horizontal and vertical hydraulic conductivity values, fraction of organic carbon content, half lives of BTEX compounds, the thickness of the vadose zone or depth to the water table etc. All the three research teams of AVAP use some of these parameters for their studies and their inputs and results can be successfully used for modeling pollutant fluxes and vulnerability with UGlf. For example, the UWC team is currently working on data collection through field work and lab analysis for porosity, water retention, bulk density, volumetric water content, hydraulic conductivities etc. These results can be the best input values for the chosen case study areas. The UWC team is also working on estimating recharge using the chloride mass balance approach, which could be compared with the predictions from the UGlf. The saturated zone team (CSIR) will include organic contamination in their approach and their input values of geochemical parameters (half lives, density, organic carbon content, etc.) can also be used in modelling with UGlf. The soil team is working on sorption properties and the resulting attenuation of pollutants. The fraction of organic carbon content in soils can also be uses in modelling with UGlf.

**Figure 13. Groundwater vulnerability for conservative contaminants in Coastal Park.**

### 4.3.1 Incorporation of Research Findings from Various Research Teams of AVAP

The UGlf model requires a substantial amount of input data: slope, soil texture, geological map with lithological descriptions and hydraulic property values (porosity, bulk density, specific retention, presence of clay (clay index), horizontal and vertical hydraulic conductivity values, fraction of organic carbon content, half lives of BTEX compounds, the thickness of the vadose zone or depth to the water table etc. All the three research teams of AVAP use some of these parameters for their studies and their inputs and results can be successfully used for modeling pollutant fluxes and vulnerability with UGlf. For example, the UWC team is currently working on data collection through field work and lab analysis for porosity, water retention, bulk density, volumetric water content, hydraulic conductivities etc. These results can be the best input values for the chosen case study areas. The UWC team is also working on estimating recharge using the chloride mass balance approach, which could be compared with the predictions from the UGlf. The saturated zone team (CSIR) will include organic contamination in their approach and their input values of geochemical parameters (half lives, density, organic carbon content, etc.) can also be used in modelling with UGlf. The soil team is working on sorption properties and the resulting attenuation of pollutants. The fraction of organic carbon content in soils can also be uses in modelling with UGlf.
Improved methods for aquifer vulnerability assessments and protocols for producing vulnerability maps, taking into account information on soils

4.3.2 Sensitivity of the UGif Model

The UGif model has many sub-models. Each sub-model deals with a specific environmental problem and it has its own controlling parameters. Some sub-models, for example the point source and the non point source model, which simulate the fate and transport of BTEX compounds, require many input parameters. Thomas (2001) carried out a sensitivity analysis of the direct recharge model using input parameters for grass land and commercial areas underlain by sandy and clayey soils. The main objective of the sensitivity analysis of the direct recharge model was to determine which parameters in the model are the most important and which are, therefore first priority for future development of the model.

The order of sensitivity of input parameters of the direct recharge model for grasslands and commercial areas underlain by wet sandy and clayey soils are as follows:

For grassland underlain by wet sandy soil

Rainfall > Initial loss ~ Curve number [> Actual evapotranspiration > Clay index > Storage coefficient > Slope > Anisotropy].

For grassland underlain by wet clayey soil

Curve number > Rainfall > Clay index > Initial loss > Anisotropy > Actual evapotranspiration > Storage coefficient [> Slope].

For a commercial area underlain by wet sandy soil

Curve number > Rainfall [> Clay index > Storage coefficient > Initial loss > Slope > Anisotropy].

For a commercial area underlain by wet clayey soil

Curve number > Rainfall > Clay index > Anisotropy > Storage coefficient [> Initial loss > Slope].

NB: Parameter in parenthesis has very little effect.

From the results of the two analyses of grassland areas it is clear that the most important parameters affecting direct recharge from grasslands are curve numbers (CN), initial loss and rainfall amount. Slope and anisotropy have the least effect on recharge from grassland underlain by sandy soils whereas in clayey soils the effect of the clay index and anisotropy cannot be ruled out.

From the sensitivity coefficients obtained for commercial areas, the curve number, rainfall and the clay index are the most sensitive input parameters in areas underlain by both sandy and clayey soils. Under both sandy and clayey soils when the curve number is near 99 or 100, the model becomes extremely sensitive to a change in the CN value. For example, a small decrease of 0.2 in CN value results in a large increase of the recharge value. An increase of anisotropy resulted in a smaller decrease of recharge. The parameters initial loss and slope are not of much concern for both cases. However, the storage coefficient cannot be ruled out in commercial areas underlain by clayey soils. Similarly, anisotropy cannot be ruled out in situations where commercial areas are underlain by sandy soils. A plot of calculated runoff/infiltration against rainfall confirms the sensitivity to the CN (Figure 14).
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Grassland and commercial areas underlain by sandy and clayey soils are examples of two extreme situations. The other land use classes underlain by sandy or clayey soils will have similar sensitivities, but as the curve number or imperviousness increases, the sensitivity to curve number also increases. The results of the sensitivity analysis reveal that for grassland (i.e. less impervious areas) the curve number and the initial loss are more important for the actual recharge than the parameters that control the interflow processes. For the more impervious commercial areas, the curve number and the interflow parameter clay index are more important than other parameters of recharge.

Figure 14. Relationship of runoff/infiltration to rainfall for grassland and commercial area using the NRCS curve method (both land uses are underlain by the same Hydrologic Soil Group).
5 CONCLUSIONS AND RECOMMENDATIONS

Preliminary results from the ReSIS method (index approach) as well as the UGI\textsubscript{f} method (deterministic, process-driven) imply a high vulnerability rating for the Coastal Park area. Data scarcity and the different scales of the input data mean that the assessment results are a first approximation, and could be improved with better data. However, two new approaches to using GIS for vulnerability assessment have been demonstrated for this site.

The ReSIS method follows the approach of rating and weighting three component layers; i.e. the soil zone, the intermediate zone (beneath the soil zone and above the saturated zone) and the saturated zone. The method can be applied to both intrinsic and specific vulnerability assessments. The weightings are based on the relative importance of the various zones. It integrates the research that is being carried out by the project consortium (i.e. soil zone research – University of Stellenbosch; the intermediate zone research – University of the Western Cape; and the saturated zone research – CSIR). It also takes into account preferential flow conditions. The method is scaleable and can be used with site specific data and if this is not available, coarser data sets can be used. The level of uncertainty associated with the final results is also assigned, thereby assisting decision makers.

The UGI\textsubscript{f} model generally requires more data, but is useful in making fine distinctions in vulnerability for different land uses, and is particularly suited for assessing aquifer vulnerability to volatile organic compounds. The modified UGI\textsubscript{f} has three screening level vulnerability assessment models and models for the assessment of the final concentration exiting the vadose zone (after accounting for sorption, volatilization and first order decay) and the travel time of reactive and conservative pollutants. The UGI\textsubscript{f} model is applicable for both contaminant specific vulnerability (especially volatile organic compounds like BTEX) and intrinsic vulnerability of conservative contaminants like chloride. The vulnerability algorithms in the UGI\textsubscript{f} model are helpful for regional planning as related to groundwater use (by identifying areas of good quality water zones and higher recharge rates) and identification of ‘hot spots’ of vulnerability zones. If decision makers need such information for the management of groundwater resources then UGI\textsubscript{f} is quite a good choice and can be useful for management decision support.

Future work on the ReSIS model will include:

- Drawing up Zone Vulnerability Matrices for each zone.
- Determining reliable weighting formula.
- Automating scalability of input data sets in GIS.
- Validation and sensitivity testing.

Future work on UGI\textsubscript{f} model application will include:

- Preparation of input data such as evapotranspiration, soil moisture deficits and collation of chemical data (EMC data) for the Coastal Park area.
- Demonstration of the UGI\textsubscript{f} model for the assessment of recharge pollutant fluxes and vulnerability for the Cape Flats aquifer (Coastal Park area).
- Incorporation of a suitable model for vulnerability assessment of fractured aquifer.
- Collation of information/data for Secunda area, preparation of model inputs and demonstration of the model, and
- A sensitivity analysis of the revised model.
6 REFERENCES


Conrad, J.E. 2004. Literature review on GIS-based vulnerability assessment methods used to date and related data uncertainty and error propogation. Deliverable 1.2. WRC project “Improved methods for aquifer vulnerability assessments and protocols for producing vulnerability maps, taking into account information on soils” (K5/1432).


Improved methods for aquifer vulnerability assessments and protocols for producing vulnerability maps, taking into account information on soils


**WRC Project: K5/1432**
Improved methods for aquifer vulnerability assessments and protocols for producing vulnerability maps, taking into account information on soils

WRC Project: K5/1432


Appendix A: (Input data sets for ReSIS Layer model)
INPUT DATA SETS FOR THE ReSIS LAYER MODEL

1. Recharge

Groundwater recharge termed natural groundwater recharge based on effective rainfall as described by Vegter (1995). Physically defined by classifying coefficient of variation of annual precipitation (Schulze, 1996) into 25 distinct classes. This gives a subjective rating to the amount of recharge that can be expected. The groundwater recharge data set as defined by GRAI I can also be used, again reclassified into 25 classes in the case when no data is available.

![Figure A1. Reclassified Coefficient of Variation of Annual Precipitation](image1)

![Figure A2. Reclassified Recharge from GRA II](image2)
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2. **Impact of the Soil zone:**

See Appendix B for the Zone Vulnerability Matrix designed for the soil zone used as guideline for assigning ratings.

Recommended input:
- Site evaluation
- Use Land Type memoirs to select sampling sites

Low-confidence input:
- ISCW 1:250 000 Land Types assigned to terrain units derived from DEM modeling rated using Zone Vulnerability Matrix

3. **Impact of Intermediate zone**

Recommended input:
- AVAP_unsat_2005.xls

Low confidence input:
- Combined
  - Depth to water table (GRA II if required)
  - Geology rated using Zone Vulnerability Matrix

4. **Impact of the Saturated zone**

Recommended input:
- AVAP relative saturated zone vulnerability
  - persistence in time of contamination
  - spatial impact (dimensions of contaminated volume)

Low-confidence input:
- Zone Vulnerability Rating for Geology based on hydraulic conductivity
Appendix B: (Zone Vulnerability Matrix – Soil Zone)
### Zone Vulnerability Matrix

<table>
<thead>
<tr>
<th>Chemical attenuation</th>
<th>Maximal</th>
<th>Intermediate</th>
<th>4</th>
<th>5</th>
<th>Minimal</th>
</tr>
</thead>
<tbody>
<tr>
<td>A. CATIONIC contaminants (inorganic &amp; polar organic)</td>
<td>1</td>
<td>Thick, clayey profiles esp. margalic soils; strongly calcareous clays; eutrophic peats</td>
<td>Mostly calcareous and eutrophic clay soils; duplex and margalitic soils; lithocutanic soils with steeper relief</td>
<td>Mostly loamy, thicker eutrophic or mesotrophic soil profiles on gentler relief</td>
<td>Extreme water surplus sustained for significant periods; sandy soil texture; absence of luvis or clay pan features in soil profile+ vadose zone; regic sands of humid climates on level topography</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>All other soils</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>All other soils</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>All other soils</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>Dystrophic sands low in humus</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>B. ANIONIC contaminants (inorganic &amp; polar organic)</td>
<td>1</td>
<td>Deep, dystrophic, ferralic clays</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>Deep, dystrophic, ferralic clays</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>Deep, dystrophic, ferralic clays</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>Deep, dystrophic, ferralic clays</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>Eutrophic sands</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>C. ORGANIC contaminants (non-polar)</td>
<td>1</td>
<td>Deep humic clays and peats</td>
<td></td>
<td></td>
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<td>Deep humic clays and peats</td>
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<td></td>
<td>5</td>
<td>Eutrophic sands low in humus</td>
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</tr>
</tbody>
</table>

**WRC Project: K5/1432**
APPENDIX C: (UGIF MODEL DESCRIPTION)
URBAN GROUNDWATER RECHARGE POLLUTANT FLUX (UGIf) MODEL

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C.1 Model Overview

Using ArcView GIS version 3.2 and its extension Spatial Analyst, an ‘urban groundwater recharge pollutant flux’ (UGIf) model has been developed for estimating recharge and pollutant fluxes to an urban unconfined aquifer system (Thomas, 2001; Thomas and Tellam, 2005). With this model, using meteorological data, land use and geological maps together with attribute tables covering chemical characteristics and reaction constraints, surface water runoff, groundwater recharge and chemical fluxes (currently a few chosen pollutant species such as nitrate, chloride, and BTEX compounds) in recharge waters can be calculated and various maps displaying the distribution of these parameters can be generated. The major outputs from the model are distribution of surface runoff, infiltration, potential recharge, loss of recharge water due to interflows, actual recharge in an urban environment, pollutant concentration and flux distribution, and travel times of each chosen pollutant through vadose zone. In this model, the estimation urban recharge and pollutant fluxes are performed though eight sub models. These models may be grouped under three types of major models viz. 1) recharge models, 2) non-point source pollutant load/flux models and 3) point source pollutant flux models. Within each of the major type, there are specific sub models, each of which deals with a specific type of recharge or pollutant. The model has been demonstrated for an urban unconfined aquifer system in Birmingham, United Kingdom (Thomas and Tellam, 2005).

Recharge Models:
- Runoff – direct recharge model supported by a soil moisture balance model / MORECS data;
- Indirect recharge through leaks from sewer network
- Indirect recharge through mains leakage

Non point source (NPS) Pollutant Load/Flux Models:
- Non point source pollutant load in runoff (nitrate, chloride, and BTEX)
- Initial\(^1\) NPS recharge pollutant flux model (nitrate, chloride and BTEX compounds)
- Final\(^2\) NPS recharge pollutant flux model (currently only for BTEX compounds)

Point Source Pollutant Flux Models
- Sewer pollution model (nitrate, chloride and toluene)
- Petrol Station BTEX Pollution Model

\(^1\): Estimates starting concentrations in the recharge (before reactive transport in the vadose zone).
\(^2\): Estimates final concentrations reaching to the water table (after reactive transport in the vadose zone).

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C.1.1 Model Input Data

The input data required for the present Runoff – Direct Recharge model and NPS Pollutant Flux models are:
1) the urban land use / land cover map;
2) hydrologic soil group map
3) rainfall amount with or without rain gauge locations,
4) evaporation (potential and actual evapotranspiration)
5) soil moisture deficit
6) the geological map i.e., solid and drift geology map with hydraulic property attributes
7) elevation map
8) vadose zone depth map or map of depth to water table

C.1.2 Model Output

The major outputs from the present model are: surface runoff distribution; distribution of cumulative infiltration, potential recharge, slope, anisotropy (ratio of horizontal and vertical hydraulic conductivity), lateral flow, actual recharge, initial pollutant loads in surface runoff, initial pollutant concentration and fluxes in the recharge; travel times of each chosen pollutant through vadose zone; and the final amounts of pollutants and final pollutant concentration reaching the water table.

C.2 Design of GIS Based Direct Recharge Model

Groundwater recharge is the process by which aquifers are replenished with water from the surface. This process occurs both artificially by anthropogenic sources and naturally as part of the hydrologic cycle as infiltration when rainfall infiltrates the land surface and as deep percolation of water into underlying aquifers. In urban environments recharge of groundwater occurs from various sources such as precipitation (direct recharge); rivers, canals and lakes (indirect recharge); and from man made activities such as irrigation and urbanisation (indirect recharge through man made drainage systems, water systems, and sewer systems). Development of a GIS based recharge model, which could account for all sorts of recharge in an urban area, is a challenging task because of the complexities involved in the urban set up. However attempts were made in Birmingham, UK to develop GIS models for estimating direct or precipitation recharge, indirect recharge through seepage from surface water bodies, indirect recharge through mains leaks and indirect recharge through sewer leaks in a city on a regional scale (Thomas, 2001).

C.2.1 The Mechanisms of Runoff, Infiltration and Groundwater Recharge

Runoff is what occurs when rain is not absorbed by the ground on which it falls and so then flows downhill. The amount of rainwater that runs off during/immediately after a rainfall event depends heavily on the amount of rainfall, ‘initial abstraction’ (i.e., initial loss as due to interception), and the type and condition of ground it lands on (i.e.
infiltration characteristics of the soil, soil moisture, antecedent rainfall, impervious surface etc). The most important factor in determining the quantity of runoff that will result from a given storm event is the percent imperviousness of the land cover. Other factors include soil infiltration properties, topography, vegetative cover, and prevailing site conditions (US EPA, 1993).

The conceptual basis for the estimation of direct recharge is shown in Figure 1. Infiltration is the initial process of water entering the soil at the ground surface from precipitation or anthropogenic sources (US EPA, 1998a). Infiltration plays a major role in runoff. Infiltration is a direct loss that governs the volume and rate of runoff, and thus it controls the shape of the runoff hydrograph (Tindal et al., 1999). Infiltration depends on the type of land use, soil type (texture class), vegetative cover, porosity and hydraulic conductivity, degree of soil saturation (moisture content), soil stratification, drainage conditions, depth to water table, and intensity and volume of rainfall. The amount of rainwater that runs off during / immediately after a rainfall event depends heavily on the amount of rainfall, ‘initial abstraction’ (i.e., the initial loss as due to interception storage, depression storage, and surface storage), and the type and condition of ground it lands on (i.e. the infiltration characteristics of the soil, soil moisture, impervious surface etc).

![Diagram](image.png)

*Figure 1. Runoff Recharge Processes.*

NB: The term ‘initial abstraction’ (I₀) incorporates rainfall loss due to interception, depression and detention storage (not abstraction from groundwater).

After having infiltrated into the soil / vadose zone, the infiltrated water is subjected to redistribution in the vadose zone where a part of the infiltrated water may be lost to the atmosphere through evaporation and / or transpiration processes. The amount of infiltrated water left behind after the evaporative loss can be called potential recharge, which may again lose a certain portion through subsurface lateral flow, also called interflow. The amount of water infiltrated left behind after all these losses is available as actual recharge.
Groundwater recharge is the process by which aquifers are replenished with water from the surface. This process occurs both artificially by anthropogenic sources and naturally as part of the hydrologic cycle as infiltration when rainfall infiltrates the land surface and as deep percolation of water into underlying aquifers. Recharge of groundwater occurs from various sources such as precipitation (direct recharge); rivers, canals and lakes (indirect recharge); and from man made activities such as irrigation and urbanisation (indirect recharge through man made drainage systems, water systems, and sewer systems). The recharge distribution in each part of an aquifer is dependent on the land use and the infiltration properties of the underlying soil hydrologic units.

C.2.2 Quantification of Groundwater Recharge Through Modelling

Quantification of groundwater recharge through field measurements is a fairly difficult task and it can be estimated using a model. Under this project GIS based models for estimating direct or precipitation recharge and indirect recharge through seepage from surface water bodies (streams), will be developed for assessment of recharge on a catchment scale. The model needs to be sophisticated enough to account for the routing of all precipitation in the form of surface runoff, infiltration, evaporation, interflow and direct recharge, and other recharge source water through leakage as well. In order to estimate it, some suitable method for estimating the areal distribution of rainfall losses through infiltration and runoff has to be chosen first when designing a recharge model. Possible models that can be adopted are the following: the rational method, SCS CN method, Horton’s model for infiltration capacity, Green-Ampt infiltration model etc.

From the previous discussion on recharge mechanisms, it becomes clear that the direct recharge (direct potential recharge) calculation procedure needs information on areal precipitation, infiltration, runoff, evaporation (potential evaporation and actual evaporation) and soil moisture status. Direct recharge in any location can be quantified through the following equations:

\[
\text{Infiltration} = \text{Rainfall} - \text{Initial abstraction} - \text{Runoff} \quad \text{(Equation 1)}
\]

\[
\text{Potential Recharge} = \text{Infiltration} - \text{Evaporation} \quad \text{(Equation 2)}
\]

\[
\text{Actual Recharge} = \text{Potential Recharge} - \text{Loss due to Lateral Flow} \quad \text{(Equation 3)}
\]

After considering various available methods for infiltration and runoff estimation, the United States Department of Agriculture, Soil Conservation Service (SCS, now known as the Natural Resources Conservation Service, or NRCS) Runoff Curve Number (CN) method was chosen.

The SCS curve number method is an empirical description of infiltration. It combines infiltration with initial losses (interception and detention storage) to estimate the rainfall excess, which would appear as runoff (Figure 2). This model is relatively simple requiring few input parameters, and has been widely applied in the fields of soil physics and hydrology (US EPA, 1998a). The method is an empirically based one, and is applicable to the situation in which daily amounts of rainfall, runoff, and infiltration are of interest (US EPA, 1998b).
The USDA NRCS curve method predicts direct surface runoff using the following equation:

\[ Q = \frac{(P - I_a)^2}{(P - L_o) + S} \]

*Equation (4)*

in which:

- \( Q \) = Total rainfall excess (runoff) for storm event (inches)
- \( P \) = Total rainfall for storm event (inches)
- \( I_a \) = Total initial loss or “initial abstraction” (inches)
- \( S \) = Potential maximum retention capacity of soil at beginning of storm or maximum amount of water that will be absorbed after runoff begins (inches).

\( S \), also called the retention parameter, is a statistically derived parameter related to the initial soil moisture content or soil moisture deficit (US EPA, 1998a). The value of \( S \) is determined based on the type of soil and the amount and kind of plants covering the ground (cover types). This is derived through its relationship to the value of the NRCS runoff curve number (\( CN \)). A curve number is a numerical description of the impermeability of the land in a watershed. This number varies from 0 (100 % rainfall infiltration) to 100 (0 % infiltration — e.g., road/concrete). The following relation relates the value of \( S \) to the ‘curve number’:

\[ S = \frac{1000}{CN} - 10 \]

*Equation (5)*

\( CN \) = runoff curve number (0-100, based on the soil and land use information).

\( CN \) is determined through several factors. The most important are the hydrologic soil group (HSG), the ground cover type, treatment, hydrologic condition, the antecedent runoff condition (ARC), and whether impervious areas are connected directly to drainage systems, or whether they first discharge to a pervious area before entering the drainage system. Soils are extremely important in determining the runoff curve number. Soils are generally classified into four HSGs (hydrological soil groups: A, B, C, and D) according to how well the soil absorbs water after a period of prolonged wetting.

The term ‘initial surface loss’ incorporates rainfall loss due to interception, depression and detention storage. The value of \( I_a \) depends greatly on the cover types (the kind of plants covering the soil or land use), the kind of soil (hydrologic soil groups, its...
treatment, and hydrologic condition) and antecedent soil moisture of the area being studied. For a given drainage basin, the values of I_a are highly variable, but generally are correlated with soil and cover parameters. A major limitation for applying the SCS model lies in that the values of the parameter I_a must be evaluated with field data for each specific site.

Estimation of potential recharge
After estimating the infiltration and runoff using the curve number method, the potential recharge in the runoff – potential recharge model is quantified using the following equations:

For non-vegetated areas
\[ \text{Potential Recharge} = \text{Precipitation} - \text{Initial Loss} - \text{Runoff} \]  \hspace{1cm} \text{Equation (6)}

For vegetated areas
In the vegetated areas, the calculation procedure takes into account the soil moisture deficit parameter and actual evapotranspiration. When the Soil Moisture Deficit (i.e., water required to bring the soil to maximum water holding capacity) at the end of the day is zero,

\[ \text{Potential Recharge} = \text{Precipitation} - \text{Initial Loss} - \text{Runoff} - \text{Actual Evapotranspiration} \]  \hspace{1cm} \text{Equation (7)}

When Soil Moisture Deficit at the end of the day is greater than zero,
\[ \text{Potential Recharge} = 0 \]  \hspace{1cm} \text{Equation (8)}

Using the Avenue programming language, (the object oriented programming language / code of ArcView GIS) the equations 1 to 8 were programmed in a model using the ArcView GIS platform. The ArcView interface of the direct recharge model is shown in Figure 3. The model predicts runoff, cumulative infiltration and potential direct recharge occurring from different land use / land cover types in Birmingham for each rainfall event or on a day.
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C.3 Estimation of Evapotranspiration

In UK, the prime source of rainfall information is the United Kingdom Meteorological Office’s (UKMO), which records precipitation on daily basis for most gauge stations. The estimation of evaporation is based upon the knowledge of the potential evapotranspiration, available water holding capacity of the soil, and a moisture extraction function. The calculation of potential evaporation and actual evaporation is a complicated and a time consuming process involving the use of other meteorological parameters (such as temperature, sunshine, humidity, wind speed etc.). In United Kingdom Meteorological Office (UKMO) also estimates the evaporation through its Meteorological Office Rainfall and Evaporation Calculation System (MORECS). Since evaporation data is already available, it was decided to use the MORECS data to reduce processing time. The MORECS calculations are performed on daily meteorological data, as this degree of time discretisation is required to best estimate recharge in the GIS model.

C.3.1 Input Data for the Potential Recharge Estimation

The input data for Potential Recharge model are the following:
- Land Use Map (shapefile or grid)
- Hydrologic Soil Group Map (Superficial and Bedrock Geology Map) (shapefile or grid)
MORECS Data (Rainfall, Actual Evaporation and Soil Moisture Deficit) in table form (Table 1; .txt file format).

Table 1. A sample meteorological input file (MORECS) needed for recharge estimation using UGIf model.

<table>
<thead>
<tr>
<th>Day</th>
<th>Rainfall</th>
<th>Pot_Evap</th>
<th>Act_Evap</th>
<th>SMD</th>
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</thead>
<tbody>
<tr>
<td>16</td>
<td>6.8</td>
<td>1</td>
<td>1</td>
<td>&gt;1</td>
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</tbody>
</table>

C.3.2 Structure/Working of Potential Recharge Model

Using land use and soil hydrologic group themes, a map showing the area under various land uses on different hydrologic soil groups is generated by intersecting/combining these two maps with the GIS. A curve number value is assigned for each unit of this map, which leads to the preparation a runoff curve number map. The hydrologic soil group map can be generated by reclassifying the various lithological units (as defined by geological map) based on their drainage potential (textures of the sediments).

Potential Recharge from Surface Water Bodies

Quantifying recharge from surface water bodies (rivers, canals and reservoirs/lakes) is not easy, because the seepage/leakage rate from them is dependent on many factors like the nature/hydraulic conductance of bed material, its thickness, area of water bodies, elevation profile of the bottom or base of bed materials, groundwater head distribution underneath the water bodies, evaporation rates etc. Recharge from surface water bodies occurs when the hydraulic head at the bottom of the surface water body is greater than that of the underlying ground water. The flow in urban rivers is dependent on many factors (disposal of foul and storm sewers, industrial waste disposal etc), the distribution of head gradient will be varied at each point and hence the development of a method for quantification of recharge from urban rivers is challenging.

Due to the paucity of data regarding these controlling parameters and because of the complexities involved in assessing recharge from surface water bodies, a pragmatic approach was adopted for the estimation of recharge rate for a regional scale assessment. In this model, the potential recharge from reservoirs and canals (unlined) is assumed as contributing a constant recharge rate of 2mm and 12.5mm respectively. Recharge from the Tame river to the unconfined through seepage is assumed to be nil because recent studies on groundwater head distribution all along the course of River Tame by Ellis, (2000) reveal that this river is discharging at many locations within the unconfined aquifer part.
C.3.3 Quantification Lateral Flow or Interflow through an Index Model

Lateral subsurface flow (also called interflow or subsurface storm) is the lateral flow of water through soil / vadose zone. Interflow is a primary source of runoff or stream flow and is a key component in water balance. This stream flow contribution originates below the subsurface but above the zone where rocks / soils are continuously saturated with water. Interflow generally occurs in the soil or in the geological deposits or aquifer lying beneath it, and is more prevalent in sloping terrain than on level terrain. The degree of interflow depends largely upon the soil and terrain characteristics such as slope, porosity, storage capacity, hydrologic soil group, average moisture content, effective permeability and anisotropy of the vadose zone, lateral continuity of a perching horizon (boulder clay) etc. Modelling of lateral flow (after taking into account all the above-mentioned processes or properties) is a complex process. Because of the lack of data on soil properties and the heterogeneity of the system in the area, it may be inappropriate to develop a sophisticated interflow model. Therefore, a simplified interflow index model as illustrated in Thomas 2001 can be used, which can account for loss of recharge water on a regional scale in a simple manner.

In this model, the important factors that can control lateral flow within the vadose zone are identified as: i) slope; ii) specific retention (soil storage capacity); iii) anisotropy ratio of the formation (Kv/Kh); and iv) clay presence (presence of boulder clay). Each factor is accounted for in the model by an index. The indices for the first three factors have a direct relationship with the interflow as illustrated in Fig. 4. The fourth index, the clay presence index, is based on the absence and presence of boulder clay underneath the outcropping deposits, and its form (sheet of boulder clay or clay lenses). When calculating lateral flow in this model, a weighting factor is assigned to each of the four interflow indexes so that relative importance of each can be assigned.

The lateral flow \( Q_L \) is thus estimated using the following equation.

\[
Q_L = PR \left( \frac{PW_{SL} \times SL_f + (PW_{SR} \times SR_f) + (PW_{Kv/Kh} \times Kh/Kv_f) + (PW_{CP} \times CP_f)}{PW_{Kv/Kh} \times Kh/Kv_f + (PW_{CP} \times CP_f)} \right)
\]

Equation (9)

where \( PR \) = potential recharge, \( PW \) = percent weight factor or weighting factor (%), \( SL \) = slope, \( SR \) = specific retention, \( Kh/Kv \) = anisotropy ratio, \( CP \) = clay presence and ‘f’ stands for ‘factor’.

Condition: \( \sum PW_{SL} + PW_{SR} + PW_{Kv/Kh} + PW_{CP} = 100\% \)

Equation (10)

The input data of this model are elevation grid, geological map with attributes of specific retention, anisotropy ratio and clay presence, and a potential recharge grid. All calculations in this model use grid data. The output from this model is a spatially distributed interflow (not accumulated from one cell to another cell).
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Field scale measurement and monitoring of water balance components such as rainfall and its initial loss, surface runoff, infiltration, lateral flow, stream flow and recharge can help in validating the model.

Figure 4. Subsurface Lateral Flow Indices Relationship.
C.4 Modelling Non Point Source Pollution in Urban Runoff and Recharge

Nonpoint source (NPS) pollution is an exceedingly complex phenomenon. Nonpoint source pollution may be defined as the introduction of impurities into surface or subsurface water supplies, generally from indirect, intermittent or diffuse sources and often associated with storm, rainfall or snowmelt events (Warrington, 2000). Nonpoint source pollution results from a wide variety of human activities on the land. It represents the cumulative effects of all of the land uses in a watershed and associated human activity. Owing to this complexity, models that try to reflect the actual processes require large quantities of data, which are rarely available. Thus, the most common method of approximating non-point source pollution uses long-term average contaminant loadings for common land uses. This approach is based on the National Urban Runoff Program called NURP (US EPA, 1984). The approach has been followed in many other countries also. The estimation of non point source pollutants in the surface runoff and recharge is based on typical Event Mean Concentrations. EMCs are standardized concentrations of a pollutant expected from a particular land use type. They are assumed to be directly related to the land uses in the watershed and remain constant independently of the duration and intensity of the rainfall events (Naranjo, 1998). Literature based Estimated Mean Concentrations of pollutant constituents associated with land use are available (Lopes and Dionne, 1998; Delzer et al., 1996 & Shepp, D.L., 1996).

C.4.1 Surface Water Pollution Model (‘NPS Load in Runoff’ Model)

Using ArcView GIS, a model called ‘NPS pollutant load in runoff’ has been generated for estimating the pollutant loads of chosen constituents in surface runoff water (Fig.). The Avenue script written in this program/model associates or links EMC value of various pollutant constituents to the land use types. The input data for this model are: a) a grid of land uses in the urban area, b) a grid of average annual runoff volume in the basin and c) associated EMC values of chosen / selected pollutant constituents (nitrate, chloride, benzene, toluene, ethyl benzene, xylene and total suspended solids).

This program creates an EMC grid (for the above selected pollutants) and multiplies it by a grid of average annual rainfall runoff in the basin. The result will be the annual loading of the NPS constituent to each grid cell in the urban area, i.e.:

\[
\text{Load} = \text{Flow} \times \text{Concentration}, \text{ or } \\
L (\text{Mass/Time}) = Q (\text{Volume/Time}) \times C (\text{Mass/Volume}) \quad (\text{Equation 10})
\]

The output from this model consists of predicted annual pollutant loading (e.g. mg/m²/year) for each land use type, which affects the urban watershed.

C.4.2 Estimation of Initial Nonpoint Source Pollutant Fluxes in Recharge

The typical pollutant concentration associated with various land use categories in the study area, i.e. EMC value, can be used for estimating the initial NPS pollutant fluxes in the infiltrated water also. Therefore, using this relationship, on multiplying the volume of infiltrated water with the EMC value would give an estimate of the initial pollutant fluxes in the recharge.
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Initial pollutant fluxes in infiltrated/recharge water = EMC_{runoff} \times \text{Infiltration volume} \quad (\text{Equation 11})

The infiltrated water is subject to evaporative loss across the soil-air interface and hence the concentration of the recharge water will be higher than that of the infiltrated water. The potential recharge value, calculated in the runoff-recharge model, is the quantity of infiltrated water left after the adjustment for loss through evaporation. The calculated pollutant flux is the starting NPS pollutant fluxes, which enter the top part of the unsaturated zone, and is subject to the various physicochemical processes while it pass through the entire thickness of the unsaturated zone. The starting or initial NPS pollutant concentration in the recharge water can be estimated as:

Initial Recharge concentration = Initial recharge pollutant flux / Recharge volume \quad (\text{Equation 12})

Using the above relationships, (equations 10, 11 and 12) another sub model called “initial NPS recharge pollutant flux model” has been programmed in the ArcView GIS to estimate the initial pollutant flux and initial pollutant concentration in the recharge water (Figure 5).

Figure 5. Interface of ArcView GIS based Non-point Source BTEX Pollution Model. This model requires the following input data:
- Typical pollutant concentration value for each land use (EMC)
- Direct Runoff Volume grid map
- Cumulative infiltration volume grid map
- Potential recharge volume grid map

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The output from this sub-model is a series of maps showing the spatial distribution of initial pollutant fluxes and initial concentration in the recharge for the chosen parameters (nitrate, chloride and BTEX compounds). The concentrations of NPS pollutants and pollutants loads of the chosen constituents in the runoff water are also estimated while running this model.

### C.4.3 Calculation of Final Non-point Source Pollutant Fluxes in Recharge

Pollutants in the infiltrating recharge water may be subjected to various subsurface natural attenuation processes (such as sorption / retardation, in case of any volatile contaminant then vaporisation / volatilisation, and biodegradation, other transformation through microbiological activities etc) as it migrate though the vadose zone, whereas few pollutants like chloride in recharge remain as ‘non reactive’. Among the six pollutants chosen for GIS modelling under the present research, chloride is the only non-reactive solute; hence, the final concentration of chloride in the recharge can be modelled as equal to the initial concentration. The BTEX chemicals in the recharge are subject to sorption/desorption, biodegradation and volatilisation processes, whereas the nitrates are subject to other biological activities such as denitrification, uptake by the plants, nitrification and nitrogen fixation etc. Prediction of nitrate fate and leaching in the unsaturated zone is difficult due to the complex nature of the processes dominating the fate of nitrogen in soils. It was for this reason that it was decided to consider nitrate as a ‘non-reactive’ solute during its vadose zone migration. Thus, the initial concentration predicted in the previous model is treated as the concentration entering to the water table.

### C.4.4 Fate of Nitrate in the Infiltrating Recharge

Nitrate in the infiltration that passes through the top soils (having plant root systems) and underlying vadose zone are subject to various processes which include mainly mineralisation, plant uptake, nitrate leaching, immobilisation, and denitrification. There is some gain of nitrate in the soil system through nitrification (conversion of NH₄ to NO₃) and mineralisation (decomposition and transformation of organic N to inorganic N. The primary processes of subsurface nitrate removal are generally considered denitrification, vegetative uptake or dilution. However, in many studies the exact mechanism of nitrate removal and gain of nitrate and the role that subsurface hydrology plays in nitrate attenuation and nitrate formation have not been well established.

While a large amount of research has focused on nitrogen cycling in the root zone of plants (0-2 m depth), little is known about the fate of nitrogen between the root zone and the groundwater table. Although many sophisticated numerical models (eg. OPUS, EPIC etc) have been developed for simulating nitrogen behaviour in the soil-plant system, applying their concepts and developing a GIS based version in practice is difficult, mainly due to the limited specific-site information and GIS coding difficulties. Because the nitrogen system is poorly understood in the urban environment, it was decided simply model NO₃ inputs and to ignore subsequent transformation. This means that the calculated NO₃ flux should be viewed as the maximum N flux derived from NO₃ at ground surface: N flux because the NO₃ will not necessarily still be in the form of NO₃ at any given depth, and maximum because source NO₃ may be retained as NH₄ (or even, possibly, organic N) in the soil and vadose zones. Thus the initial concentration predicted in the previous model is treated as the concentration reaching the water table.

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C.4.5 Modelling Steps / Procedure for Estimation of BTEX Pollution in Recharge

In this model, the BTEX recharge pollutant fluxes are estimated through four stages viz.:

1. Estimation of volumetric water content in the unsaturated zone (using Clapp and Hornberger method)
2. Calculation of soil-water partitioning coefficients, \( K_d \)
3. Calculation of unsaturated zone retardation factor for BTEX compounds
4. Calculation of final BTEX concentration and BTEX pollutant mass flux entering to the water table.

C.4.5.1 Estimation of Volumetric Water Content in the Vadose Zone

The volumetric water content in the vadose zone can be calculated using the Clapp and Hornberger method (Clapp and Hornberger, 1978). Using Clapp and Hornberger method the volumetric water content of soil, \( \theta_w \) is given by

\[
\theta_w = \theta_s \left[ \frac{V_d}{k_s} \right]^{1/2b+3}
\]

where \( \theta_s \) is the saturated water content of the soil (total porosity)

\( V_d \) is the recharge rate

\( k_s \) is the saturated hydraulic conductivity at the saturated water content \( \theta_s \)

\( b \) is the Clapp and Hornberger constant for the soil.

Clapp and Hornberger constant \( (b) \):

This is the constant in the Clapp and Hornberger equation relating the relative saturation of the soil to the relative conductivity of the soil (Clapp and Hornberger, 1978).

\[
\frac{\theta}{\theta_s} = k^{2b+3}
\]

where \( k \) is the hydraulic conductivity of the soil at a volumetric water content \( \theta \) and \( k_s \) is the saturated hydraulic conductivity at the saturated water content \( \theta_s \) (total porosity in fraction). \( b \) is Clapp and Hornberger constant. If \( b \) is not known it can be estimated using the values presented by Clapp and Hornberger for different soil textures (Clapp and Hornberger, 1978).

The required input data for this sub-model are soil texture type, porosity, saturated hydraulic conductivity and Clapp and Hornberger constant values for the respective soil texture present. The various sub programs/codes provided in this model (Figure 5) help in assigning input data and later calculate the volumetric water content.

C.4.5.2 Calculation of Soil-water partitioning coefficient, \( K_d \)

Soil-water partitioning coefficient, \( K_d \), (also called soil adsorption coefficient or soil water sorption coefficient, \( K_s \)), expresses the tendency of a chemical compound to be adsorbed onto soil or rock or sediment particles. Over time, the dissolved contaminant will migrate from the free water and sorb onto soil or rock particles by a process called sorption. Sorption is an important process affecting the transport of contaminants in the subsurface. Due to sorption the leading edge of dissolved chemical plume moves slower than the infiltrating recharge water flow and or the groundwater flow, thus the

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contaminant migration rate or velocity is retarded, which is called retardation. The magnitude of the sorption for most soil water system is dependant on the hydrophobicity (measured by aqueous solubility) of the chemical and organic carbon content of the soil (Gustafson et al., 1997).

At equilibrium, a non-polar organic compound is seen to distribute itself between aqueous concentration $C_a$ and sorbed concentration $C_s$, as a function of their ratio: $K_d = C_s / C_a$, with $K_d$ the soil sorption or partition coefficient. The slope of the line plotting the liquid phase (aqueous) concentration versus the solid phase (sorbed) concentration for various combinations of liquid phase and solid mass concentrations is $K_d$. The value of $K_d$ for a given organic compound is not constant, however, but researchers (Karcikoff et al., 1979) have found that for non-ionic hydrophobic chemicals the primary soil property controlling sorption is the fraction of organic carbon content of the soil. It has been that the value of $K_d$ tends to increase linearly for soils with increasing organic carbon and clay contents. The slope of the relationship between $K_d$ and % organic carbon is the amount of sorption on a unit carbon content basis called $K_{oc}$ (Hassett and Banwart 1989), in which $K_{oc} = K_d / f_{oc}$ where $f_{oc}$ is the fraction of organic content in the soil.

Therefore, the soil-water partitioning coefficient, $K_d$ can be estimated using the following relation/equation

$$K_d = K_{oc} f_{oc}$$

where

- $K_d$ = soil-water partitioning coefficient
- $K_{oc}$ = organic carbon partitioning coefficient
- $f_{oc}$ = fraction of organic carbon within the soil matrix (the fraction of organic carbon $f_{oc}$ must be measured for each drift deposit/soil type of the study area)

### C.4.5.3 Transport of Aqueous Phase Contaminants in Vadose Zone

This sub-model simulates the downward movement of pollutant with soil solution and estimates the concentration of pollutants and pollutant flux from the vadose zone.

**Velocity of aqueous phase contaminant migration**

In a steady state single-phase flow system the average linear ground water velocity, $v_i$, is given as

$$v_i = \frac{q_i}{n}$$

where $q$ is flux of ground water in steady state single phase flow; $n$ is porosity of the medium.

In the unsaturated zone, the term $q$ can be replaced by the recharge flux and porosity can be replaced by the volumetric water content, $\theta$. Therefore, the velocity of the pollutant in the vadose zone is given by

$$V_p = V_a \div R_i$$

where $V_a$ = Aqueous or pore water velocity;

$R_i$ = Retardation factor for the pollutant in the vadose zone.

$$V_a = \frac{\text{Recharge rate, } V_\theta}{\text{Volumetric water content of the soil, } \theta}$$
Recharge rate $V_d$ is the average downward flux density of water through the vadose zone (expressed as m / day).

Or in other form the velocity of the pollutant in the vadose zone can be calculated as

$$v_p = \frac{q}{\theta R_f} = \frac{V_d}{\theta R_f}$$

**Retardation factor**

The retardation factor, $R_i$, for the vadose zone is calculated using the following equation:

$$R_i = 1 + \left( \rho K_d + (\theta_s - \theta) K_i \right) / \theta$$

where $\rho$ is the bulk density of the soil,

- $\theta$ is the water content of the soil on a volume basis (volumetric water content of the soil)
- $\theta_s$ is the saturated water content of the soil on a volume basis
- $K_d$ is the partition coefficient for pollutant in the soil
- $K_i$ is the dimensionless value of Henry's law constant, $(c_a / c_w)$.

The partition coefficient $K_d$ can be calculated as $K_{oc}$ where $K_{oc}$ is the organic carbon partition coefficient and foc is the fractional organic carbon content of the soil. On comparing the above equation to the normal retardation equation for a saturated situation, $R_i = 1 + (\rho K_d / \theta)$ which does not contain any air phase, the difference noticed in it is the incorporation of Henry's law. The incorporation of Henry's law in the above equation helps to account for the loss of dissolved volatile organic compounds like BTEX through volatilisation. However, it has been noted that if volumetric water content is too low the use of the above equation might lead to the prediction of higher retardation factors.

**Time taken to reach water table**

The leading edge of the contaminated pulse will reach the water table at a time $T$ that can be calculated as:

$$T = \frac{z \theta R_i}{q}$$

where $z = \text{depth of the vadose zone}$; $q = \text{net recharge rate}$

**C.4.5.4 Estimation of Pollutant Concentration**

In subsurface environments, biodegradation is known to reduce the level of contamination. Biodegradation is a natural process by which naturally occurring microorganisms such as bacteria, fungi breakdown soil and groundwater contaminants into less toxic substances. It can take place in the presence of oxygen (aerobic condition) or without oxygen (aerobic condition). In an aerobic environment, biodegradation of fuels generally follows a first order relationship. In addition, Salanitro (1993) has pointed out that biodegradation in laboratory microcosm experiments follows the first order reaction rule for aromatic hydrocarbons, including BTEX. Therefore, concentration of BTEX pollutants exiting the vadose zone can be estimated using the following first order decay model equation

$$C_2 = C_1 \exp (-Tk)$$
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where

\[ C_i = \text{initial concentration of chemical applied at the ground surface or in the soil through leaks} \]
\[ C_2 = \text{concentration of chemical exiting the vadose zone} \]
\[ T = \text{percolation time or travel time through the vadose zone} \]
\[ k = \text{first order degradation rate coefficient for the chemical} \]

\[ T_{1/2} = \frac{\ln(2)}{\lambda} \approx \frac{0.693}{\lambda} \]

where \( T_{1/2} \) is the half life period; \( \gamma \) = first order degradation rate coefficient for the chemical. Half-life period is the time required for getting one half of the original contaminant degraded. Substituting the travel time and degradation rate in the above equation, it becomes

\[ C_i = C_i \exp \left( \frac{-0.693 \cdot T_{1/2} \cdot Z_0}{qT_{1/2}} \right) \]

This is the final equation programmed within the GIS to calculate final concentration reaching to the water table.

\subsection*{C.4.5.5 Pollutant Flux to the Water Table}

The one dimensional pollutant mass flux due to advection is equal to the quantity of the water flowing / infiltrating (net recharge rate) times the concentration of dissolved solids or solutes. Therefore, the BTEX pollutant loading / flux to the water table from the vadose zone is given as

\[ \text{Pollutant Flux} = \text{Net Recharge Rate} \times \text{Concentration of Pollutant reaching water table} \]

or Flux = \( qC_2 \)

i.e., \( \text{Recharge Pollutant Flux} = qC_i \exp \left( \frac{-0.693 \cdot T_{1/2} \cdot Z_0}{qT_{1/2}} \right) \)

\subsection*{Petrol Station BTEX Pollution Model}

Small-scale petrol spill/leakage incidences happening either from underground storage tanks leaks or from surface spills while transferring / storing gasoline are common in fuelling stations. Such smaller spill/leakage incidences result in the formation of residual nonaqueous phase liquid releases in the unsaturated zone that serves as source of aqueous contaminants to urban groundwater. Infiltration passing through such regions dissolves the soluble contaminants forming leachates of contaminants comprising mainly benzene, toluene, ethyl benzene and xylene (BTEX), which further migrate through the vadose zone and finally reach the water table as BTEX recharge pollutant fluxes resulting in groundwater pollution. The fate and transport of the BTEX in the unsaturated zone is a complex physical process controlled by their chemical properties and the properties of the transport media.

The 'petrol station BTEX pollution' model is meant to assess the reactive dissolved phase migration from ground level to the water table for cases where spills of (multi-compound) non-aqueous phase liquids (NAPLs) have occurred. It uses inputs from an urban recharge GIS model and estimates the BTEX recharge pollutant fluxes entering to the water table, which results from small scale spill incidents in urban areas.
model calculation is undertaken in seven steps, viz. 1) input of data; 2) estimation of volumetric water content in the unsaturated zone; 3) calculation of soil/water, air/water, and NAPL/water partitioning coefficients; 4) multiphase partitioning of spilled NAPLs into the four phases; 5) calculation of initial leachate concentrations for each NAPL component; 6) calculation of retardation factors; 7) calculation of final concentrations and fluxes reaching the water table taking into account degradation. The main input data required are: locations, volumes, and areas of spills; soil texture, hydraulic properties, and organic carbon content; recharge rates; and water table depths. The results from the model can also help in identifying those petrol station posing maximum threats to groundwater contamination from BTEX chemicals. The model has been trialled on the Birmingham (UK) aquifer for the case of petrol stations (Thomas and Tellam, 2005). In this aquifer, the risks to groundwater appear to be minimal for small-scale spills, except where the unsaturated zone thickness is limited close to the main river. The program could be used for vulnerability mapping, determining threats from existing NAPL storage sites, and in planning locations for future sites.

**Sewer Pollution Assessment Model**

The GIS based sewer leakage and groundwater pollution assessment model estimates the amount of indirect recharge through sewer leaks and pollutant fluxes from sewage networks to an unconfined urban aquifer. This model, which runs in steady state mode, estimates pollutant fluxes to water table based on one-day sewer leakage estimates from urban sewers (currently foul, combined and industrial sewers) classified based on grading of their leaking condition. It is implemented in ArcView GIS platform using Avenue programming language scripts, and estimates sewer leakage and pollutant loads of the chosen pollutants (currently toluene) through simulation of mainly five steps of calculations (viz. infiltration and exfiltration condition of sewers, sewer flow rate estimation, indirect recharge through leakage, initial pollutant fluxes calculation, and pollutant transport through vadose zone). The out put from the model are leaking flow rate distribution, distribution of initial pollutant loads and fluxes, travel time of each chosen pollutant through vadose zone and the final amounts of pollutants entering the water table. This model can be used to identify the portions of unconfined urban aquifers, which are more likely to the risk of contamination.

**Other Applications or Uses of UGI f Model**

The developed models can be used for predicting the net change in recharge and chemical flux distribution due to change in rainfall amount (e.g. climate change), installation of “permeable pavements”, and change in land use. Urban cover permeability alteration is achieved by either increasing the amount of paved areas (concrete or asphalt cover) or by replacing the paved areas by materials, which allow more infiltration of rainwater, for example bricks. Increasing the permeability of the paved areas is useful in helping to reduce peak flow rates in the drainage systems. Land use change in urban areas occurs for example due to increasing demand for housing facilities. Examples of land use changes are conversion of agriculture areas to residential area or erection of more residential buildings in open areas or low-density residential areas.

The developed model can be extended to model the recharge and pollutant flux distribution back in time (e.g. 1850-1990) in an urban area provided all the input data are available for the period chosen for modelling. Historical land use and hence the typical concentration associated with each land use would be quite different. However
the geology, and hence the HSG would be the same. The rainfall data for a period back in time might be available; however, MORECS data may well not be available. For extending the method back to a time for which the MORECS data are unavailable, a new model of evapotranspiration and soil moisture would have to be developed. The extended model could then be tested by using its output in a regional groundwater flow model.

The recharge and pollutant flux distribution obtained from the model could also simply input to a three-dimensional groundwater flow model to investigate migration of pollutants within the groundwater system. The new model could be modified to include explicitly leaching from contaminated land sites, and to include phase partitioning now only included in the point source (PS) model. The point source model (petrol station pollution) can be used to model the pollutant flux distributions associated with an accidental spill of NAPL (or dissolved phases) from a storage tank in the industrial area. Even though the storage tank occupies perhaps only a little area (e.g. 1% of total industrial land use), the effect of many such spill incidences from different locations in the industrial area can cause groundwater contamination of a wider region underneath the industrial areas. The model could indicate which region would have the greatest effect (in terms of concentration and flux rates of BTEX compounds).

Groundwater pollution aspects related to contaminated lands were not covered in the study conducted at Birmingham. However, the PS model and the NPS model could be used to simulate the pollutant loading from the contaminated land sites. In principle, the model could be run in risk assessment mode, i.e. be repeatedly run with input parameters sampled from user-defined probability density functions, in order to obtain an idea of the likely recharge/flux probability distributions. However, this would require very large computer resources, and some means of avoiding ArcView’s in built limitations. One possible method of getting around these problems might be based on simplifying the recharge calculations using the correlations of recharge with rainfall. This would involve losing spatial data.

Many vulnerability schemes use geology as their basis (e.g. that of the Environment Agency in UK). Others are more sophisticated (e.g. DRASTIC). The GIS model developed here could potentially be used in vulnerability assessment, with the advantage of being a fairly detailed recharge model, including reaction terms related to geology, and in being able to undertake spatial assessment as well as site specific assessment. A model run with a uniform EMC across the area would quickly indicate the vulnerable zones in the aquifer, and hence be of some use for regional planning. Work would be necessary to compare the output of the model and more standard approaches.

Although the model is primarily developed for the Birmingham unconfined aquifer, it can also be extended to other similar UK cities (Nottingham, Manchester and Liverpool) provided all the basic inputs (mainly land use, hydrologic soil group, surface elevation, geology, meteorological data, depth to water table and hydraulic properties of the geological units) are available. In particular, Manchester and Liverpool have many similarities to Birmingham in their basic geology and land use description. If drift deposits are taken out then the Birmingham is similar to Nottingham. In some respects Birmingham is similar to all other cities in UK, and therefore the developed method could be tried/used there also provided the basic inputs of land use, soils,
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meteorological data, EMC values, surface elevation, geology and its hydraulic properties and water table depths are available.

C.5 Major Limitations of the UGlf Model

Although the UGlf model is not very sophisticated one, it does take into account the principal processes involved, and, as it is incorporated in a GIS, it allows the complexities of spatial heterogeneity to be investigated. A major limitation is in the way that time is dealt with. It is assumed that land use and land use-related properties do not vary within the ‘time-slice’ or period being considered by the model. Daily recharge estimations are undertaken, and summed over the user-specified period. Within this period, steady-state conditions are assumed for the movement of water and solutes through the unsaturated zone. Thus individual recharge pulses are not tracked: residence time in the unsaturated zone is calculated on the basis of the averaged recharge rate, but it is only used, with a delay arising from any sorption, to estimate degradation/decay of the pollutant concentration. Without incurring very considerable computer run times, it would be difficult to track individual recharge pulses simultaneously.

In Nonpoint source pollution assessment, all units of the same land use type are assumed to have the same Event Mean Concentrations (EMC) value regardless of their spatial location within the city. However, in reality the concentration of pollutants in recharge water will vary depending on the soil type/vadose zone chemistry. From the made ground/drift/solid geology formations, release of the pollution to the recharge is possible due to past human activities e.g. accumulation of waste material. Sometimes the background concentration in the subsurface may add to the recharge. At present, this variability is not accounted in the model due to lack of information on the background chemistry and heterogeneity of the urban hydrogeologic system. Another weakness is the lack of feedback between water table elevation and recharge rate: this could conceivably be added by iteration with a regional groundwater flow model, though again there are computer run time implications.

C.6 Prediction Accuracy

The different sub models in UGlf make use of many input parameters (both spatial and non spatial data) and the accuracy of their predictions is dependent on the assumptions made in each sub models and the accuracy of the input data used.

C.6.1 Uncertainty Aspects

Environmental models are simplified representations of systems in reality, and uncertainty is always associated with their representations. In many cases the systems, especially urban groundwater systems, are heterogeneous, where a wide range of parameters with a wide range of possible values for them control the complex behaviour of the system. In the case of recharge and solute transport simulations of urban environments, the hydraulic and transport parameters are never known in sufficient detail. If the input parameters used in the present GIS based urban pollutant flux models are based on literature based typical inputs the predictive runs and the results obtained from them are subject to much uncertainty in relation to the complex heterogeneity of the urban system being modelled. There may be additional
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uncertainty relating to whether the conceptual model with simplified analytical equations is fully applicable to the field situation in an urban area.

C.7 Model Running Issues

The simulation of the fate and transport of BTEX compounds in the NPS pollution model involves combining of eight grids into a single grid. The input grids needed are the following: direct recharge grid, land use grid, geology grid, vadose zone depth grid, and initial concentrations of benzene, toluene, ethyl benzene and xylene. After combing these grids, the value fields of respective grids are queried and text values of land use types, geology names and lithology are assigned to the combined grid. This grid combining and assigning of various text attributes takes long time (nearly two hours on Pentium III 450 MHz speed and 128 MB RAM PC). Later, other hydraulic properties (bulk density, soil texture, porosity, and hydraulic conductivity), organic carbon content and Clayp and Hornberger constants have to be assigned to each geological unit. The combined grid of a simulation for average daily recharge during six months may have enormous records and will occupy large file size (e.g. 23MB size) before assigning the geologic attributes. A whole model run takes 6-7 hours in such situations.

C.8 Conclusions

Estimation of recharge pollutant fluxes reaching an urban unconfined aquifer system is challenging because of the complexity of urban hydrogeological systems and is a complex spatial environmental problem. GIS is an appropriate tool for such environmental analysis. The urban groundwater recharge fluxes being spatially variable the best way to model them is through an integrated modelling approach involving use and analysis of various thematic data (aerial photographs, satellite imagery and various vector and raster maps) and other attribute information within a Geographic Information System. An ArcView GIS based methodology developed herein can provide reasonable estimates of groundwater recharge and pollutant flux rates in urban environments. This study could develop a GIS based urban groundwater recharge and pollution source distribution model and a GIS based vadose zone reaction model for selected pollutants viz. nitrate, chloride and BTEX compounds.

The model described herein has attempted to address the process of groundwater recharge and pollutant transport through the vadose zone region of urban unconfined aquifers. It essentially combines the normally separated disciplines of contaminant hydrogeology and water resources hydrogeology, examining regional scale contaminant issues in urban areas. The model presented is an abstraction of reality, so errors will always be present. However, the model provides a way to better understand a problem and to test alternatives. Using the above methodology, clearly, it was not possible to develop a definitive representation of urban recharge processes, but a framework was set up, which will allow investigation of the main issues. With this initial framework, it will be possible and appropriate for progressive upgrading in future studies. This model can provide information on the quantity and quality of recharge water, potentially of UK to help the various administrative bodies in formulating decisions on the use of urban groundwater and its management for future use. It can also be used to identify the risks of groundwater use on a sustainable basis on a regional scale.
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C.9 References


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