Subsurface transport from contaminated sites
Part 2 - groundwater

D.S. Sheppard Geochemical Solutions, P O Box 33 224, Petone

L.J. Brown 61 Penrose St, Lower Hutt

W.J. McCabe and M. Burgess c/- Institute of Geological and Nuclear Sciences Ltd,
P O Box 31 312, Lower Hutt

Abstract:

Contaminants on or in soils may move into the wider environment through a number of routes. The soils themselves retain contaminants to a greater or lesser extent, and the movement of contaminants from soil occurs through evaporation and dust generation, intake into plants through their roots and flushing by or dissolution into water seeping through the soil. Water transport of contaminants will usually result in contamination of surface water bodies, through surface water drainage and by way of groundwater aquifers. The relevance of specific hydrogeological factors to groundwater contamination and the subsequent transport of contaminants to distant ecosystems are assessed in this study. The factors specified in this extension are the nature of the aquifer material, the type of aquifer (confined and unconfined) and the physical nature of the soil and vadose zone connections between the surface and subsurface waters (Part 1).

Table of Contents

1. Introduction

A very common route for the transport of contaminants from on or in soils to the wider environment and distant ecosystems is through the seepage of water through the soils and sub-soils in the unsaturated (vadose) zone into groundwater aquifers. The effects of the subsoil on the rate and distance of movement of contaminants is largely unknown for specific New Zealand soils in particular and for most subsoil types in general. It was beyond the scope of our study to carry out comprehensive and definitive measurements of the effects of all, or even a selection of, significant variables affecting the properties of the subsoils and their adsorptive capacity. Instead, we have ascertained the relative retention properties under typical conditions for important types of New Zealand subsoils (Part 1).

The effectiveness of the subsoil in inhibiting contaminant migration determines to a large degree the rate and degree of contamination of the underlying groundwater. Groundwater is a principle vector for the transfer of contaminants into the wider environment and so the contaminants can affect the ecosystems in the media through which the water and groundwater move, the systems into which they discharge (springs, lakes, rivers, harbours, estuaries, wetlands, coastal zone) and the users of the groundwater, for drinking, in process industries or agricultural irrigation (see Fetter, 1993). The risks posed by contaminated sites must take these into account and this involves good understanding of the groundwater systems themselves, the transport processes through the unsaturated zone and in the saturated zone, the chemical and biological processes acting on the contaminants which will affect the transport, and the exposures of the at risk receivers.

The second part of the study examined the hydrogeological factors, built on the New Zealand experience with the DRASTIC groundwater vulnerability assessment model (Aller *et al.*, 1987) and modified and extended the factors, weightings and methods of combinations of the factors to suit the problem being addressed. While DRASTIC provided a starting point, the end product cannot be compared to the DRASTIC system.

2. Groundwater Contamination Vulnerability

2.1 Introduction to groundwater vulnerability studies and concepts.

Margat (1968) first introduced the concept of vulnerability of groundwater to contamination. Since then various techniques have been developed to assess relative vulnerability. The scale of the investigation and whether the concern is for specific or intrinsic (natural) vulnerability often determine technique selection. "Specific vulnerability" considers the potential impacts of specific land uses and explicit contaminations such as pesticides, which may prove detrimental to present or future uses of the groundwater resource. "Natural vulnerability" is a function of hydrogeological factors – the characteristics of an aquifer and the overlying soil and geological materials (Bates & Otto, 1998). In the assessment of environmental tolerances both "specific vulnerability" and "natural vulnerability" need to be considered. The environment is affected by land-use and misuse and any resulting contamination of groundwater, and by the dewatering of the aquifer and resultant deterioration of groundwater quality due to inflow of inferior quality groundwater.

DRASTIC (Aller *et al.* 1987) is a mnemonic for a technique for assessing groundwater vulnerability to contamination. It is based on a simple additive overlay approach and considers seven hydrogeological factors that are considered to be readily available, able to be mapped and critical for assessing groundwater contamination potential in terms of an index. These are:

D	Depth to water table
R	Recharge
A	Aquifer media
S	Soil media
T	Topography
I	Impact of the vadose zone
C	hydraulic Conductivity

The seven factors are incorporated into a relative ranking scheme that uses a combination of ratings and weights to produce a numerical value, called the DRASTIC index. Each of the factors is ranked between 1 and 10 and the rank is multiplied by an assigned weighting which ranges between 1 and 5. The weighted ranks are summed to give a score for the particular hydrogeological unit with the higher scores indicating greater vulnerability to contamination.

Agricultural DRASTIC weights have been developed to rank groundwater vulnerability to contamination from pesticides or other specific contamination sources. These place more weight on the soil media and topography and reduced weight on the impact of the vadose zone and hydraulic conductivity.

New Zealand aquifer systems occupy a wide variety of geologic environments within a latitudinally elongated but small land area which results in highly diversified and compact hydrogeological environments. Groundwater investigative techniques designed in other countries usually require modification to be generally applicable in New Zealand and this include vulnerability assessment systems.

In New Zealand the DRASTIC index method has been used for assessing contamination vulnerability of groundwater resources on an aquifer scale (Brown, 1998a – Heretaunga Plains aquifer system), on a regional scale (Brown *et al.*, 1994 – Wellington), and on a national scale (Close, 1993). The index methodology is simple, pertinent, and can be extended to meet the diverse format, quality and type of data inputting to the hydrogeologic settings.

However, several studies stated reservations with regard to the application of DRASTIC and the results. For example:

- Close (1993) noted that *"for New Zealand it appears that the ranking of the top three regions is relatively insensitive to the weightings used."*
- Brown (1998a) included a disclaimer designed by Zaporozec (1987) – *"this vulnerability map is designed for general and planning use only. It shows the sensitivity of groundwater to contamination in a generalised way; local details have been generalised to fit the map scale. The map does not show areas that have been or will be contaminated, or areas that cannot be contaminated, and the map cannot be used for any site-specific purposes. Detailed studies of individual areas may be necessary when specific information is needed. Characteristics of individual contaminants or the likelihood of contaminant release have not been taken into account when constructing the map.*
- Webb and Lilburne (1999) expressed *"doubts about the application of DRASTIC ratings to New Zealand aquifers"*, and that the application of the LEACHM leaching index model they used for their study *"be tested against field data and modified"* before it is applied to floodplain soil units.
- Brown (1998b) in reviewing the application of DRASTIC to New Zealand floodplain aquifer systems concluded, *"at present there is no clear answer as to whether vulnerability assessment and mapping (DRASTIC in particular) can represent the highly diversified hydrogeology of New Zealand alluvial plain aquifer systems."*

In this study the application of DRASTIC is reviewed with regard to New Zealand hydrogeologic settings and New Zealand soils and adaptations made to DRASTIC methodology to achieve more meaningful application to New Zealand aquifer and soil systems. The soil media factor is considered and modified by incorporating the results of laboratory experiments as detailed in the following chapter 3 case studies of actual contaminated sites.

The eventual aim of the project was to contribute to effects on ecosystem tolerances at the surface water – groundwater interface. These interfaces occur where surface water (river, lake or rain) infiltrates to groundwater, and where groundwater contributes to surface water bodies. A final vulnerability assessment model combining all factors discussed and assessed in this report is available on this website as an Excel-based spreadsheet which takes data supplied by an investigator concerning a site and calculates the risk of contamination of the underlying groundwater and downstream surface-water ecosystems.

2.2 New Zealand Aquifer Systems

The hydrogeological properties of groundwater aquifers can be broadly grouped according to the type and age of the sediments and rocks in which they occur (Brown and Gregg, 1994). These are volcanic, sedimentary and metamorphic rocks, and floodplain and river valley gravel, sand and silt deposits.

The uplifting axial mountain ranges in New Zealand are predominantly composed of greywacke (indurated sandstone). The erosion of the mountains, and transport and deposition of the eroded debris by rivers, has been enhanced by the alternating cold and temperate climate cycles of the Quaternary (last 2 million years). Deposition and reworking of durable gravel clasts has built floodplain surfaces with extensive fans at the mountain margin, braided river channels crossing the plains, and deltas, lagoons and estuaries at the coast.

Where deposition is infilling subsiding coastal basins or fault angle depressions, layered sequences of fluvial gravels forming aquifer systems are interbedded with marine silts and sands forming aquicludes and aquitards. Marine deposition occurred when the climate was warm and sea level similar to the present. Aquifer systems of this type underlie the northern Canterbury Plains, Manawatu Plains, Heretaunga Plains, Hutt Valley, and Wairau Plain.

Floodplains in tectonically stable or uplifting areas are usually underlain by gravel with variable sand, silt and clay matrix. Water table and confined aquifers occur within the fluvial gravel deposits of inland basins and floodplains, beneath river valleys and within coastal dune, beach and estuary-lagoon deposits of gravel, sand and silt. These aquifers occur throughout New Zealand and include the Ruataniwha Plains, Waimea Plains, Kaikoura Plain, Southland Plain and the Clutha River valley.

Extensive areas of volcanic rock formed from lava flow deposition, with cooling fractures, lava tubes and porous rubbly interflow layers, contain groundwater and form another important aquifer group. Relatively young volcanic rocks (one million years old) in central North Island, Bay of Plenty, northern Northland and Auckland City contain groundwater in near-surface water table aquifers and deeper confined aquifers.

Older volcanic rocks such as andesite at Coromandel-Thames and Dunedin, and basalt at Timaru, North Otago and Banks Peninsula also contain aquifers.

Older (early Pleistocene to Cretaceous) sedimentary rocks are less likely to contain high yielding groundwater aquifers than the fluvial gravel and volcanic rock aquifers. Shale, mudstone and siltstone are usually of low permeability unless the rocks are extensively fractured or jointed. Sandstone may be more permeable because of the spaces between sand grains. Conglomerate permeability depends on the sorting of the sediment during deposition and the degree of cementation that has occurred since deposition.

Limestone is most likely to contain high yielding aquifers with the accessible groundwater present in solution channels (including caves), fracture zones and joints. The permeability of coquina limestone and cemented shellbeds depends on the degree of cementation.

Metamorphic rocks (including greywacke and schist) form the mountain ranges and the "basement" of most of New Zealand. Greywacke with zones of fissures or cracks can transmit and yield groundwater to wells. In Wellington City wells produce water from greywacke (Henderson 1937) and landfills sited in former quarries or in valleys in greywacke are problematic in that leachate can leak from the site into the greywacke rock and appear hundreds of meters away (Sheppard *et al*., 1999).

Metamorphic rocks with pronounced schistosity and associated aligned foliation and fractures have a high permeability that can extend through extensive thicknesses of these schist rocks. Schist rocks in northern Marlborough, North and Central Otago provide reliable water supplies.

Other metamorphic rocks such as granite and gneiss tend to be relatively impermeable. It is difficult to predict the potential for location of significant water supplies within metamorphic rocks where fracture zones or fissures are present and, because the connection to the recharge source is often convoluted, the sustainability of supply may be uncertain.

2.3 New Zealand Applications of DRASTIC

Several applications of DRASTIC groundwater contamination vulnerability assessments to New Zealand groundwater systems have already been mentioned (Close, 1991, 1993; Brown *et al.,* 1994; Hadfield & O'Regan, 1994; Brown, 1998a). Close (1993) compared 17 "regions" throughout New Zealand where significant amounts of pesticides were used to maintain horticultural production. He applied the agricultural ranking index which places more weight on topography and soil media than the standard DRASTIC, to assess groundwater contamination vulnerability with regard to pesticide usage, mobility, and persistence. Six of the higher-ranked regions were selected for groundwater sampling for pesticide analyses and more detailed study. In terms of the definitions of Bates & Otto (1998) this is a "specific vulnerability" application of DRASTIC.

Brown *et al.* (1994) applied the DRASTIC index method to compile a 1:500 000 scale groundwater contamination vulnerability map of nine hydrogeologic settings for the Wellington Region. The study was designed to provide information for planning purposes to formulate policies on the land-use activities with potential to discharge contaminants on land.

Hadfield & O’Regan (1994) studied the potential for groundwater pollution from the land treatment of dairy industry waste on the Hauraki Plains. GIS databases were utilised to incorporate and compare data relevant to DRASTIC, to identify the areas most vulnerable to surface contamination and where more rigorous investigations could be focused. The DRASTIC evaluation indicated distribution of contamination potential consistent with a previous study of the distribution of nitrate in groundwater in the Hauraki Plains.

Brown (1998a) applied the DRASTIC method to a comprehensive groundwater database for the aquifer system underlying the Heretaunga Plains, Hawkes Bay. Several adjustments and modifications were made to DRASTIC to utilise the most appropriate data for the assessment. In addition to the DRASTIC index factors, data sources contributing to this assessment were direction of groundwater flow (piezometric contours), source of groundwater recharge (oxygen 18 isotope analyses), rate of groundwater flow (tritium isotope analyses), transmissivity and thickness of overlying strata. There were also groundwater chemical data which indicated where groundwater might already be contaminated with nitrates and where mixing was occurring with groundwater of higher mineral content from sediment or rock aquifer types other than the greywacke gravel of the Heretaunga Plains aquifers. The product of this study was a GIS-based map for the Hawkes Bay Regional Council showing the aquifer contamination vulnerability of the Heretaunga Plains based on specifically modified DRASTIC factors for confined aquifers.

The Hawkes Bay Regional Council have used the map to:

- review the Heretaunga Plains and adjacent areas groundwater quality monitoring network to make sure that there were monitoring wells sited in areas with a high vulnerability to contamination (prioritisation of monitoring);
- eliminate duplication of monitoring wells especially where there was low vulnerability potential;
- identify areas where either additional or less stringent protection measures might be required;
- manage/design land-use practices such as pesticide application and septic tank density in vulnerable areas to reduce contamination risk;
- prevent threatening practices such as bulk transport of particularly lethal contaminants through a high vulnerability area or their storage in high vulnerability areas;
- identify particular hydrogeologic settings where further reconnaissance might provide a better database for future groundwater resource assessments or another DRASTIC analyses.

Brown (1998a) applied several of the modifications to DRASTIC proposed by Brown (1998b) as appropriate for New Zealand river floodplain aquifer systems. For vulnerability assessments of the high transmissivity multi-layered confined aquifer systems that are typical of New Zealand floodplains, natural vulnerability rather than specific vulnerability is the important consideration.

2.4 Adaptation of DRASTIC concepts to groundwater-mediated transport of contaminants.

There are two important requirements for applying DRASTIC methodology to identify surface water – groundwater interfaces where groundwater contamination might be a relevant factor for assessing ecosystem disturbances. These are access to a range of hydrogeological data, and to means of assessing the interaction of specific contaminants and the soil, vadose zone and aquifer media factors. The first of these requirements depends on the availability and accessibility of databases. The second requirement depends on availability of specialist research, resources and expertise. The experiments carried out for this study and reported in the subsequent sections are designed to enable assessment of the transport of metals and BTEX through a range of significant New Zealand subsoil types overlying typical New Zealand aquifers.

Depth to water table

Depth to water table is an important factor (weight of 5) for all aquifer categories (except confined aquifers) in the DRASTIC index because a shallow water table and potential for discharge of contaminant almost directly into groundwater results in high vulnerability. Conversely a deep water table and the overlying cover of soil, vadose zone and confining strata (for confined aquifers) can reduce the vulnerability of the groundwater to contamination.

For metamorphic, sedimentary and volcanic rocks the water table depth is often difficult to measure. Factors such as porosity for volcanic rocks and secondary structural features such as fractures, joints, faults and dissolution channels for all rocks are important factors in assessing contamination vulnerability as they provide preferential infiltration or flow paths regardless of water table depth. This applies to the sub-soils also. The rate of movement of contaminants is increased where such flow paths are significant, and so the risks to downstream ecosystems become very high. Preferential flow paths due to soil cracking, deep rooting, construction activities and the like are responsible for such accelerated rates of transport. These can vary seasonally – a dry summer in the Waikato in 1999, for instance, showed increased movement of pesticides into a soil profile by factors between 100 to 1000 (Bob Lee, Landcare Research, pers. comm. 1998). If preferential permeability is likely to be significant at a site, the highest possible factor should be ascribed to this vulnerability factor for the aquifer.

The depth to water table is obtained by measuring water levels in wells. If long-term groundwater level monitoring is available, this will provide a range of water level fluctuations through the seasons and years, which may indicate significant variation between winter highs and summer lows. These can translate into different DRASTIC indices and hence different vulnerabilities in different seasons. Summer low water levels are usually related to reduced or no recharge of the groundwater system or to high abstraction rates for water supply.

Piezometric contours are used to determine the direction of groundwater flow which is important for the derivation of subsurface connections to recharge sources, contamination sources and likely discharge areas. This applies for floodplain and river valley aquifer systems, but may not always be appropriate for volcanic, sedimentary and metamorphic rocks where highly permeable channels (lava or limestone caves), volcanic rubble or scoriaceous layers or fracture zones may occur. In these situations preferential infiltration routes are promoted with, tenuous but rapid flow compared with the main body of rock, and connection to surface water at sites seemingly isolated from the original recharge zone. In these cases it may be difficult to reconcile groundwater flow direction as suggested by topography and regional piezometric contours with the flow induced by localized permeability.

The depth to water table is not relevant to groundwater contamination vulnerability in confined aquifers, particularly in the extensive layered fluvial gravel aquifer systems underlying floodplains, as the confined groundwater will be under pressure with a positive hydraulic head. Depth to the top of the confined aquifer (base of the confining layer) is more relevant as it embodies the degree of protection provided by the overlying confining strata and the distance from the more vulnerable unconfined sector of the aquifer. Herr (1990) in an application of DRASTIC to the predominantly confined Floridian Aquifer System, Florida, USA designated a series of depth to aquifer ratings to replace the depth to water index. This practice was adopted by Brown *et al.* (1994) (Wellington) and Brown (1998b) (Heretaunga Plains). Well logs provide information about the thickness of the confining layer and this can be collated as a depth to confined aquifer map for input to a modified hydrogeological setting.

For floodplain aquifer systems where confined aquifers underlie the water table aquifer, or for different rock type confined aquifers underlying shallow water table aquifers, it may be worthwhile carrying out separate vulnerability assessments for the water table unconfined aquifer and the underlying confined aquifer system. This will provide a comparative range of vulnerability that extends the relevance of the DRASTIC application.

Recharge

Unless the pollutant is in liquid form and can flow independently of water, movement of contaminants will require their dissolution in or mobilisation by infiltrating water recharging groundwater and subsequent transport by groundwater. Recharge to aquifers can be from any of a number of sources. Recharge mechanisms are either direct or indirect (Lerner *et al.* 1990). Direct recharge is water added to the groundwater reservoir by direct vertical percolation through the unsaturated zone in excess of soil moisture deficits and evapotranspiration. Indirect recharge is percolation of water to the water table through the beds of rivers, lakes, water races and canals, and the mixing of groundwater from adjacent or underlying aquifers.

Rain recharged unconfined aquifers usually have a DRASTIC vulnerability index rating higher than river recharged aquifers because on a local scale infiltrating rain volumes are intermittent and small compared with high volume localised river recharge and contaminant dilution and dispersion is more effective with the high volume river recharge regime. Rain recharge also varies in response to seasonal climate and land-use patterns.

For river valley and floodplain aquifers, rain recharge is most effective where the water table is high over a wide area, and soils and vadose zones are porous and thin and give rise to high contamination vulnerabilities. The porous nature of volcanic and limestone rock aquifers with surface outcrops or the presence of weathering features such as sink holes, facilitate the infiltration of surface water. Although these conditions equate with a high contamination vulnerability, downstream dilution and dispersal can reduce the impact of contamination.

Sedimentary rocks (except limestone) with low permeability have low recharge rates which result in low vulnerability to contamination. Metamorphic rocks also generally have low permeability and low recharge rates, and hence low contamination vulnerability. However the presence of discreet zones of fractures and joints within metamorphic rocks can collectively allow significant water infiltration and movement of contaminants through the rock, and inhibit dispersal and dilution. Conventional DRASTIC under-estimates the vulnerability of fractured rock compared to unconsolidated aquifers (Rosen 1994).

Land usage in the area of groundwater recharge is an important influence on vulnerability to contamination. This was quickly discovered last century by the European settlers in towns like Auckland (porous volcanic rock), Gisborne (sand dunes), and Christchurch (gravel river channels and swamp). Household and animal waste soon polluted the shallow wells and springs that provided water supplies. Water-borne disease resulted.

Contamination sources include leaking sewers, leaking underground storage tanks, contaminated industrial sites, urban drainage and septic tanks. Groundwater contamination in shallow water table aquifers in rural areas has become a problem when settlement density and livestock numbers on farms have increased. High groundwater contamination potential for shallow water table aquifers are spread over all rock and sediment types, and the “flow through” affects all ecosystems and wetlands within or adjacent to cities and areas of intense livestock and horticulture production.

Floodplain aquifer recharge from rivers is normally restricted to a relatively short reach of the riverbed where aggradation and changes of course result in the riverbed overlying and intersecting former riverbed channels. Underflow beneath the riverbed discharges into these channels which form conduits conveying groundwater into the associated aquifers. The river-recharged aquifers have very permeable fluvial gravels and groundwater flow in the unconfined sector can be in the order of hundreds of meters per day. This translates into reduced risk of harmful contamination as dilution and diffusion of contaminants is effective dispersal processes. The river-derived groundwater recharges confined aquifers where the flow rates may be measured in terms of years, tens of years or hundreds of years. These are factors that DRASTIC does not weigh and thus have to be incorporated into modifications similar to those adopted for confined aquifers with regard to depth to water table.

For several New Zealand river-recharged floodplain aquifer systems (e.g. Heretaunga Plains, Hutt Valley, Wairau Plain, Christchurch) a proportion of the river-recharged groundwater into the unconfined portion of the aquifer emerges down plain as springs which form wetlands and can be the sources of coastal rivers. This occurs at the boundary of the unconfined and confined aquifer sector where the groundwater flow in the almost continuous gravel sequence of the unconfined aquifer is impeded by relatively impermeable clay, silt and sand of the interbedded marine postglacial and interglacial deposits. The potential for regional aquifer contamination in the unconfined sector and downstream contamination of wetlands and spring-fed surface water bodies would seem to be high. That such contamination has rarely been detected suggests that the dilution and dispersion characteristics of these high flow volume-high transmissivity river-recharged aquifer systems are sufficient to prevent contamination of ecologically significant concentrations. The DRASTIC recharge factor needs to be rated lower for these systems as a consequence.

The quantification of recharge volumes is extremely difficult due to spatial variation of critical factors such as geology and soil, and spatial and time variation of climate, hydrology, evapotranspiration and land-use. For the application of the DRASTIC index to ecological risk assessment, the recharge factor has to be considered in conjunction with the aquifer media and hydraulic conductivity hydrogeological factors, as all three factors are influenced by permeability.

Aquifer Media

By “aquifer media” is meant the unconsolidated or consolidated sediment or rock in which the aquifer occurs. The hydrogeologic properties of sediments and rocks have already been outlined above. The available groundwater is present in aquifers within the pore spaces of granular and clastic rock, within layers and cooling fabric of volcanic rocks, and in the fractures and solution openings of non-clastic and non-granular rock.

The rate of movement of groundwater depends on the permeability of the soil, vadose zone and aquifer. Permeability is a general term used to describe the ease with which a fluid can flow through the voids and pore spaces of a rock or sediment. Permeability depends on the porosity, pore size, void size and tortuosity of flow path. Permeable rocks such as gravels, and some limestones and basalts allow rapid groundwater flow at tens or even hundreds of metres per day. Relatively impermeable strata composed of clay and silt restrict groundwater flow to rates of less than a few metres per year. Fractured greywacke can be quite permeable while massive greywacke is relatively impermeable.

The aquifer media will normally be determined using a geological map, but often there will be areas where the aquifer media may be distinct and separate from the surface geology. An example of this is North Otago where volcanic rock aquifers underlie river floodplains and limestone deposits. The aquifer media is more accurately identified from well logs, and the regional distribution is achieved by cross-sections and 3D lithostratigraphic lattices constructed from well logs. The availability of geological maps and drillers’ well logs means that the aquifer media parameter can be fairly accurately described and weighted for the DRASTIC index. The application of aquifer media to the freshwater – groundwater interface gives an indication of the potential for groundwater contamination extending to surface water bodies. The combination of low permeability aquifer, shallow depth and water table, and intensive land-use, provide a high contamination vulnerability potential and conditions where associated ecosystems would be affected.

Soil Media

In the standard DRASTIC weighting system, soil media has a relatively low weight of 2 whereas the agricultural DRASTIC weighting system has soil media with the highest weight of 5. The latter is considered to be a more relevant weight assessment for application to New Zealand aquifer systems in the context of the effects on the ecosystems contacted by the groundwater.

Subsoils occupying the vadose zone are important as they separate the ground surface and aquifers. The role of the subsoil in inhibiting infiltration of contaminants to groundwater is poorly dealt with in the DRASTIC contamination vulnerability index model. The different subsoils and consolidated materials in the vadose zone result in a wide variation in the transmission of contaminants. Different contaminants exhibit a wide range of adsorption, diffusion, precipitation and solubility characteristics which affects their transmission through the subsoil materials. Because of the almost complete lack of information about soil properties a major part of this study was the measurement of the relative mobilities of the interactions between two classes of contaminants (heavy metals and BTEX) in significant New Zealand subsoils.

The type of subsoil media will normally be suggested by soil maps and described by accompanying text. Agricultural, engineering and other land-use activities which may affect their permeability as well as composition commonly disturb the soil and subsoil. Land-use maps are another data source that provide direct evidence of soil type and soil fertility, and indicate contamination vulnerability as associated with the land use activity. The type of crop grown on the land also influences the movement of water and associated contaminants through to the groundwater. Different crops and trees vary in the capacity of their root systems to take up water and contaminants, before the water and the water borne contaminants seep into the underlying vadose zone beyond the reach of tree roots. Tree roots, cracking from drying out and subsoil disturbances from activities such as drain-laying provide preferential routes for the percolation of surface and soil water.

Despite the large areas occupied by buildings, roads and sealed surfaces, urban and industrial areas are where most groundwater contamination problems arise due to these being where many potential contaminants are manufactured, stored, transported and used, in concentrated forms and large quantities. Contaminant leaks, spills and the by products of their use are often concentrated in surface runoff and can enter the subsoils through drains and soak pits. The presence of underground storage tanks and excavations for foundations, services, infilling and site levelling and landscaping all exacerbate the potential for infiltration of concentrated contaminants. It is common practice to remove the soil and topmost subsoil layers before areas are developed for urban expansion.

Soil is the product of the interaction of depositional, weathering, climate, biological and land use processes. The parent material from which the soil was derived affects the properties of the soil and can influence the associated chemical properties and composition of the water within and percolating through the soil. There is a wide range of soil parent material which include:

- *in situ* weathering of the parent rock;
- the *in situ* accumulation and subsequent alteration of vegetation to form peat or organic material within the soil;
- the transport by wind of dust to form loess;
- the localized accumulation of air fall volcanic debris or more distant wind transport of volcanic ash;
- the deposition by rivers and streams of the gravel, sand and silt material derived from the mechanical erosion and abrasion of the eroded clasts.

The properties of each soil are characterised by the site conditions – location, vegetation, climate, slope and land use. The subsoil is less directly affected by superficial processes, but it must be borne in mind that the site conditions required to characterise a site extend vertically downwards from the ground surface to the rock or sediment material from which the soil is formed, or on which it rests (Gibbs, 1980).

The texture and grain size together control the ability of a soil to retain or drain soil moisture, both saturated and unsaturated. The structure and density of a soil influence the progress of permeating fluids, and how the presence of clay minerals and organic matter influence the so-called leaching ability of a soil (the ability of a soil to transmit non-conservative pollutants). Slow percolation through a fine-grained soil material increases the possibility for physical, chemical and biological reactions. The processes and reactions associated with the infiltration of water through the sub-soil can continue in the underlying vadose zone.

In addition to the variables associated with different soils, soil materials and soil structure, different contaminants also introduce another range of variables. Contaminants exhibit different retention and decay processes in different subsoil and aquifer media, with the consequence of a very large matrix of variable transport properties. Elapsed time since contaminant application is also an important variable, as liquid contaminants can travel further, but also because the nature of binding of contaminant species changes with time – usually to become stronger (e.g. McLaren *et al*. 1994). Travel time will include the time taken for the contaminant to seep through the soil and the vadose zone and will range from minutes to days for porous sediments and mobile contaminants, and from weeks to years for silt–clay sediments. The density of the contaminant is another variable factor influencing retention and dispersion – contaminants that are lighter than water will float on the water table and require different remedial techniques to heavier contaminants. Volatility is another important variable factor.

The solubilities of the contaminants have an obvious effect on the rate of transport, but even sparingly soluble components can be of serious environmental concern because the ecological effects require very low concentrations. High distribution coefficients favouring the sequestration in the solid phase certainly remove contaminants from the water phase but also act as reservoirs of the contaminant for leaching into the groundwater for very long periods of time. This is of particular significance if the contaminant is toxic in very low concentrations.

The chemical properties of the subsoils are very variable and of considerable significance. Variable pH, redox conditions and complexing can change the efficiency of water and contaminant percolation, absorption and release processes. Cation exchange attenuates some pollutants within the subsoil zone. Besides introduced contaminants, chemical reactions within the subsoil and aquifer media can produce toxic substances in groundwater. For example in Bangladesh and West Bengal, India, arsenic in groundwater is a product of desorption of iron oxyhydroxide grain coatings under reducing conditions in Quaternary age deltaic alluvial aquifers.

The chemical mobility experiments conducted in this project have resulted in relative ratings (see Tables 2.3 and 2.4 in Part 1) of very high, high, medium and low in different subsoil and for different metals. Because the boundaries between these ratings are logarithmic the factors used in calculating risks of the contaminant reaching the groundwater aquifer have been given values of 1000, 100, 10 and 1 respectively.

Topography

The topography of a site (i.e. slope) influences whether water and pollutants will preferably run off or remain on the surface long enough to infiltrate. The topography can be obtained from a topographic map or a GIS topographic database. In terms of the DRASTIC index, topography is weighted 1 for the standard rating factors and 3 for the agricultural rating. The effect of topography in flat areas such as floodplains is underestimated using standard DRASTIC and is more appropriately covered by using the agricultural DRASTIC weighting factor.

Impact of the Vadose Zone

The vadose zone (unsaturated zone) is above the aquifer or water table and below the soil proper. Along with the subsoil it is part of the regolith. The thickness of the vadose zone varies with the depth of the water table and the water table can fluctuate daily and seasonally and in response to groundwater recharge and abstraction. The vadose zone has an important role in the protection of groundwater from contamination and so it is given a heavy DRASTIC weighting. Along with recharge, this factor is the least quantified of the factors because it is difficult and costly to obtain the necessary data. Drilling of exploratory and monitoring boreholes, field and laboratory measurements and observations, and isotope studies are desirable for the assessment of the unsaturated zone. Investigative techniques often involve disturbing the vadose zone, which will disrupt the natural water movement and affect measurements and observations of the water content and properties.

As part of this study, eight sub-soils have been assessed for their relative ability to retain metals and BTEX. These experiments do not assess preferential pathways and other factors that may only operate in the undisturbed natural state. Water movement in the vadose zone is complex and the attenuation of contamination is largely dependent on pollutant pathways and residence times (Foster 1998). For unconfined aquifers underlying river floodplains, the vadose layer is usually gravel and sand deposits, which provide an almost unimpeded infiltration media for water and contaminants to unconfined aquifers. Jointing and fracturing within metamorphic rocks in the vadose zone provide preferential pathways for contamination to reach groundwater. As metamorphic rocks are generally chemically inert and combined with relatively rapid water infiltration there is not much scope for the processes such as biodegradation, neutralization, mechanical filtration, chemical reaction, volatilisation and dispersion which can remove contaminants from the infiltrating water.

Relatively impermeable layers of sediment or rock overlie confined aquifers and as such the impact of the overlying vadose zone is negligible. The recharge area(s) of the confined aquifer is generally remote from the contamination site, so except for the situations where the confined aquifer has be depressurised and/or the contaminant is of higher density than water, or where the aquiclude is disrupted (by wells, for instance), confined aquifers are not at significant risk from overlying contamination.

Hydraulic Conductivity

Hydraulic conductivity is a measure of the permeability of the aquifer. Hydraulic conductivity is controlled by the amount and interconnection of void spaces within the aquifer which may occur as a result of intergranular porosity, fracturing, jointing, bedding planes, and cooling and solution channels. The best data indicative of hydraulic conductivity is transmissivity which is obtained from pumping tests. Aquifer transmissivities derived from most pumping tests are generally understated, because the construction of the well and the design of the pumping test rarely satisfy all the requirements for obtaining the optimum results. In particular the complete aquifer interval is not normally screened, the well is often underdeveloped and the duration of the pumping tests is not long enough to allow achievement of equilibrium conditions. These pumping test deficiencies are common because even badly constructed and developed wells provide water supplies that satisfy the requirements of most well owners.

3. New contaminant transport risk evaluation system

The factors used in the DRASTIC vulnerability mapping system have been adapted and extended to provide a more relevant system to determine the risk of the transport of contaminants through the sub-soils and underlying groundwater aquifers. The factors are then combined in a unique way which bears minimal relationship with DRASTIC – for instance, instead of being added after weighting, the factors in the new model are multiplied together to reflect the more dramatic effects that specific circumstances – such as preferential flow paths caused by construction activities, drains etc - can have on the potential for transport of contaminants. Additionally, added factors cannot allow a null result, when for instance, the aquifer at risk is confined under the site and so there is no risk. Multiplied factors allow very small or zero results.

General Factors

1.1 **Rainfall**. The factors for the annual rainfall at a site are 10 for > 1 m/a, 2 for < 0.5 m/a, and 5 for values in between. The weighting given to this factor is 0.5.

1.2 **Direct Discharge**. Does the stormwater or other surface water in contact with the contaminated soil discharge directly to a drain, stream, river or seawater? If so, then the assigned value is 10, if not, then 1. This factor flags a direct contamination problem for surface waters from runoff, and is not included in subsequent calculations, so the weighting factor is zero by definition.

1.3 **Specific Factors.** Is there any significant factor (or factors) - other than those associated with the properties of the subsoils and aquifer materials and properties - which would enhance the mobility of the contaminant(s) in the subsoil and aquifer e.g. highly acidic, heavily flushed with water or detergent, long term spillage or a long time since large volumes of contaminated fluid were deposited on or in the ground? If there are any such factors, then the factor is 10, and if no, then 1. The weighting factor is unity. Note that this factor can be very important as the modification to chemical conditions and long time or large volumes greatly increase the potential for movement of the contaminant(s).

1.4 **Quantity.** This factor is used to calculate if the volume of the spillage will cause the direct interaction of the material with the groundwater. The assumption is made that if it does so, the contaminant must reach the groundwater. The weighting factor is thus unity. The assumption relies on the spilled material being at high concentrations or sufficiently mobile in the soils to not be completely held up by the soils and the subsoils.

1.5 **Area.** The area of the spill needs to be estimated so that it can be combined with the volume and depth to the water table. The actual area is entered or the descriptions small (<1 m^2), medium (between 1 and 4 m^2) or large (> 4 m^2) are allocated the factors of 1, 2.5 and 4 respectively. If the area isn't known, the value of 1 is assigned, on a precautionary basis. The weighting assigned must be unity as the factor is used directly in a calculation of the potential for the contaminating liquid to reach the groundwater.

1.6 **Substance of concern.** The metals for which data is available from this study and diesel and petrol have been included in the system. The table derived from the initial K_d values (for short duration spills) has been included in the calculations. The risk is related to the mobilities of the materials, and is the inverse of the distribution coefficient (K_d). the descriptors used are based on the following mobility descriptions:

- Very high for K_d values < 10
- High for K_d values 10 < 100
- Medium for K_d values 100 < 1000
- Low for K_d values > 1000

Site-specific factors

2.1 **Minimum seasonal depth to water table.** Depth to the top of the water table can vary considerably with rainfall or season. The shallowest depth provides the most conservative risk estimate. The cases of the aquifer being confined, or if indeed the water table is above the site present different scenarios, which are catered for in 1.2 or is not relevant, as is the case for the confined aquifer. The depth is inversely proportional to the risk of transfer to the groundwater, to the values allocated to the factor are 2 for depths > 10 m, 5 for intermediate depths between 2 and 10 m, and 10 if the depth is < 2m. This factor is one of the most significant in determining the risk or vulnerability, so it has a weighting factor of 1.

2.2 **Soil media physical integrity.**

1. This factor is included to ensure that some account is taken of the bulk permeability of the subsoils. In the extreme case it becomes somewhat redundant as if the material is impermeable, the aquifer must be confined. The assignment of values requires some judgement, so examples are given. The factor is assigned a value of 1 for very impermeable subsoils such as clays and mudstones, 5 for more permeable formations, and 10 for very permeable subsoils, such as those composed of coarse sands and gravels. The weighting is assigned at 0.5.

2. The presence of preferential flow paths through the subsoil to the aquifer enormously increases the risk of transference of contaminant to the groundwater, both by increasing the potential flow, and because they allow the bypassing of the much of the absorptive capacity of the subsoil. We have taken a precautionary approach such that if this factor cannot be assessed or is unknown, it is assumed that there will be, i.e. the highest value of 10 is assigned. Such preferential flow paths must be expected where there are known permeable layers, large deep tree roots, trenches, faults, wells, foundations, excavations, ground cracking, or where these may have existed in the past. If the answer to the question is negative, then the value assigned is 1, otherwise it is 10. The weighting given is 1, in recognition of the significance

that these can have for the transfer of contaminants.

2.3 **Aquifer media, hydraulic properties.**

1. This factor applies to the dominant aquifer material as opposed to that of the vadose zone or subsoil. In assigning values, a simplification has been made to relate permeability not only to flow rate but to tortuosity of the flow. A very tortuous flow path will maximise the opportunity for adsorption, precipitation or degradation of the contaminating species. The converse also applies – open channels as in limestone or fractured basalt, will minimise the opportunities for such processes. The process of dilution is also relevant but is accounted for in the distance factor (3.1). A weighting factor of 0.7 is applied.

2. Preferential flow paths in the aquifer, between the contaminated site and any surface receiving water, has the potential to dominate in allowing transport of contaminants, so a high weighting factor (1) is allocated. The precautionary principle is once again applied, in that if the situation is unknown, particularly in urban situations, the highest value is assigned (10). If there are no such preferential flow paths for all or most of the distance, then the value is held at 1.

2.4 **Impact of the vadose zone.** This factor is included to take into account the chemical adsorption properties of the vadose zone, or subsoil. The eight significant subsoils for which measurements have been carried out in this study are listed, as well as the situation where no subsoil is present i.e. the water table is very near the ground surface, or there is no fine-grained material between the surface and the water table, as for where clean coarse fill has been used on a site. The user should bear in mind that the chemical adsorption by a soil is principally determined by the particle size and the organic content of the soil. The chosen soil type should be the most absorbent of those on the list, where some uncertainty between types exists.

2.5 **Aquifer materials.** The chemical absorption properties of the aquifer material, as distinct from the subsoil, are also relevant. A selection is to be made of the description of the material which most closely describes the bulk of the material in the aquifer, but if there is a significant proportion of a more absorptive material in the aquifer, its properties should be used, or some interpolation of the values made. The highest value (10) is allocated to fractured or channel eroded rock, such as fractured basalt or channels in limestone. If the aquifer contains fine-grained sands and silts, the assigned value should be 4, for volcanic ash, clays and/or weathered rock the value should be 3, and the lowest value of 2 is assigned to formations with a high organic content. The weighting assigned is 0.5.

Other factors

3.1 **Distance.** The longer the distance that a contaminant has to flow in an aquifer from source to discharge the more opportunity exists for mixing and dispersion, dilution, adsorption, precipitation and degradation. The factor values are 10 for short distance (< 10 m), 5 between 10 and 30 m, 2.5 between 30 and 100 m, and 1 for distance longer than this. The weighting factor is 1.

3.2 **Sensitivity.** This factor takes into account the type and sensitivity of the surface receiving water. Dilution potential and ecological significance are accounted for. Water bodies with a large dilution (large river, open sea) is given the low value of 1, if the receiving water is a lake then the factor value is 2, if the receiving water sometimes has a low flow, or is a wetland, the value is 5, and if the receiving water has special ecological significance, or is a small pool, then the value assigned is 10. If there is no apparent surface receiving water then the assigned value has to be 0. The weighting factor assigned is 1.

Combining the factors.

The factors listed above are combined or used to assign default values, which are translated into statements about risk. The methodology for this has been tested against a number of examples from New Zealand situations and modified as a result. The resulting risk statements are to be judged as a conservative, or worst case, assessment, as that is the most responsible approach to take, in our view. It is much easier and cheaper to prevent and restrict potential damage than it is to repair it, particularly where large volumes of soil and aquifer material may have been affected.

The first operation selects the greater of the aquifer permeability and the preferential flow path factors (2.3.1 and 2.3.2).

The second operation indicates that the aquifer below is confined, or that the water pressure is upwards, so no contamination transport from the site is likely. The conclusion is based upon the depth to groundwater entry.

The third operation calculates the volume of free space in the subsoil (making the assumption that the free space (porosity) is 20% above the water table and compares it to the volume of contaminant. If the volume of contaminant is greater than the free space, or thereabouts, then the message is displayed to this effect.

The next operation multiplies all the relevant factor values by their weightings to determine if the contaminant is likely to reach the groundwater, and a statement evaluating the risk is called up depending on the value calculated. The sequence of these statements is – “special factors are likely”, “will”, “may”, “may possibly”, and “will probably not” result in the contaminant reaching the groundwater.

The final operation multiplies all the relevant factor values by their weightings to determine if the contaminated groundwater is likely to reach a surface ecosystem and cause problems for that ecosystem. A statement evaluating the risk is called up depending on the value calculated. The sequence of these statements is – “high”, “some”, “little” and “no” risk of transfer to a surface water ecosystem.

The statements are subjective judgements and may need to be recalibrated as more examples come to hand. They have been tested against a small number of examples of systems where contaminated sites and the aquifers below have been explored or are well enough known to be able to make assumptions about the relevant factor values. It is unfortunate that the most well explored situations cannot be applied, as the data is not publicly available.

4 References

Aller, L.T., Bennett T., Lehr, J.H., Petty, R.J. and Hackett, G. 1987. A standardised system for evaluationg groundwater pollution potential using hydrogeologic systems. U.S. E.P.A./600/2-87/035. Washington, DC. 455 pp.

Bates, L. and Otto, C. 1998. An overview of integrated groundwater vulnerability assessment: methods, interpretation and best use. *In:* Groundwater sustainable solutions (Weaver, T.R. and Lawrence, C.R. *eds.*) International Groundwater Conference, International Association of Hydrologists, Melbourne, Australia, Feb. 1998, 795-805.

Brown, L.J. 1998a. Groundwater contamination vulnerability map of Heretaunga Plains, Hawkes Bay. Institute of Geological and Nuclear Sciences Client Report. 62796D.

Brown, L.J. 1998b. A review of groundwater vulnerability assessment and its application to New Zealand floodplain aquifer systems. *In:* Webb, T (ed) Issues in aquifer protection: Proceedings of the Workshop, Lincoln, N.Z. March 1998.

Brown, L.J. and Gregg, R.C. 1994 Groundwater in New Zealand N.Z. Mining **14** 11-14.

Brown, L.J., Kroopnick, P.M., Lillico, S.B. and Wood, P.R. 1994. Compilation of a groundwater contamination vulnerability map of the Wellington Region. Institute of Geological and Nuclear Sciences Sciences Report. 94/43.

Close, M.E. 1991. Pesticide contamination of groundwater in New Zealand. *In*: Peters, H.J. (*ed.*) Groundwater in the Pacific rim countries Proc. Symp. Am. Soc. Civil. Eng. Honalulu, U.S.A. July 1991 168-174.

Close, M.E. 1993. Assessment of pesticide contamination of groundwater in New Zealand. 1. Ranking of regiona for potential contamination. *N.Z. J. Marine and Freshwater Res.* **27** 257-266.

Fetter, C.W. 1993. Contaminant Hydrogeology. McMillan Publishing Co, New York.

Foster, S.S.D. 1998. Groundwater rcharge and pollution vulnerability of British aquifers: a critical overview. *In:* Robins, N.S. (ed) Groundwater pollution, aquifer recharge and vulnerability. Geological Society Special Publication no. 130, The Geological Society, London. 7-22.

Gibbs, H.S. 1980 New Zealand soils – an introduction. Oxford University Press, Wellington, New Zealand 115 pp.

Hadfield, J. and O’Regan, P. 1994. A DRASTIC/GIS evaluation of groundwater contaminant potential of the Hauraki Plains. N.Z. Hydrological Symposium, Hamilton, N.Z.

Henderson, J. 1937. Underground water, Wellington Hospital. Annual Report, NZ Geological Survey **31** 23.

Herr, J.W. 1990 Application of DRASTIC ground water pollution mapping methodology to the SFWMD. South Florida Water Management District, West Palm Beach, Florida, Research and Evaluation Department, Water Quality Division. Technical Pub. 90-02.

Lerner, D.N., Issar, A.S. and Simmers, I. 1990. Groundwater recharge. A guide to understanding and estimating natural recharge. Heinz Heise, International Contributions to Hydrogeology **8**.

Margat, J. 1968. Vulnerabilite des nappes d'eau souterraine a la pollution. Bases de la cartographie, (Doc.) BRGM, 68 SGL 198 HYD, Orleons, France.

McLaren, R.G., Carey, P.L., Cameron, K.C., Adams, J.A. and Sedcole, J.R. 1994 Effect of soil properties and contact period on the leaching of copper, chromium and arsenic through undisturbed soils. Trans. 15th World Congress Soil Sci., Acapulco, Mexico, vol. 3a 156-169.

Rosen, L. 1994. A study of the DRASTIC methodology with emphasis on Swedish conditions. *Groundwater* **32** 278-285.

Sheppard, Zhan, X., Rait, G. and Chagué-Goff, C. 1999. Leachate movement from unlined landfills in fractured rock. Institute of Geological and Nuclear Sciences Science Report 99/7.

Webb, T.H. and Lilburne, L.R. 1999. Use of the LEACHM model and the DRASTIC index to map relative risk of groundwater contamination by pesticide leaching. J. Hydrol. (New Zealand) **38** 271-288.

Zaporozec, A. 1987 Groundwater pollution potential of Barron County, Wisconsin. Wisconsin Geological and Natural History Survey, Madison, Wi, map 87-2 scale 1:100,000.