COST Action 620

Vulnerability and Risk Mapping for the Protection of Carbonate (Karst) Aquifers

Final Report

François Zwahlen (Chairman, Editor in Chief)

Summary

The biggest issue to face European Hydrogeologists over the last decade has been the need to protect the quality and quantity of groundwater resources. The European Water Framework Directive published on 23rd October 2000 reinforces the efforts that have been made, requiring member states to develop and implement plans to maintain and improve the aquatic environment. This directive puts into context the work of the COST Action 620, which began in 1996 and has lasted 5 years.

Our Action, entitled "Vulnerability and Risk Mapping for the Protection of Carbonate (Karst) Aquifers" was tasked with the development of an improved and consistent European approach for the protection of karst groundwater. The Action saw the production of new tools to assist in the management of karst areas and their water resources. One of the main criteria for these tools was to ensure that their transparency and ease of use, whilst primarily for expert, still reflected the needs of practical hydrogeologists. As a result, the participants of the Action worked hard to establish common approaches for delineating vulnerability in the field. The identification of the most vulnerable recharge areas where protection measures must be at their greatest is critical, whilst greater flexibility in land use planning can be considered in those less vulnerable areas.

Action 620 brought together experts from several disciplines including; hydrogeology, karst geomorphology, environmental chemistry and microbiology, all having specialist knowledge of varying aspects of karst aquifers. These specialists brought together their expertise to holistically consider the specific behaviour of these aquifers and their particular sensitivity to anthropogenic impacts. The fact that karst groundwaters are extensively used for drinking water supply in most countries with Europe is remarkable. The importance of these vulnerable karst groundwater resources for potable supply emphasised the importance of the work undertaken by the Action, so as to protect its quality for future generations.

The concept of groundwater vulnerability can be approached in many ways and the groundwater community's approach to it could be considered to be in a state of constant flux. As a result it was important to consider precise definitions in order to reach a level of agreement across the participating countries. Particular areas of debate included; the usefulness of intrinsic and specific vulnerability as concepts and the need to consider the risk of particular contaminant scenarios. The work undertaken by the Action was both interesting and difficult, not least due to the regulations and practises carried out in the 15 participating countries and the varying points of view of the numerous experts.

By proposing new transparent procedures based on detailed knowledge of karst groundwater behaviour we believe we have made a significant contribution to the groundwater protection 'tool box', opening new perspectives for future development on this important topic.

Our results are presented in this final report, which is subdivided in two parts. Part A presents the new approaches that were commonly developed by the Working Groups of COST Action 620. Part A consists of 7 chapters: Introduction, Groundwater Vulnerability, Intrinsic Vulnerability, Specific Vulnerability, Hazard Mapping, Risk Assessment, and Data Collection and Validation. Part B presents Methods and Applications that were worked out by the various research teams that contributed to our Action.

The European Approach to vulnerability, hazard and risk mapping is based on an origin-pathway-target model, which applies for both groundwater resource and source protection. Origin is the term used to describe the location of a potential contaminant release. The target is the water, which has to be protected. For resource protection the target is the groundwater surface, for source protection it is the water in the well or spring. The pathway includes eve-

rything between the origin and the target. For resource protection, the pathway consists of the mostly vertical passage within the protective cover, for source protection it also includes horizontal flow in the aquifer.

A major task of the COST Action was to develop a general, non-prescriptive approach to **Intrinsic Vulnerability Mapping**, which could be adapted into methods appropriate for use individual karst areas of Europe. European karsts include alpine and lowland, Mediterranean and continental and management of karst groundwater also varies from country to country. Hence, flexibility had to be an essential attribute of the approach.

The European Approach uses four factors in assessing intrinsic vulnerability: Overlying layers (O), Concentration of flow (C), Precipitation regime (P) and Karst network development (K). The factors O, C and K represent the internal characteristics of the system, while the P factor is an external stress applied to the system. The O factor may comprise up to four layers – soil, subsoil, non-karst rock and unsaturated karst rock. The C factor recognizes that in karst areas the overlying protective layers may be bypassed by runoff, which is concentrated at or near the surface of the ground and which then enters the groundwater system via a doline or a stream sink. For resource vulnerability mapping, where the target is the top of the saturated zone, the factors O, C and P should be taken into consideration, while, in addition, the K factor should be taken into account for source vulnerability mapping where the target is a karst water supply such as a borehole or a spring.

The European Approach does not specify how the component factors should be measured or categorized or how vulnerability ratings should be established and thus it is not a methodology. However, the approach has been applied, using locally developed methodologies, at a variety of test sites in seven of the participating countries (see part B).

A final aim in developing the European Approach was that the approach, though karst sensitive, should not be completely karst centred to the extent that it could not be used in other groundwater environments. This aim has been realised to the extent that the P and O factors have universal applicability in assessing vulnerability, whilst the C and K factors relate to the particular characteristics of karst aquifer systems.

The intrinsic groundwater vulnerability assessment accounts only for the hydrogeological characteristics of the system but is, by definition, independent of the properties of specific contaminants. However, each contaminant or group of contaminants behaves differently in the different layers. These contaminants, due to their own physical and chemical properties, can be retarded or degraded during their underground transit.

Thus, COST Action 620 also proposes an approach to **Specific Vulnerability Mapping**, which combines two types of information:

- Information about the physical and chemical behaviour of contaminants (or groups of contaminants). These parameters have been obtained from the scientific literature. They are different for each contaminant, but common for all field applications.
- Information about physical and chemical properties of layers. Those are different for each layer and due to different layer combinations and particular properties (tectonics, karstification etc.) for each monitoring point of the assessed area.

The proposed approach has been conceived as an additional weighting based on the intrinsic assessment. This means that it has to be used in addition to intrinsic assessment.

Using the properties of the contaminants and those of the layers, the principle of the method is to determine the effectiveness of processes that can play a role in the attenuation (retardation and degradation) of the contaminant. It means that a process, likely to act on the concentration of the contaminant, can become effective if the conditions are met in a given layer. So, the processes that are occurring through the different layers will tend, in almost all the cases, to increase retardation and degradation of the contaminant.

Once the effectiveness of the process has been determined in a layer, the intensity of the process accounts the hydraulic conductivity, the thickness, which all together represent the transit time of water, and the rate of diffuse flow of the layers. Of course, water flowing in the conductive drains is only slightly submitted to the attenuation processes, due to its generally insufficient residence time in the layer. The rate of diffuse flow includes the part of water, which is flowing in the fine fissures and in the porous matrix, outside the main drains of the system. Mostly water transiting through the low permeable areas (diffuse flow) undergoes such processes.

Assessing the potential degree of harmfulness of **Hazards** to groundwater is the object of a logical, 7-step work plan which starts from a definition and inventory of hazards and leads to the eventual production of hazard maps. Throughout the elaboration of the work plan, Cost 620 recognized the essential requirement that hazard maps must be simple if they are to serve as efficient tools in planning and decision-making processes.

In the context of groundwater contamination, a hazard is defined as a potential source of contamination resulting from human activities taking place mainly at the land surface. Consequently, the hazard inventory starts from a differentiation between three main types of land use: infrastructure, agricultural and industrial activities. The main aim of the proposed hazard inventory is to cover all the various hazards that are considered relevant to groundwater and to allow, through a reasonable subdivision, the mapping, evaluation and assessment of the hazards in an economically feasible and practical manner.

Easy-to-use software, developed as a specific activity within Cost 620 and which can feed directly into common GIS systems, facilitates the collection of data on hazards. A mathematical algorithm is proposed for the calculation of the potential degree of harmfulness for each hazard. The algorithm considers weighting values to enable a direct comparison between the different type of hazards, a ranking procedure for hazards of the same type as well as a scheme to assess the likelihood of a contaminant release. According to the resulting Hazard Index, five Hazard Index Classes are defined and assigned a suitable colour for presentation on a map. To further assist the map layout, appropriate symbols and signatures (point, line or polygon) for each type of hazards are given according to their spatial properties.

Although discussed in more detail in Chapter 7, the work plan also incorporates the use of mapping techniques, including GIS and remote sensing, as well as the requirements of proper data evaluation for the purpose of hazard mapping. Finally, some further observations are presented, especially with regard to the scale effect of the hazard maps produced for the test sites.

The protection of our natural groundwater resources requires a sustainable groundwater management, which should be based on a comprehensive **Risk** analysis. With regard to a possible damage of groundwater the term “risk” is used for the likelihood of a specific adverse consequence. Following the origin-pathway-target model risk depends on three elements: (1) the hazards and their probability that a hazardous event occurs, (2) the vulnerability of the geological sequence and (3) the consequences for the groundwater. In a logic system risk assessment is split in two parts: step 1, “risk estimation”, is analysing the potential intensity of the relevant impact reaching the groundwater and therefore deals with point (1) and (2). Step 2, the “risk evaluation”, focuses on the adverse consequences. These depend on the groundwater sensibility, like flow condition, and on the ecological and economical value of the damages.

In order to separate clearly the different parts of risk assessment and to quantify their importance a “risk intensity index (RII)”, a “risk sensitivity index (RSI)” and a “total risk index (TRI)” were introduced.

In past groundwater studies, including the recent work of COST 620, risk assessment is dealing mainly with the risk estimation only. For land use planning or other tasks of groundwater management, like decisions on protection or remediation measures, it is essential to include also risk evaluation and to consider the total risk assessment. Especially for non-hydrogeological orientated decision makers, shareholders and utilities managers it is important to go beyond the vulnerability analysis and to come up with a comprehensive risk assessment.

For all these tasks a documentation of the regional risk distribution on the available groundwater resources is urgently demanded. Using GIS technology certain strategies and concepts for presenting risk maps are recommended, which were partially also transposed in the different COST test sites.

The **Quality of Data** is critical to the development of both hazard and groundwater vulnerability maps. Such data ensure that the conceptual models, on which any hazard or vulnerability map is based, are sound representation of the systems of concern. A comprehensive strategy for the collation and processing of collected data is also considered vital to the map-making process.

The introduction of both **Verification and Validation** procedures into such map making is essential if the final products are to stand up to examination and prove useful for both planning and decision-making. The methods used for verification and validation must be independent of the map-making process and can range from; physically testing the mapped area using such techniques as tracer tests, through to employing numerical models to ratify the conceptual understanding represented by the map.

Addressing data collection, collation and processing needs as well as designing appropriate verification and validation procedures at the beginning of any map making program will prove beneficial in terms of; the efficiency of the map making process and the robustness of the final product.

The concepts and methods proposed by COST Action 620 were applied and tested in twelve **Test Sites** in eight European countries. The test sites cover a wide range of geological, hydrological and climatic settings. Intrinsic vulnerability mapping was done in all test sites, and the PI method was applied most often (6 times). The other proposed methods of intrinsic vulnerability mapping (COP, LEA, VULK, Time-Input) were applied in each one country. Specific vulnerability maps were prepared for two sites; hazard mapping was done in four sites; risk mapping in two sites and validation of vulnerability assessment in only one site.

The test sites Sierra de Líbar (Spain) and Engen (Germany) represent the most comprehensive applications of the proposed COST 620 approach. In these test sites, different methods of vulnerability mapping were applied and compared: the PI and COP method of intrinsic vulnerability mapping, the proposed new method for specific vulnerability mapping, and two previously existing methods (EPIK and the German method). The vulnerability maps were combined with hazard maps to produce risk maps. In the test site Vaulion (Switzerland), intrinsic and specific vulnerability maps were done both for the groundwater resource and source. In this test site, the VULK model and mapping method was used and the vulnerability assessment was validated at selected points with tracer tests.

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Legend
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Editor: Member of the Editors Team
WG1: Member of Working Group 1
Leader: Leader of the Respective WG (two Leaders per WG)
Chairman and Vice Chairman of COST Action 620
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Part A – Methodology

1 Introduction

Water resources are vitally important for the future of humankind. COST Action 620 focuses on the protection of groundwater within karst aquifers to assure its quality for potable use and in supporting varied ecological habitats.

Groundwater from karst aquifers is among the most important drinking water resource in Europe: carbonate terrains occupy 35 % of the land-surface and a significant portion of the drinking water is abstracted from karst aquifers. In some European countries, karst water contributes 50 % to the total drinking water supply and in many regions it is the only available source of fresh water (COST 65, 1995).

Fig. 1: Carbonate rock outcrops in Europe (COST 65, 1995).

Groundwater resources are often highly vulnerable to contamination from human activity, none more so than those found in karst aquifers. As a result, appropriate protection measures must be put in place, a point recognised in the European Commissions' Water Framework Directive (2000).

The work of COST Action 65 highlighted the need to develop an integrated method that addresses the question of groundwater vulnerability and risk in karst environments. As a result, the Directorate General for Science, Research and Development of the European Commission set up COST Action 620. The objective was to develop an approach to "vulnerability and risk mapping for the protection of carbonate (karst) aquifers". This new Action, made up of delegates from 16 European Countries, worked from 1997 to 2003 in order to achieve this goal. The philosophy of the new Action was based on the requirements of the European Water Framework Directive (2000), which aims to establish a framework for community action in the field of water policy. The directive demands sustainable water use based on long-term protection of water resources.

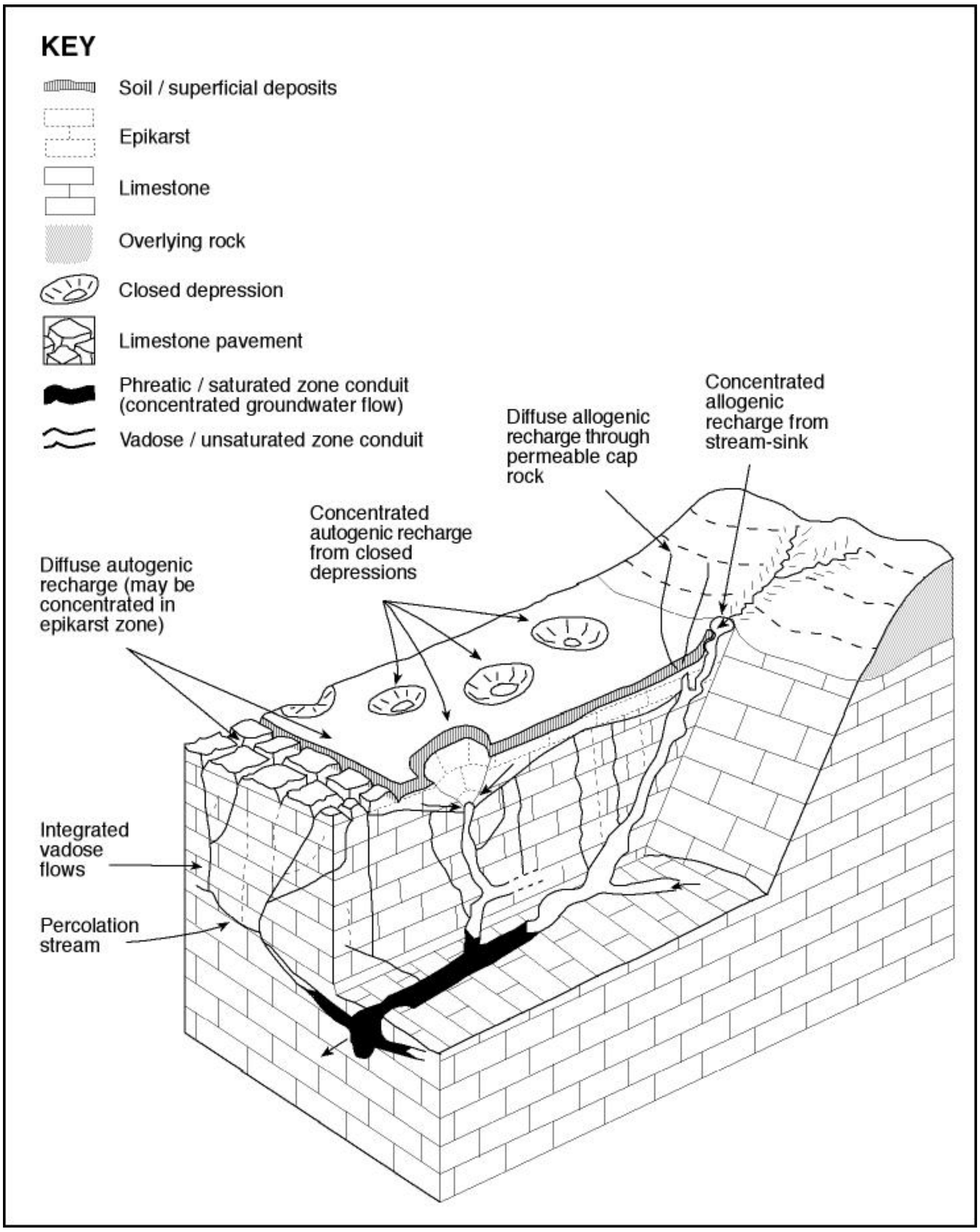

Fig. 2: Conceptual Model of a Karst Aquifer (Gunn 1986)

Karst aquifers are well known for their particular vulnerability to contamination arising from their special characteristics, like thin soils, point recharge in dolines, shafts and swallow holes, as well as concentration of flow in the epikarst and vadose zone. Such characteristics result in contaminants easily reaching groundwater, where they are transported rapidly in karstic conduits over large distances. As the residence time of contaminants in the system is often short and their interaction with the aquifer limited, many processes of contaminant attenuation like filtration and adsorption, as well as chemical and microbiological decay often do not work effectively in karst systems.

COST Action 620 established a program to address the objectives agreed by the Commission; however the complexity of the task soon became apparent. Experts participating in this Action soon realised that the philosophies behind each of the existing methods were often very different. This was is no small part due to the legislative background of the country within which the methods were developed. The variation in legislative backgrounds influenced not only the practical implementation of individual methods, but also the conceptual approach to dealing with the vulnerability question.

The diverse views help by experts within the Action often led to passionate debate, which has significantly enhanced the work undertaken, making it both interesting and profitable. A consensus of opinion has at times been difficult to achieve and there continue to be areas that remain open to debate. The Action worked both through Management Committee meetings held biannually in participating countries as well as Working Group and Task Group meetings, and short-term scientific missions, each serving to distil and refocus our ideas.

We believe that the work undertaken by Action 620 is a significant contribution to the development of sustainable long-term protection strategies for karst groundwater bodies. We achieved our objective by utilising 3 Working Groups (WG):

- WG1 Developed an approach to the mapping of intrinsic vulnerability of karst groundwater, based on sound scientific principles. As described, intrinsic vulnerability is a function of the geological, hydrogeological and hydrological properties of a system and is by definition independent of the properties of specific contaminants. The main intrinsic factors, which decide on the vulnerability of groundwater to contamination, are the overlying layers, flow concentration, the precipitation regime and the properties of the aquifer itself, in this case the degree of development of the karst network. Methods arising from the developed approach were trailed at national test sites in various karst settings across Europe.
- WG2 established a system to characterise the vulnerability of groundwater to specific contaminants or groups of contaminants. As described, specific vulnerability takes into account both the properties of the system and those of the contaminants, such as; nitrates, bacteria, chlorinated solvents and heavy metals. Important processes for specific contaminant attenuation include; cation exchange, biodegradation, precipitation, filtration and decay. A significant amount of effort has been put into the development of a new specific vulnerability methodology, whilst promising; the testing of this work is still in progress.
- WG3 worked on hazard and risk mapping. Hazards are activities and land-use practices that pose a threat to groundwater, such as agriculture, tourism, traffic and industry. Risk maps consider the activities that present the "risk" that threatens groundwater. They are obtained by synthesising the information presented on both hazard and vulnerability maps.

The results of the three working groups are presented in this final report. This work is founded on far more extensive studies within the 15 participating countries. The final report provides the reader with key references to these various national projects.

It must be appreciated that whilst vulnerability maps provide are a vital tool with which to protect karst groundwater, they remain a simplification. This is due to extrapolation and interpretation of data sets over relatively large areas, as well as the occasional use of data of unknown quality. These difficulties are offset by the validation process, which considers the success of the map to reproduce prevailing hydrogeological conditions. We would however advise that prior to embarking on any major development project within a karst area; a site-specific impact assessment is undertaken to a standard commensurate with the degree of risk posed.

As the years passed bye, time was running out, were our ambitions too great? Our aim to develop an integrated methodology for assessing both intrinsic and specific vulnerability, as well as allowing the degree of "risk" posed by certain activities to be evaluated and mapped, was well advanced but not yet finalised. We had however made significant advances and whilst the Action has now come to an end, work will continue. Surely future work will be less focused than during the years of COST Action 620 but will benefit from the strong network of collaborative workers, indeed friends, developed by this COST.

COST has provided a unique opportunity to bring together experts from across Europe to develop ways to protect our valuable karst groundwater resource. This opportunity has not only benefited research into vulnerability, but also each and every individual that has taken part.

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2 Groundwater Vulnerability

2.1 The concept of groundwater vulnerability

2.1.1 Background of the concept and definitions

The term 'vulnerability of groundwater to contamination' was introduced by MARGAT in 1968. However, the term 'vulnerability' is not restricted to groundwater but is used in a wide sense to describe the sensitivity of whatever to any kind of stress, e.g. the vulnerability of global climate to human impacts. As this report deals with the vulnerability of groundwater to contamination, the term is always used in that sense.

The concept of groundwater vulnerability is based on the assumption that the physical environment provides some natural protection to groundwater against human impacts, especially with regard to contaminants entering the subsurface environment (VRBA & ZAPOROZEC 1994). The term „vulnerability to contamination" has the opposite meaning to the term „natural protection against contamination" and the terms can be used alternatively.

VRBA & ZAPOROZEC (1994) emphasise that vulnerability is a relative, non-measurable and dimensionless property. They suggest distinguishing between intrinsic (natural) and specific vulnerability. The former should only depend on the natural properties of an area, while the latter should additionally take into account the properties of the contaminant.

COST 65 (1995) presents an overview on the various definitions of vulnerability that have been proposed until present. Most of them are quite similar. The COST Action 620 evaluated and discussed this issue and consequently proposes the following definitions:

- The **intrinsic vulnerability** of groundwater to contaminants takes into account the geological, hydrological and hydrogeological characteristics of an area, but is independent of the nature of the contaminants and the contamination scenario.
- The **specific vulnerability** takes into account the properties of a particular contaminant or group of contaminants in addition to the intrinsic vulnerability of the area.

The advantage of such qualitative and descriptive definitions is that the term 'vulnerability' is often intuitively understood, particularly by decision-makers in the planning process. Vulnerability maps are a means of presenting various complex hydrogeological properties in an integrated, comprehensible way. A map showing areas of different colour symbolising different degrees of vulnerability (or natural protection respectively) is easily to interpret and can be used as a practical tool for land-use planning, protection zoning and risk assessment.

However, there are also disadvantages in using a qualitative approach alone. A property, which is not precisely defined, cannot be derived unambiguously from measurable quantities. Furthermore, validation is problematic. Therefore, some ideas of how theses qualitative definitions could be linked with a quantitative point of view are outlined in section 2.2.

2.1.2 The origin-pathway-target model

The concept of groundwater vulnerability is based on an origin-pathway-target model for environmental management (Fig. 3).

- The **origin** (also referred to as source of contamination) is the assumed place of release of a contaminant. For vulnerability mapping, it is assumed that the contamination takes place at the land surface. This refers to hazards like cattle pasture and the spreading of pesticides or fertiliser. However, some hazards are located below the surface, e.g. a leakage in a sewerage system.

- The **pathway** is the flow path of a potential contaminant from its point of release (origin), through the system, to the point that has to be protected (target). For resource protection, the pathway consists of the mostly vertical passage within the protective cover, for source protection it also includes horizontal flow in the aquifer.
- The **target** (receptor) is the water, which has to be protected. For resource protection (see next section) the target is the groundwater surface in the relevant aquifer under consideration, for source protection it is the water in the well or spring.

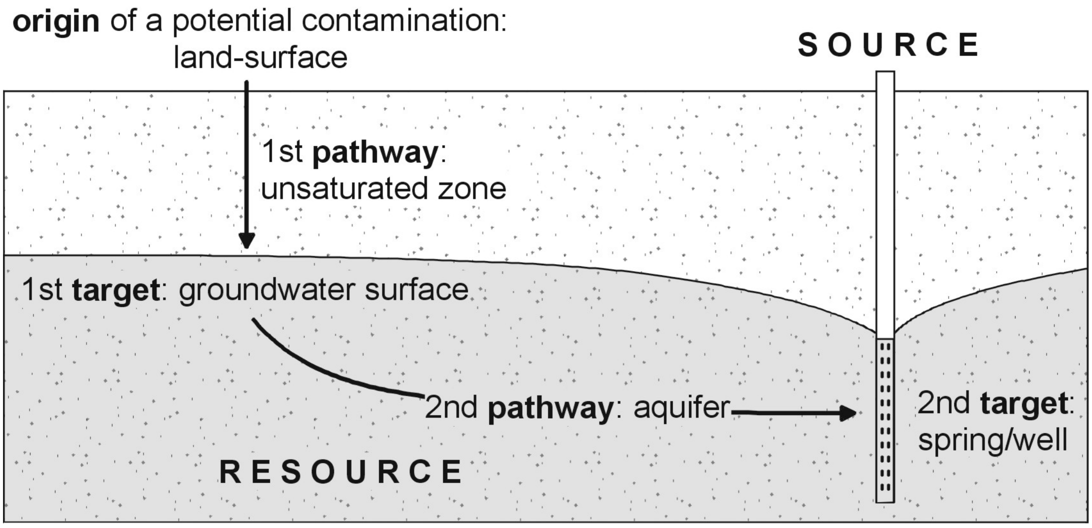

Fig. 3: The European Approach to groundwater vulnerability mapping is based on an origin-pathway-target conceptual model. The possible contamination event is assumed to originate at the land-surface. For resource protection, the groundwater surface in the aquifer is the target; for source protection, the spring or well is the target. For resource protection, the pathway consequently consists of the passage through the unsaturated zone (also referred to as the overlying layers); for source protection, it includes the passage through the aquifer (Goldscheider 2002). The adoption of this general scheme for karst aquifers is presented in the chapter on intrinsic vulnerability.

2.1.3 Resource and source vulnerability

In all European countries, groundwater is considered to be a valuable resource. The European Water Directive (2000) states that „water is not a commercial product like any other but, rather, a heritage which must be protected, defended and treated as such“. The directive establishes a strategic framework for managing the water environment and sets out a common approach to protecting and setting environmental objectives for all groundwater and surface water within the European Community.

The highest priority is to protect groundwater, which is used for drinking water supply. Thus, there are special regulations in all European countries, which aim on the protection of drinking water sources. The source might be a captured spring, a pumping well, a drainage gallery or any other groundwater abstraction point.

It is practicable to distinguish between resource and source protection, although both concepts are closely related to each other – it is impossible to protect a source without protecting the resource. Consequently, COST 620 proposes to distinguish between two types of intrinsic vulnerability maps: resource and source vulnerability maps. To do so, it is essential to define the target precisely (Fig. 3): For resource protection, the groundwater surface in the aquifer is the target. In case of confined and/or artesian conditions, the target is identical to the top of the aquifer under consideration and not to the potentiometric surface. The pathway consequently consists of the mostly vertical passage through the layers above the groundwater surface or surface of the aquifer respectively. These overlying layers are mostly unsaturated but may be temporally and locally saturated. For source protection, the water in the well or spring is the target. The pathway consequently additionally includes the mostly horizontal flow route in the aquifer.

2.1.4 The special situation in karst

The concept of groundwater vulnerability is applicable for all types of aquifers – granular, fractured and karst. However, due to the special properties of karst, it is essential to develop a concept, which takes into account the nature of karst. Until present, this was done by developing two kinds of method:

- methods specially dedicated to karst (e.g. EPIK method, DOERFLIGER & ZWAHLEN 1998);
- methods applicable for all types of aquifers but providing methodological tools for karst (e.g. PI method, GOLDSCHEIDER et al. 2000).

COST 620 proposes an approach of the second type, which is considered to be more applicable for the following reasons: (1) there are many intermediate conditions between a purely fractured and an extremely karstified carbonate aquifer; (2) there are transitional forms between granular and karst aquifers, e.g. karstified carbonate gravel or intensively fractured dolomites; (3) there are often several types of aquifers in one area which interact in some cases, e.g. a granular aquifer overlying a karst aquifer; (4) a method applicable for all type of aquifers is more likely to be accepted and applied by land-use planners and regional decision makers.

The following characteristics of karst systems are relevant with respect to groundwater vulnerability and should consequently be taken into account:

- Each karst system has its individual characteristics. Thus, the detailed hydrogeological investigation of a karst system is the precondition for vulnerability mapping.
- Karst systems are highly heterogeneous and anisotropic. Interpolation and extrapolation of field data is more problematic for karst areas than for other areas.
- Karst groundwater is recharged both by diffuse infiltration through the soil and by concentrated point recharge via dolines and swallow holes.
- The layers above the groundwater surface provide some protection. However, lateral surface or subsurface flow has to be expected in areas covered by low permeable layers. These lateral flow components maybe tributary to a stream sinking into the karst aquifer via a swallow hole and bypass the protective function of the overlying layers.
- The presence of an epikarst zone has to be expected. The functions of the epikarst are water storage and concentration of flow. The first process increases the natural protection of the system while the second process increases vulnerability. The structure and function of the epikarst zone are difficult to assess. A large portion of it is not visible at the land surface.
- Karstic aquifers are characterised by a dual porosity due to fractures and solutional voids (conduits) and frequently by a triple porosity due to the additional presence of intergranular pores (matrix). Groundwater storage takes place in the pores and fractures, while

conduits act as drains. As there are both extremely fast and slow flow components within a karst system, contaminants can be transported very fast or stored for a very long time.

- Karst systems are characterised by a fast and strong hydraulic reaction to hydrologic events. The temporal variations of the groundwater table often reach several tens of metres. In many karst systems, the groundwater table is discontinuous and difficult to determine.
- Karst catchments are often extremely large and hydraulically connected over long distances. Watersheds are often difficult to determine and variable in time, dependent on the respective hydrologic conditions. The catchments of karst springs often overlap and the flow paths proved by tracer tests often cross each other.

With respect to the main characteristics of karst environments relevant to specific vulnerability, two groups of properties play a significant role in the: hydraulic and mineralogical-geochemical properties. Combined, these properties are linked to the mobility and persistence of contaminants

In the karst system, hydraulic properties play a significant role in the migration of contaminants

- The existence of three different superimposed layers, with three hydraulic behaviours: epikarst is a high conductivity medium, with horizontal-dominant two-phase flowpaths. It collects infiltration and gathers it to the vertical conduits of the unsaturated zone. This medium contributes to the quick transit of the low-persistent contaminants to the water table.
- The existence of dual permeability, which differentiates residence times. Transit can be very rapid in conduits, slow in fissured blocks. The consequence is that conduits can quickly convey non-persistent pollutants, whilst blocks preserve conservative contaminants, enabling self-purification or reduction of the other pollutants. Furthermore, these voids control the flux of contaminants. Several types of contaminant (hydrocarbons) or particles (pathogens) require a minimum opening to transit through.
- Thus, the percentage of diffuse flow will be a significant parameter in the retardation and attenuation processes acting in the specific vulnerability.

Mineralogical and geochemical properties are also important. For example the presence of different minerals creates specific retention conditions:

- The carbonate medium restricts mobility of reactive contaminants such as phosphate, which precipitates as apatite, and heavy metals that can precipitates as carbonate species. The H+ proton, which can originate in acid rains and pedogenetic processes, is quickly buffered by the carbonate medium, enabling the production of Ca2+ and HCO3-. Generally, this carbonate medium is absent from the covering soils, which are residual formations, and this role is played by the subsoil and the epikarst. The pH value, despite its low variability in karst environments, due to the buffer effect of the carbonate medium, can play together with Eh a role in the solubility of inorganic metals.
- Clays play a double mineralogical and geochemical role: their specific surface enables adsorption of non-ionic substances (organics, bacteria) and their cationic exchange capacity (CEC) enable retention of cations, especially heavy metals. These clays exist in covering soils (residual clays), non-karstic covering formations (geological clays), and also detrial sediments which are present in the karstic network, both in the unsaturated and saturated zones.

- Organic matter, which is abundant in topsoil (which is often scarce and thin), and in the soil infillings of karren, play two roles: adsorption of organic contaminants and formation of ligands with metals.
- Eh is generally high in a karst environment (epikarst), which is widely open to the atmospheric reservoir. In this case, dissolved heavy metals can precipitate (oxides, hydroxides), organic matter and nitrogen species oxidise. Nevertheless, reducing conditions can exist in a carbonate environment, in different layers: in the soil, which can be hydromorphic and anoxic, and in the saturated zone which can be confined, when it is covered by impervious layers. In both cases, Redox potential becomes low and mobilisation of metallic oxides and hydroxides is possible. Nitrogen can also be reduced, in the best case as gaseous N_2, in the worst as NO_2^- or NH_4^+. Oxidizing or reducing conditions can select aerobial and anaerobial bacteria, which role can improve or worsen water quality.
- Temperature sometimes plays an active role. Several degradation processes are more active at higher temperatures, so seasonal variations may modulate the attenuation.

2.1.5 Vulnerability and the European Water Directive

The European Water Directive (2000) aims to establish a framework for community action in the field of water policy. The directive demands sustainable water use based on a long-term protection of water resources. The term „vulnerability" is only used in relation to coastal aquatic ecosystems. However, the idea of groundwater vulnerability assessment is indirectly included in the directive.

Annex II, Section 2.1 requires an initial characterisation of all groundwater bodies to assess their uses and the degree to which they are at risk. As part of this initial characterisation, Member States must „employ existing hydrological, geological, pedological, land-use, discharge, abstraction and other data" to identify a number of characteristics including the general character of the overlying strata in the catchment area from which the groundwater body receives its recharge. Annex II, Section 2.2 requires that those groundwater bodies which have been identified as being 'at risk', shall be assessed more precisely. This characterisation shall include information on (shortened):

- geological and hydrogeological characteristics of the groundwater body, including hydraulic conductivity, porosity, confinement and stratification of the groundwater body;
- characteristics of the superficial deposits and soils, including the thickness, porosity, hydraulic conductivity and absorptive properties;
- an inventory of associated surface systems, including bodies of surface water, with which the groundwater body is dynamically linked;
- the directions and exchange rates of water between the groundwater and surface systems;
- sufficient data to calculate the long term annual average rate of overall recharge.

Although this is rather an inventory list to characterise a groundwater body, it can also be used as a list of data that should be included in the approach to mapping groundwater vulnerability. The vulnerability assessment and mapping approach presented by COST 620 is directly linked to the information requirements of the Directive.

2.2 A quantitative point of view of the concept of vulnerability

2.2.1 The needs and advantages of a physically-based definition

As previously mentioned, vulnerability is often considered as a qualitative, non-measurable notion than as a quantitative property. This allows for some flexibility in the vulnerability assessment, while providing results, which are easily understood also by non-scientists.

However, the lack of a physically based precise definition also has some drawbacks. Vulnerability assessments are often subjective. If different methods are tested in one area, the resulting maps are often different and sometimes contradictory. The results are difficult to compare and, more fundamentally, to validate.

Consequently, there is a need for an examination of vulnerability concepts from a quantitative point of view, and for the establishment of clearly identified reference criteria for quantification, comparison and validation purposes.

2.2.2 Basis of a practicable definition of groundwater vulnerability

2.2.2.1 Introduction

To derive both a physically based definition and criteria for groundwater vulnerability assessment, one can start from the point of view that this concept should reflect natural mechanisms and processes that make the aquifer more or less sensitive to contamination. The applied definition of vulnerability should thus reflect the capacity of the aquifer to reduce any type of contamination. This reduction can occur mainly in two ways: (a) a decrease of contaminant concentration or (b) a decrease of pollution duration. In the case of an accidental contamination event, there are three practical questions, which a water user wants to have answered by a hydrogeologist (Fig. 4):

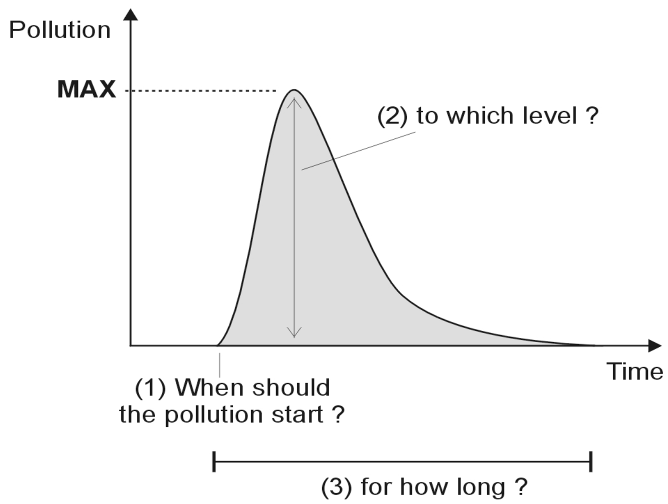

Fig. 4: The three questions that a water user wants to be answered in the case of an accidental contamination in the catchment.

1. how long does it take until the contamination reaches the target,
2. at what concentration level will the target be contaminated,
3. for how long will the contamination last?

A method of vulnerability assessment should consequently take into account the properties, which control the transit time of a contaminant from the origin (land surface) to the target, the contaminant concentration and, in the case particularly of instantaneous pollution, the duration of the contamination at the target. The transit time mainly depends on the permeability, effective porosity, hydraulic gradient and thickness of the layers along the pathway. The contaminant concentration level depends on the attenuation capacity of the aquifer, and also on the proportion of recharge versus runoff – a significant portion of water and contaminants may leave the catchment via surface runoff. Any potential contaminant "breakthrough" can thus be mapped into a three dimensional graphic. The axes of this graphic hold transfer time, concentration level and duration criteria. However, it is clear that a vulnerability map is, by definition, independent from the contamination scenario. It should not only be applicable for an accidental (instantaneous) point contamination, but also for diffuse and long-term scenarios.

2.2.2.2 Definition of the cube axes

The cube attempts to show graphically how a contaminant input in the system is transformed on its pathway from the origin to the target. Only the quantity of contaminants, which actually enters the system, is considered within the cube concept. However, in reality a significant portion may leave the system via surface runoff. Three basic criteria have been defined: transfer time, duration and concentration level. These criteria have to be specified for their use for intrinsic vulnerability assessment. In the following sections, it is explained how the axes of the vulnerability cube are defined and how field observations can be plotted into this cube.

Theoretically, the **transfer time** and **duration** axes should be associated with the time of first arrival of a (conservative) contaminant and the whole pollution duration respectively. From a practical point of view, the actual time of first arrival and the end of the contamination are almost impossible to determine experimentally, as they depend on the detection limit or background level of the respective contaminant. It is thus suggested to use adapted definitions presented hereafter.

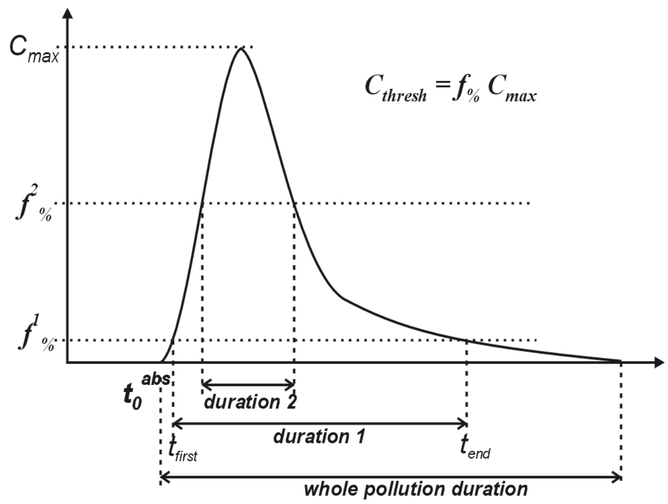

Fig. 5: Definition of the concentration limit as a percentage of the output maximum concentration

Pollution starting and ending times are associated with time flags for which the pollutant concentration is equal to some arbitrary threshold concentration C_{thresh}. In order to keep the intrinsic character of the definitions, this threshold concentration should be set equal to an arbitrary percentage $f_{\%}$ of the maximum concentration $C_{\max}$, observed at the target. The transfer time axis definition is associated with the first time t_{first} when $C = C_{thresh}$, while the duration axis definition is associated with the difference between starting time t_{first} and ending time t_{end}. According to the value considered for $f_{\%}$, the criteria of first arrival and duration are more ($f_{\%}^{1}$) or less ($f_{\%}^{2}$) severe (Fig. 5). Setting $f_{\%}$ to zero leads to the theoretical definition of pollution first arrival and whole duration.

Transfer times potentially range from zero (the pollution occurs directly at the target) to infinity (the pollutant never reaches the target). The pollution duration also ranges from zero (Dirac-type arrival at the target) to infinity (continuous arrival). As we are mainly concerned by orders of magnitudes, logarithmically transformed axes are more convenient.

The **intrinsic attenuation capacity** of the aquifer is evaluated on the basis of a comparison between the **maximum concentration** observed at the target $C_{\max}$ and the input concentration C_0 at the origin, providing an evaluation of the minimum attenuation capacity of the medium affecting the contaminant during its transit in the underground. The natural reduction of concentration is thus represented by the ratio between the maximum concentration at the target and the concentration at the origin:

$$C^* = C_{\max} / C_0$$

Concentrations are supposed to be relative to unit mass of contaminant. The relative concentration C^* is non dimensional and varies between zero (corresponding to complete disappearance of the pollutant) and unity (absolutely no attenuation, leaving initial concentrations unchanged).

2.2.2.3 Entering and moving in the cube

For an instantaneous point pollution scenario, the "entry point" in the cube is the corner characterised by zero values for transfer time and duration (practically, a short duration) and a relative concentration of one, representing the contamination Dirac function. As the contaminant evolves in the underground, it is affected by different intrinsic hydrodynamic and hydrodispersive mechanisms, altering progressively its spatial and temporal distribution. The corresponding evolution can be tracked simultaneously in the cube.

The main intrinsic processes are the following: advection (pollutant displacement at the mean effective velocity of groundwater), hydrodynamic dispersion (contaminant spreading around the mean advective position), physical attenuation (e.g. dual porosity effects), dilution (lowering of concentration directly related to mixing of different water fluxes) and recharge. All these mechanisms can be examined one by one, to highlight their consequences in terms of displacement in an opened-view of the cube (Fig. 6). For reasons of clarity, the respective influence of the different hydrodispersive mechanisms are added one by one:

- If the pollutant moves by **advection** only, there is no lessening in concentration. This means that concentration and duration are not changed along the flow path. In that case, the displacement in the cube follows the transfer time axis, the final point corresponding to the advective transfer time between the hazard and the target (point a).
- Due to **hydrodynamic dispersion**, the contaminant plume reaches the target sooner than by pure advection, the maximum concentration being lower and the duration longer. Compared to pure advection, the displacement along the transfer time axis is decreased, associated with a longer duration and a relative concentration less than unity (point b).
- **Physical retardation** is of kinetic nature and does not influence significantly the first arrival of a contaminant (BROUYÈRE et al. 1999). Due to storage in the immobile or less mo-

bile water, concentrations are lowered, while tailing effects can strongly enhance the contamination duration. In the cube, compared to advection and hydrodynamic dispersion, the final point reached is moved in a plane orthogonal to the transfer time axis, to lower relative concentration and longer duration (point c).

- **Dilution** effects should, by definition, only affect the concentrations. This means that such a mechanism moves the final point in the cube along the relative concentration axis to lower values of C^* (point d).

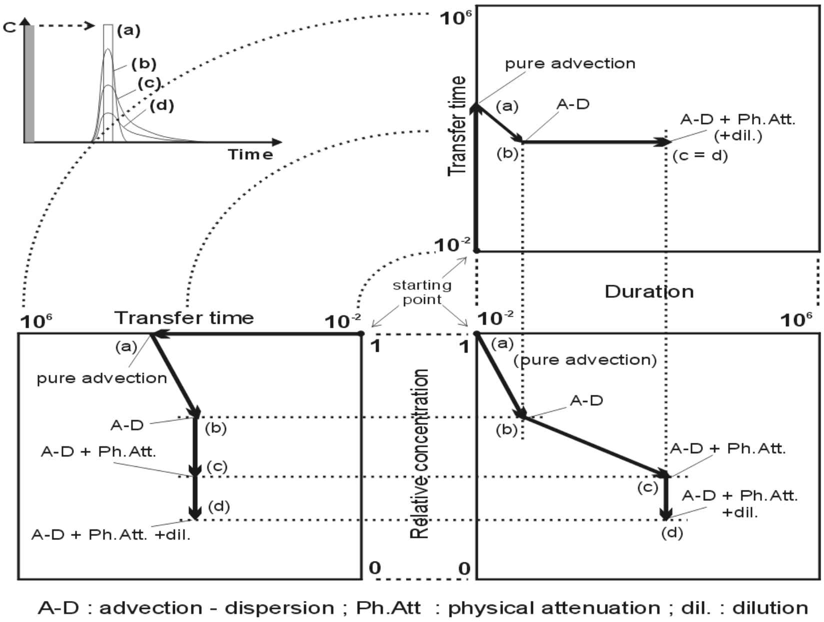

Fig. 6: Displacement in the cube according to the main intrinsic hydrodispersive mechanisms.

In reality, all mechanisms act simultaneously. The displacement in the cube is more complex, according to the length of the pathway followed between the origin and the target and the diversity of mechanisms affecting the contaminant behaviour in the underground. Conceptually, using the cube allows an intuitive tracking of the influence of each process, progressively, following the actual evolution of the contaminant in the underground.

For specific vulnerability, retardation and attenuation (chemical and microbial) will additionally modify the displacement in the vulnerability cube.

2.2.2.4 Definition of vulnerability classes

The final purpose is to provide a methodology to assess a value for the vulnerability index at each location on the land surface and a classification method of these indexes in several vulnerability classes. Different vulnerability classes have thus to be defined along the cube axis and within the cube. This is not a straightforward problem but some rules can be proposed, considering, as a first attempt, one criterion at a time.

If two points A and B in the catchment are compared in terms of vulnerability, it can be said that A is less vulnerable than B

- if the transfer time from A to the target is longer than the transfer time from B;
- if the maximum concentration of a contamination coming from A is lower than the maximum concentration of the same contamination coming from B;

- if the duration of a contamination coming from A is shorter than the duration of the same contamination coming from B.

These rules provide a way to define vulnerability classes along the cube axes. The class boundaries on the axes of transfer time and duration may be 3 days, 30 days, 1 year, 10 years and 100 years. The concentration axis may be subdivided into classes corresponding to ratios of 0.0001, 0.001, 0.01, 0.1 and 1. The highest vulnerability is associated with short transfer time, long pollution duration and low concentration attenuation.

However, the evaluation of three independent criteria for intrinsic vulnerability can lead to ambiguous situations (Fig. 7): For case a, point B is more vulnerable than point A because the transit time to the target is shorter, the maximum of concentration is higher and the contamination duration is longer. However, in case b, B is as vulnerable as A in terms of transfer time, B is more vulnerable than A in terms of maximum of concentration and A is more vulnerable than B in terms of pollution duration. As a consequence, it is not obvious to define which of the two points A and B is actually the most vulnerable.

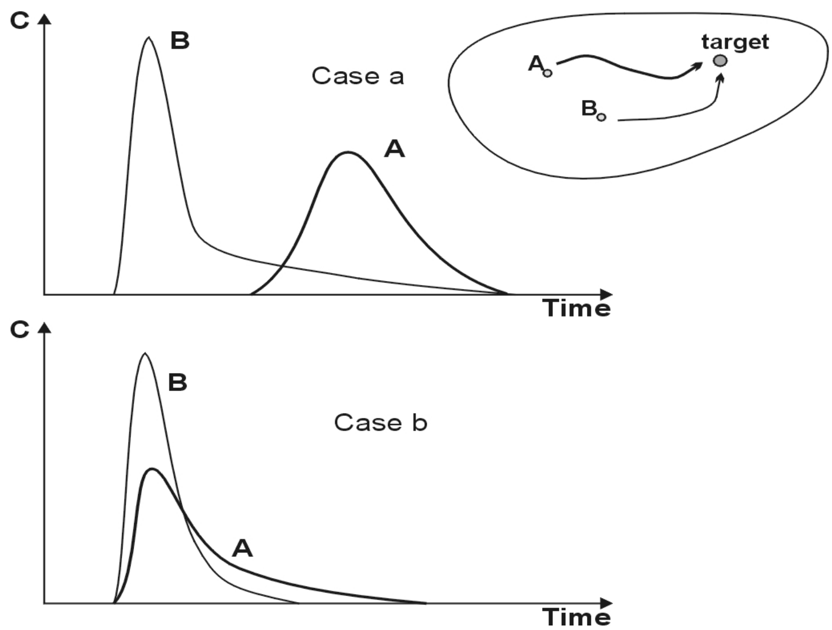

Fig. 7: Relative vulnerability of two points in terms of the different criteria

These examples show clearly that decisions have to be made in order to define vulnerability classes within the cube. Complementary criteria could be considered for making that choice. For example, remediation costs could be a decisive factor: a water producer may be more concerned by a long-term contamination at low concentrations, than by a highly concentrated, but short contamination.

2.2.2.5 Potential use of the cube

The cube allows the concept of groundwater vulnerability to be defined more clearly. It helps to select factors, which should be taken into consideration for vulnerability assessment and to decide how these factors can be combined. For example, based on the cube concept, a simple computer programme called VULK (JEANNIN *et al.*, 2001) has been developed in order to relate field observations to concentration level, transfer time and duration. This tool is described in part B of this report in the context of intrinsic vulnerability.

Furthermore, the cube can be used for validation purposes. This can be done by locating field test results (e.g. tracer experiments) or the results of analytic or numeric modelling in the cube and to compare this with the attributed vulnerability class.

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3 Intrinsic Vulnerability

3.1 Overview

The European Approach was developed as a conceptual framework for mapping the intrinsic vulnerability of karstic aquifers. The fact that the approach is non-specific and non-prescriptive reflects the need for the approach to be appropriate for use under the wide range of conditions (physical, administrative and cultural) that exist in the participating countries. Karst regions in Europe range from Alpine to lowland and from Mediterranean to continental to temperate oceanic in climatic type. Using the definitions and concepts presented in chapter 2 on groundwater vulnerability, Working Group 1 of COST 620 developed a conceptual framework for vulnerability assessment and mapping with special consideration of karst aquifers, the "European Approach". Four factors are considered: overlying layers (O), concentration of flow (C), precipitation regime (P) and karst network development (K). The O and P factors are the core elements necessary for resource protection mapping. The other factors are peripheral ('bolt-on' factors). P may be used to refine the precision of the vulnerability map. K may be used if source protection is the aim. Individual groups and individuals within the COST620 Action have taken this basic concept and converted it into usable methods appropriate to the particular karstic terrain in which they were working. A summary of some of these methods is given in this chapter but a full treatment together with examples of the application of methods at test sites throughout Europe is presented in part B of this report.

3.2 How the European Approach was developed

COST Action 65, essentially an information gathering and sharing project, formed the basis on which the present action was developed.

At the beginning of the COST620 Action, members of Working Group 1 (concerned with intrinsic vulnerability) completed a questionnaire to assess the present state of knowledge and practice in assessing the vulnerability of karst aquifers and the factors that were considered important in such assessment. Substantial agreement was obtained concerning those factors that were regarded as essential components of an adequate karst vulnerability mapping approach. The relationship of the karst groundwater system with surface features such as dolines was regarded as particularly important, as was the nature of recharge. It was accepted that only the EPIK method was wholly karst oriented and that other methods that had been used in karst were adaptations of methods designed for use in porous or fractured media. It was also decided that methods of valuing or weighting the selected parameters would not be attempted.

Subsequently small task groups were established to consider individual factors separately (for example; epikarst, protective cover, precipitation), to assess their relevance to the new approach and, if adopted, how the factors should be used.

In the early stages of development of the European Approach discussion was greatly influenced by the only existing karst-specific methodology, that of EPIK (Doerfliger and Zwahlen 1998). The K (karst network), P (protective cover) and I (infiltration) factors were adopted for the European Approach though much modified. Epikarst, though considered an important factor, was incorporated into the P factor rather than being a stand-alone element of the new approach as uncertainty existed as to the precise hydrological role of the epikarst. The most significant input to the European Approach came from consideration of the PI method developed by Goldscheider (2002) which used a minimum number of components (protective cover (P) and infiltration conditions (I)) to produce a karst-oriented GIS based system for mapping vulnerability. The method is described more fully in section 3.5 and in part B of this publication. The European Approach diverges from the PI method in that it does not specify data requirements in any detail, nor does it suggest how the factors should be combined.

A final aim in developing he European Approach was that the concept, though karst sensitive, should not be completely karst centres to the extent that it could not be used in other groundwater environments. This aim has been realised to the extent that the P and O factors have universal applicability in assessing vulnerability, whilst the C and K factors relate to the particular characteristics of karst aquifer systems.

3.3 The European Approach

3.3.1 Basic concepts

The intrinsic vulnerability of groundwater to contamination is, by definition, independent of both the contaminant nature and the contamination scenario. According to the COST 620 concept, when assessing intrinsic vulnerability of groundwater, three aspects are to be considered (DALY et al. 2002):

- advective transport time from the origin to the target;
- physical attenuation, e.g. by dispersion, dilution and dual porosity effects;
- relative quantity of contaminants which can reach the target (a portion of the contaminants may never reach the target but leave the catchment via surface runoff).

Vulnerability mapping is consequently based on assessing those properties of an area, which control these aspects. The advective transport time is mainly controlled by permeability, effective porosity, hydraulic gradient and distance between origin and target. A long transit time means that there is a long time for people to react to contamination events. Furthermore, most specific processes of contaminant attenuation are directly or indirectly dependent on the travel time (e.g. mortality of bacteria). The physical attenuation decreases the concentration, even for conservative contaminants and the effective or relative recharge decides on the quantity of water and contaminants that may actually enter the underground. A portion of the contaminants may leave the catchment via surface runoff.

3.3.2 Overview

The European approach to intrinsic groundwater vulnerability mapping and assessment suggested by COST 620 is a general and flexible framework rather than a prescriptive method. Several methods can be used and new methods can be developed within the framework of this approach, for example the methods described in chapter 4. According to the concept described in the previous sections, the approach aims at mapping and assessing those properties, which influence the travel time of a potential contaminant from the origin to the target, as well as the concentration level and the duration of a potential contamination.

Four factors are considered within the conceptual model presented in Fig. 3: Overlying layers (O), Concentration of flow (C), Precipitation regime (P) and Karstic network development (K). The factors O, C and K represent the internal characteristics of the system, while the P factor is an external stress applied to the system. For resource vulnerability mapping, the factors O, C and P should be taken into consideration, while the factor K should be additionally taken into account for source vulnerability mapping.

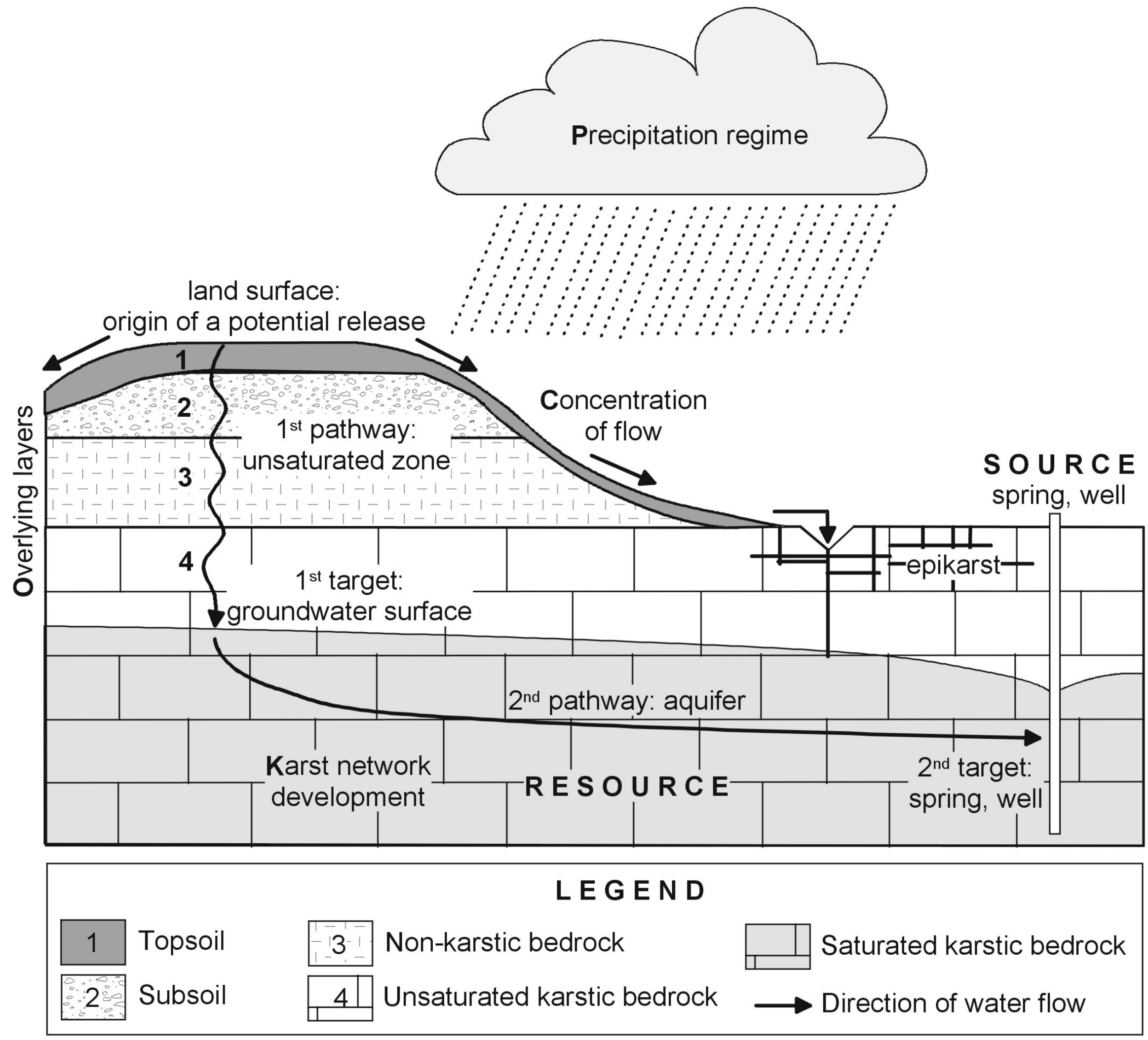

Fig. 8: The European Approach to groundwater vulnerability mapping is based on an origin-pathway-target conceptual model. The possible contamination event is assumed to originate at the land-surface. For resource protection, the groundwater surface in the aquifer is the target, for source protection, the spring or well is the target. The pathway consequently consists of the passage through the overlying layers for resource protection, and includes the passage through the aquifer for source protection. The main factors for vulnerability assessment are the **P**recipitation regime, the **O**verlying layers, the lateral **C**oncentration of flow and the **K**arst network development.

3.3.3 Overlying layers (O factor)

The overlying layers are those located between the land surface and the groundwater surface. They can consist of up to four types of layers: topsoil, subsoil, non karst rock and unsaturated karst rock. Some of these layers may be separated into several sub-layers. Each layer is not always present. Many karst areas only consist of two layers: unsaturated karst rock and topsoil; the latter is absent in the zone of bare karst. Granular (sand/gravel) aquifers only comprise the layers topsoil and subsoil. Fractured aquifers are often protected only by the topsoil and the unsaturated non karst rock.

The **topsoil** (layer 1) is the biologically active zone of weathering of the earth crust. It is composed of minerals, organic substance, water, air and living matter. It comprises the A and B pedological horizons. The topsoil is of major importance for specific contaminant attenuation but less relevant for intrinsic vulnerability assessment. Important characteristics to be mapped are the thickness, porosity and permeability; the two latter are mainly controlled by the grain size distribution, which can thus be used as a means to evaluate the protective function of the topsoil. Macropores play an important role as they enable the topsoil to be bypassed. The effective field capacity (eFC) can be used as a means to assess the protective function of the topsoil: A high eFC means a high capacity to store water and, consequently, to delay and attenuate contaminants. Indirect means are the soil type, vegetation and drainage density.

The **subsoil** (layer 2) is the granular, non-lithified material below the topsoil, for example Quaternary deposits made of gravel, sand, silt and/or clay or alluvium. The most relevant factors for the subsoil are thickness, porosity and permeability. Preferential flowpaths (macropores) are usually far less likely to occur than in the topsoil. The degree of saturation and the vertical hydraulic gradient in saturated, low permeability parts of the subsoil may be relevant for site specific situations. The lateral continuity of each layer should also be considered. In particular, low permeability strata can be bypassed if they are not laterally extensive, but occur in the form of lenses. A certain minimum thickness may be used as a criterion to decide whether a stratum is taken into account or not. The grain size distribution can be used as a means to evaluate the protective function of the subsoil. In many cases, the subsoil type as given on the geological map (e.g. „ground moraine“) will be the only available source of information.

The **non karst rock** (layer 3) consists of lithified, non karstified rocks, for example sandstone, schist, shale, basalt. The most relevant factors for the **non karst rock** are thickness, permeability, type and degree of porosity. Three situations can be distinguished: Fissured permeability is least protective (e.g. basalt with cooling fissures, granite with release joints); only intergranular porosity provides the best protection (e.g. a porous, non-fissured sandstone); the combination of both provides intermediate protection (e.g. a fissured sandstone with intergranular porosity). The most important means to assess these characteristics are geological information, particularly lithology (rock type) and tectonics. The density, width, continuity, spatial distribution, roughness and infilling of fissures control the hydraulic function and protection provided by the bedrock.

The **unsaturated karst rock** (layer 4) is the unsaturated (vadose) zone of the water bearing, karstified unit. The **epikarst**, if present, is a part of the unsaturated zone of the karst aquifer. It plays an important role in infiltration and percolation from the surface to the groundwater and consequently influences the protective function of the unsaturated zone. Two extreme situations can be defined: If the epikarst allows for diffuse infiltration, significant water storage and diffuse percolation, some protective function can be assigned to the **unsaturated karst rock**. In this case, it may be practicable to define a separate epikarst sub-layer. On the other hand, if flow concentration is the dominant process in the epikarst zone, it must not be treated as a protective sub-layer as it may enable the unsaturated zone of the karstic bedrock to be partially or totally bypassed. This situation is present for karrenfields with or without soil cover, drained by visible or hidden vertical shafts.

The characteristics controlling the protective function of layer 4 are thickness, permeability, degree and spatial distribution of karstification. The presence of a fissured and/or intergranular porosity increases the protective function of the **unsaturated karst rock**. In the case of strong flow concentration in the epikarst zone, the thickness of the unsaturated zone loses its relevance and the protective function of the whole layer might be insignificant. Indirect means of determining these characteristics are the analyses of surface and underground karst features, the lithology (purity, hardness, fracturing), tectonics, landscape history, analyses of surface waters, springs and swallow holes, the soil development and the vegetation.

Several existing methods provide possible assessment schemes for the O factor, for example the German GLA method (HÖLTING et al. 1995), EPIK (DOERFLIGER & ZWAHLEN 1998), the

Irish method (DoELG/EPA/GSI 1999), the PI method (GOLDSCHEIDER et al. 2000), and SINTACS (CIVITA & DE MAIO 2000). The PI method uses the four layers proposed by the European Approach.

In many cases, the O factor is the most important factor controlling the vulnerability – or natural protection – of groundwater to contamination. However, areas made of thick impervious formations often generate surface runoff. While in some circumstances this runoff may flow away and leave the area under consideration, in others it may enter the karst aquifer via a swallow hole. In this case, the overlying layers are bypassed and the O factor is insufficient to describe the vulnerability of the karst groundwater.

3.3.4 Concentration of flow (C factor)

The O factor may be sufficient to describe the vulnerability (or natural protection) of groundwater to contamination if all precipitation infiltrates diffusely into the soil and percolates through the unsaturated zone towards the groundwater. However, this is often not the case in karst systems, particularly in areas where the karst aquifer is confined by formations of low permeability. These areas often produce surface runoff which may sink into the karst aquifer at another place, e.g. via a swallow hole. In this case, the protective function provided by the overlying layers is bypassed. On the other hand, surface runoff, which does not sink underground but flows out of the karst systems, provides good protection to contamination.

Therefore, the C factor was introduced. It represents the degree to which precipitation is concentrated towards places where fast infiltration can occur. If infiltration occurs diffusely without significant concentration of flow, the C factor is not an issue, as the overlying layers are not bypassed. On the other hand, precipitation can be concentrated and the overlying layers can be completely bypassed by a swallow hole through which surface water and contaminants directly enter the karst aquifer. In such a case, the C factor is a significant issue in determining vulnerability. The degree of flow concentration depends on the parameters, which control the generation of surface runoff and/or subsurface flow, like slope gradient, surface properties (e.g. thickness, permeability and infiltration capacity of the soils) and vegetation, and on the presence of features which allow concentrated infiltration (swallow holes, areas with higher infiltration capacity). The C factor of the European Approach was adapted from the I factor (infiltration conditions) of the EPIK (DOERFLIGER & ZWAHLEN 1998) and PI method (GOLDSCHEIDER et al. 2000).

3.3.5 Karst network development (K factor)

For source vulnerability assessment, the mostly horizontal flow path in the saturated zone has to be considered. Therefore the K factor was introduced; it represents the degree of karst network development in the aquifer. It is based on a general description of the bedrock, giving a range of possibilities from non-karstified carbonate rocks with only intergranular porosity to karst aquifers with fast active conduit systems (Tab. 1). This K factor is similar to the one used in the EPIK method (DOERFLIGER & ZWAHLEN 1998). The K factor should be used together with a criterion of distance and/or travel time in the karst aquifer.

The means of assessing the karst network factor are the following: geology and geomorphology; cave and karst maps; groundwater-tracing results; pumping-tests results; spring hydrograph and chemograph analyses; remote sensing and geophysical prospecting; borehole data and geophysical-logging results; bedrock sampling and laboratory experiments; calibrated modelling results. Indicators, which may provide information on the underground characteristics, are drainage density, soil and vegetation.

Tab. 1: Classification of carbonate aquifers.

network	intergranular porosity	type of carbonate aquifer
absent	no	(no aquifer)
absent	low-high	**intergranular aquifer**
fractures	high low no	**fractured aquifer**
solutionally enlarged fractures	high low no	**karst aquifer**
slow active conduit network	high low no	**karst aquifer**
fast active conduit network	high low no	**karst aquifer**

increasing degree of flow concentration within the aquifer

3.3.6 Precipitation regime (P factor)

The P factor considers not only the total quantity of annual precipitation, but also the frequency, duration, and intensity of extreme events, which can have a major influence on the type and quantity of infiltration and, consequently, on the vulnerability.

A large amount of precipitation – together with favourable infiltration conditions and limited evapotranspiration – causes a high recharge rate, a fast percolation through the unsaturated zone and, consequently, a fast contaminant transport. However, a larger quantity of recharge also means greater dilution and a shorter duration of pollution. Extreme precipitation events lead to significant surface runoff and lateral subsurface flow (interflow), which may sink into the karst aquifer via a swallow hole. Extreme events also allow for fast contaminant transport in the karst network. The P factor is consequently an external stress, which influences the O, C and K factor.

In many cases, there is not a large variation of the precipitation regime within one catchment, but there may be large differences between different test sites in different climatic zones. Thus, the precipitation factor may not be an issue on the catchment scale, but it is relevant on a national or European scale.

3.4 Resource and source vulnerability maps

The four factors of the COST 620 approach can be combined in order to create resource and source vulnerability maps. For resource vulnerability, the groundwater surface is the target and ~~the~~ horizontal flow in the aquifer is not considered. Thus, the resource vulnerability map takes into account the O, C and the P factor (Fig. 9). For source vulnerability, the drinking water spring or well is the target, and the horizontal flow in the aquifer has to be considered. The source vulnerability map shall consequently be obtained by a combination of the factors O, C, P and K. The source and resource vulnerability map can be used as a basis for the delineation of source and resource protection zones respectively. Together with a hazard map, it can also be used for risk assessment.

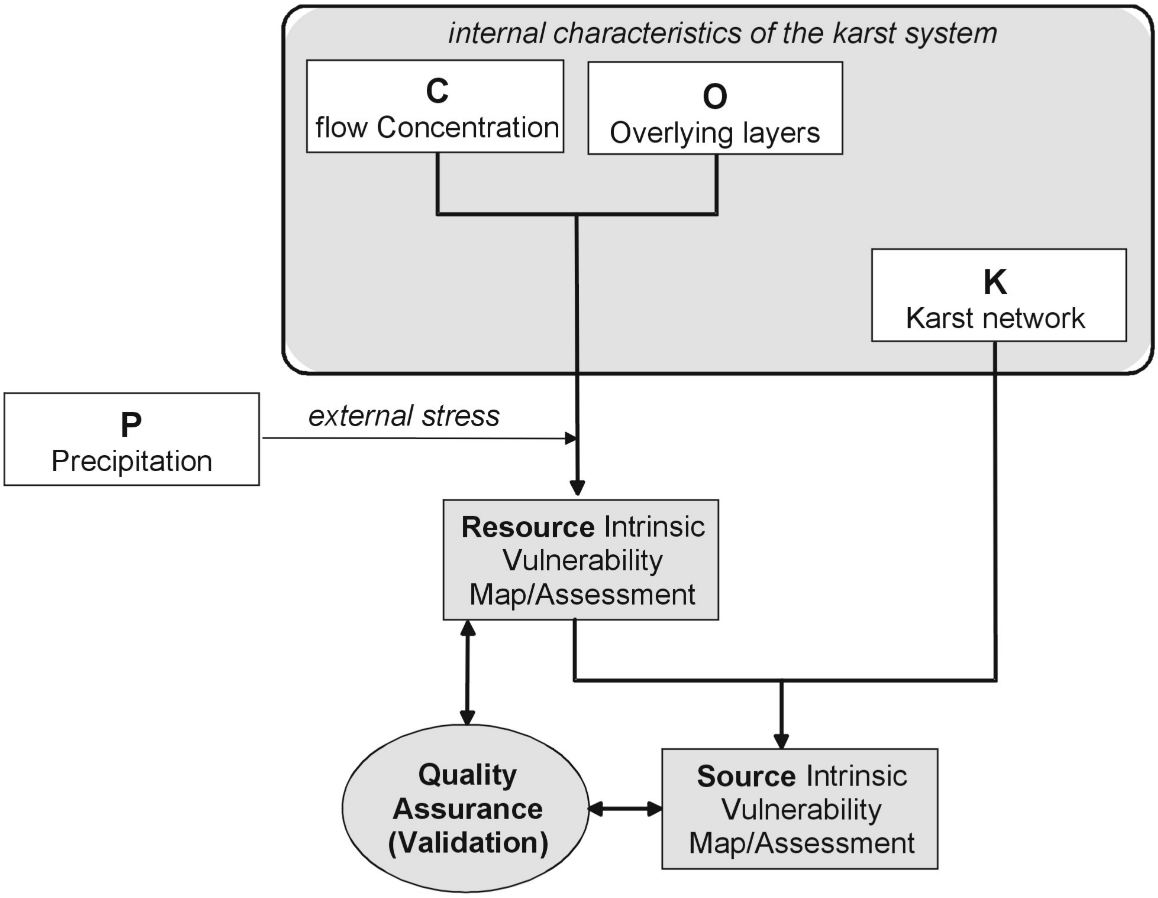

Fig. 9: Creation of resource and source vulnerability maps by a combination of the factors O, C, P and K.

3.5 Derived Methods

As mentioned earlier, European Approach to intrinsic vulnerability mapping is a conceptual framework but does not prescribe detailed guidelines, tables and formula to quantify vulnerability. Based on this approach, different methods have thus been developed. A short overview of those derived methods, which conform most closely to the European Approach, is presented in this section. In addition the structure of a computer programme designed to model contaminant transport in karst and developed as a part of this Action, is outlined. A detailed description is given in **Part B** of this final report of these methods and their application together with a consideration of other methods less closely related to the European Approach.

The PI Method (Goldscheider 2002) is a GIS-based approach to mapping intrinsic groundwater vulnerability with particular reference to karst aquifers. It was developed largely before the European Approach, which it closely resembles and hence the terms used, protective cover (P) and infiltration conditions (I), differ from the O and C terms of the European Approach. However, they are essentially the same in conceptual terms. The P factor describes the protective function of the layers between the ground surface and the groundwater table – the soil, the subsoil, the **non karst rock** and the unsaturated zone of the karstic bedrock. The P factor is calculated according to a slightly modified version of the German (GLA) method (HÖLTING et al. 1995) The I factor describes the infiltration conditions, particularly the degree to which the protective cover is bypassed as a result of lateral surface and subsurface flow in the catchment of swallow holes and sinking streams. As it is a method rather than just a conceptual approach, the parameters are assigned values and are combined to yield degrees of vulnerability. The method has been tested in eight karst areas in five countries to date.

A "Localised European Approach" (LEA) to intrinsic resource vulnerability mapping in England and Wales was developed during 1998–2002 by Dunne. It takes into account the overlying layers (O) and the concentration of flow (C). The method largely follows the concept of the PI method but is simpler and thus appropriate for areas with a less extensive database. Thus far LEA has been applied in six test sites. It is a methodology for resource rather than source protection and hence the K and P factors are not taken into account. The method appears to work equally well on limestones that are only slightly karstified as well as limestones that are regarded as highly karstified.

The COP method of intrinsic resource vulnerability mapping was developed in 2001 and 2002 by the Hydrogeology Group of the University of Malaga (Vías et al. 2002). Vulnerability is assessed taking into account the protective function of the overlying layers (O), the concentration of flow (C) and the precipitation (P). The factors O and C are quantified in a similar but slightly simplified way as it is done in the PI method. The O factor takes into consideration the thickness of each layer, the soil texture, the lithology and fracturing, and confined conditions in the aquifer. The C factor takes into account the presence of swallow holes, the slope and vegetation. The P factor is assessed on the basis of annual precipitation depth and rainfall intensity. To date the COP method was applied in two test sites in Southern Spain.

The Time-Input Method (Kralik 2001) was specially developed for the application in mountainous areas. It assesses vulnerability on the basis of two factors: travel time and input, i.e. groundwater recharge. Vulnerability is expressed in real time, which allows for validation.

VULK is a simple analytical computer programme, which can be used to model the transport of contaminants through the different compartments (sub-systems) of a karst system. It was developed at the Hydrogeology Centre of Neuchâtel, Switzerland by JEANNIN et al. (2001). It is a theoretical tool, which may be used to help to validate the output from methods of intrinsic vulnerability assessment. The model assumes an instantaneous release of a conservative contaminant at a given point on the land surface and simulates the resulting breakthrough curves after each sub-system. For resource vulnerability assessment, only the vertical transport through the unsaturated zone (overlying layers) is considered. For source vulnerability assessment, the lateral transport in the saturated zone (karst aquifer) is additionally taken into account. VULK only takes into account advection and dispersion, while reactive processes like retardation and degradation are not considered. Required input data are the flow path length through each sub-system, flow velocity, dispersivity, and dilution. Output data are travel time, concentration and duration of contamination. VULK can also be used for vulnerability mapping, if it is coupled with a GIS. The main advantage of VULK is that it aims at a quantitative understanding of vulnerability. However, it requires data that are difficult to assess for large areas, and it makes strongly simplified assumptions concerning the mechanisms of contaminant transport. At present, VULK is being further developed, in order to implement the process of concentration of flow (C factor) into the model.

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4 Specific vulnerability

4.1 Principles of specific vulnerability assessment

4.1.1 Need of specific vulnerability mapping

Intrinsic vulnerability mapping provides a generalised view of vulnerability. However, due to the numerous kinds of contaminants, to which European carbonate environments are exposed, specific vulnerability mapping is also required. The impact of pesticides, fertilisers, hydrocarbons or heavy metals caused by agriculture (irrigation, tillage, stock rearing etc.), accidents and industrial pollution often leads to the contamination of groundwater. Whereas intrinsic groundwater vulnerability refers to hypothetical conservative and persistent substances, each particular contaminant is still characterised by its own nature and behaviour. Thus intrinsic vulnerability inadequately characterises contaminant fate and transport as it accounts only for the inherent geological and hydrogeological setting of an area but makes no allowance for the nature of the contaminants concerned. As defined by COST Action 620, specific vulnerability of groundwater to a particular contaminant or group of contaminants takes account of the properties of the contaminants in addition to the intrinsic vulnerability of an area.

Groundwater vulnerability mapping has proven to be a valuable practical tool for land-use planning and groundwater protection, in helping to avoid the contamination of water present beneath sensitive land. To achieve this objective, it is useful under certain conditions to compile specific resource or source vulnerability maps. These maps are more sophisticated and contain more information than intrinsic maps reflecting a more detailed potential migration picture of a particular contaminant. Moreover, intrinsic vulnerability maps generally display a worst case scenario and fail to take into account the positive effects deriving from specific contaminant properties such as retardation and degradation. This ignores the additional attenuation potential of contaminants under certain conditions and thus may overestimate the vulnerability of an area.

It is not necessary to carry out specific vulnerability assessment in every catchment where vulnerability mapping is requested. It should only be undertaken when a particular need can justify the additional costs of obtaining the extra data. This could be due to the presence of a dominant contaminant or group of contaminants with similar behaviour. Another use is the compilation of different specific vulnerability maps for a catchment, representing the various contaminants, the impact of which has to be considered. They should be linked to the different existing or potential hazards in a catchment and as a result improve risk assessment and land-use planning. With current GIS tools, several specific maps can be produced relatively easily, which may be used for a variety of purposes.

4.1.2 Main karst characteristics relevant to specific vulnerability

Hydraulic properties play a significant role in the migration of contaminants in a karst system (Fig. 10). The existence of a dual porosity differentiates residence times. Water flow is very rapid in karst conduits and slow in the fissured matrix of the carbonate rocks. Preferential flow is very pronounced in carbonate environments, and in non-karst layers that enable preferential flow to bypass the protective cover. Most specific processes can only take place where enough time and contact surface is available to enable reaction with the medium. This usually is the case for slow diffuse flow in the layer matrix. A layer may be suitable for process occurrence, but loses its effectiveness due to rapid preferential flow enabling the bypass by even non-conservative substances. Epikarst transfers water from diffuse to preferential flow by collecting infiltrated water and concentrating it in the vertical conduits of the unsaturated karst zone.

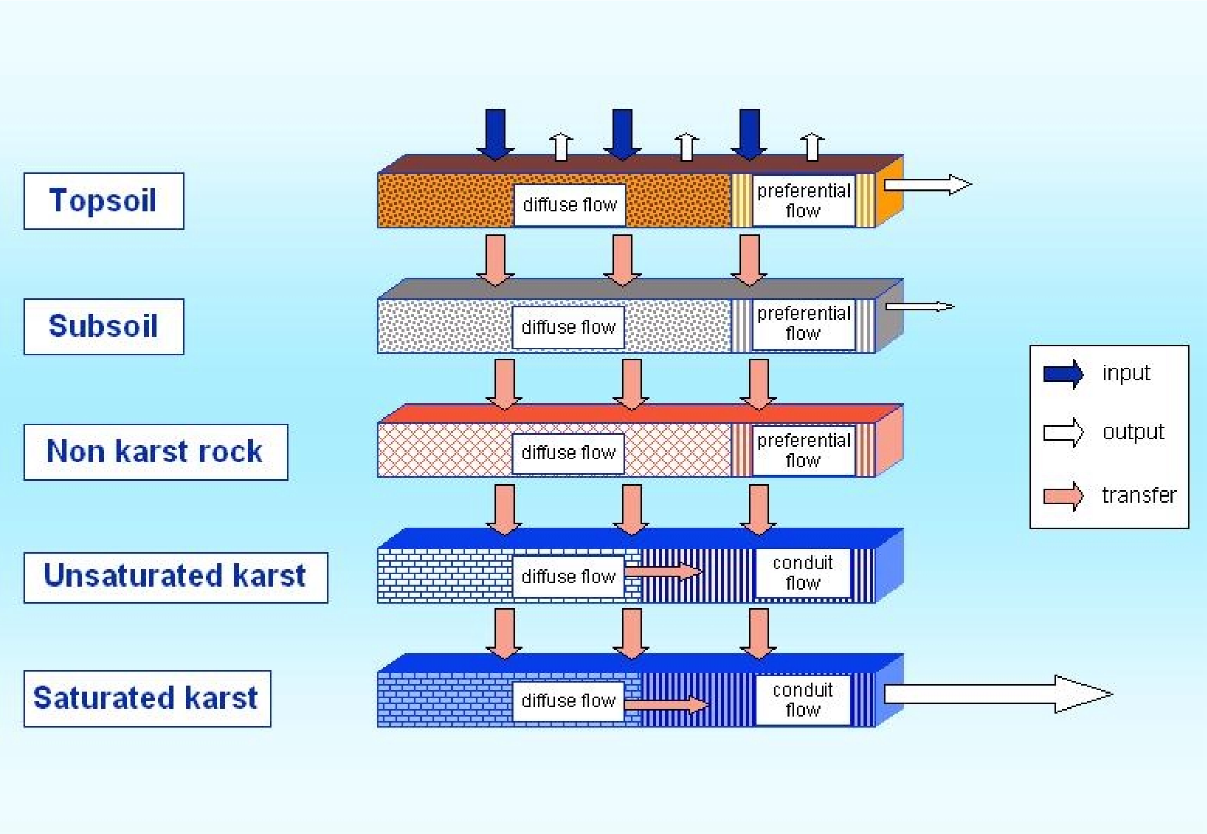

Fig. 10: Conceptual model of flow in a carbonate (karst) aquifer.

A second feature is the typical physical and chemical strata composition. In spite of its absence in residual topsoil, carbonate is abundant in many types of subsoil and in the karst layers. The carbonate medium restricts the mobility of several reactive contaminants (phosphates, heavy metals), which precipitate. Residual soils contain high clay content and oxides, which usually react with sorbable contaminants. Clay minerals exist also as detrital sediments in the karst network, both in the saturated and in unsaturated zones.

4.1.3 Effects of specific contaminant behaviour

Contaminants can be affected in transport and transformation by a number of different attenuation processes, which are not directly comparable one to another. However, in spite of the many effects on contaminant behaviour, they express themselves in a broader sense either in *retardation* or *degradation* of the contaminant load. Retardation results in decreased mobility due to lowered transport velocity related to water flow, which is taken into account for intrinsic vulnerability. Such a migration takes longer than conservative subsurface transport and the arrival of retarded contaminants at the target is delayed and more dispersed. Nevertheless, retardation cannot reduce contaminant load, but it provides additional reaction time for degradation processes. Degradation is the permanent loss of contaminant load from a water system. It shows itself in lowered concentration values and a reduced mass recovery at the target. This may happen by disintegration, but also by irreversible transformation or fixation.

Both retardation and degradation benefit aquifer protection and reduce vulnerability even if the contaminant does not fully disappear. Hence, specific vulnerability is nearly always lower than or at least equal to the intrinsic vulnerability of a particular area.

4.1.4 Processes relevant to specific contaminant fate and transport

Intrinsic vulnerability takes into account only advection, hydrodynamic dispersion and dilution. Whereas these processes are valid for each substance, the fate and transport of contaminants is a result of additional reactions occurring in the subsurface. The influence of the following physical, chemical and biological processes is regarded to be potentially significant for attenuation in the subsurface, depending on the contaminant nature: adsorption, cation exchange, filtration, sedimentation, biodegradation, oxidation and reduction, complexation, precipitation, volatilisation, as well as decay and die off.

Adsorption is the process whereby ionic solutes, undissolved organics or charged particles become attached to solid material. In many cases sorption activity is reversible after a certain time or after a change of environmental conditions and affects contaminant retardation. However, if fixation is permanent, adsorption leads to a loss of contaminants.

Cation exchange occurs when attracted cationic solutes displace other adsorbed ions at the surfaces exchange sites of substrate constituents. This kind of bond may be reversible or permanent.

Filtration is the physical retention of particles because of narrow voids smaller than the particle itself. Physical filtering may occur either in a fine- grained unconsolidated layer or in the matrix of hard rocks. If water flow is not rerouted, filtered particles are subject to permanent retention.

Sedimentation is the process of removing particles by gravitational settling in open voids at various stages of the lateral saturated water flow. If these particles are not fixed by compaction and diagenetic cementation they are frequently remobilised by changing hydrological conditions.

Biodegradation is a microbial-mediated transformation process and plays an important role in the attenuation of organic compounds. The biodegradation process eliminates organic molecules by creating metabolites. Depending on microbe oxygen requirement, biodegradation can occur under aerobic and anaerobic conditions.

Oxidation and **reduction** are reactions that chemically transform elements by changing their valence state, and equals, under stable conditions, the degradation of the contaminant in this form.

Several inorganic contaminants undergo **complexation** with ligands to form together a coordination compound, whose mobility can vary greatly.

Precipitation is another important abiotic transformation reaction. It occurs when a solid comes out of solution due to chemical change. Co-precipitation represents the incorporation of contaminants into mineral structures during the formation process, which leads to a concentration degradation in the liquid phase.

Molecules can be subjected to chemical or biological cleavage by the addition of water, a process termed **hydrolysis**.

Volatilisation is the loss by evaporation of chemicals into soil gas and the atmosphere within the unsaturated zone. Even if this process enhances mobility, it extracts and thus degrades volatile contaminants from the aqueous phase.

Decay is the spontaneous disintegration of radionuclides into daughter elements due to radiation emission. This process can produce other radioelements. Because reproduction is reduced in the subsurface, the amount of pathogens is also decreased in a natural way, called **die off**. Both processes lead to the elimination of the respective contaminant.

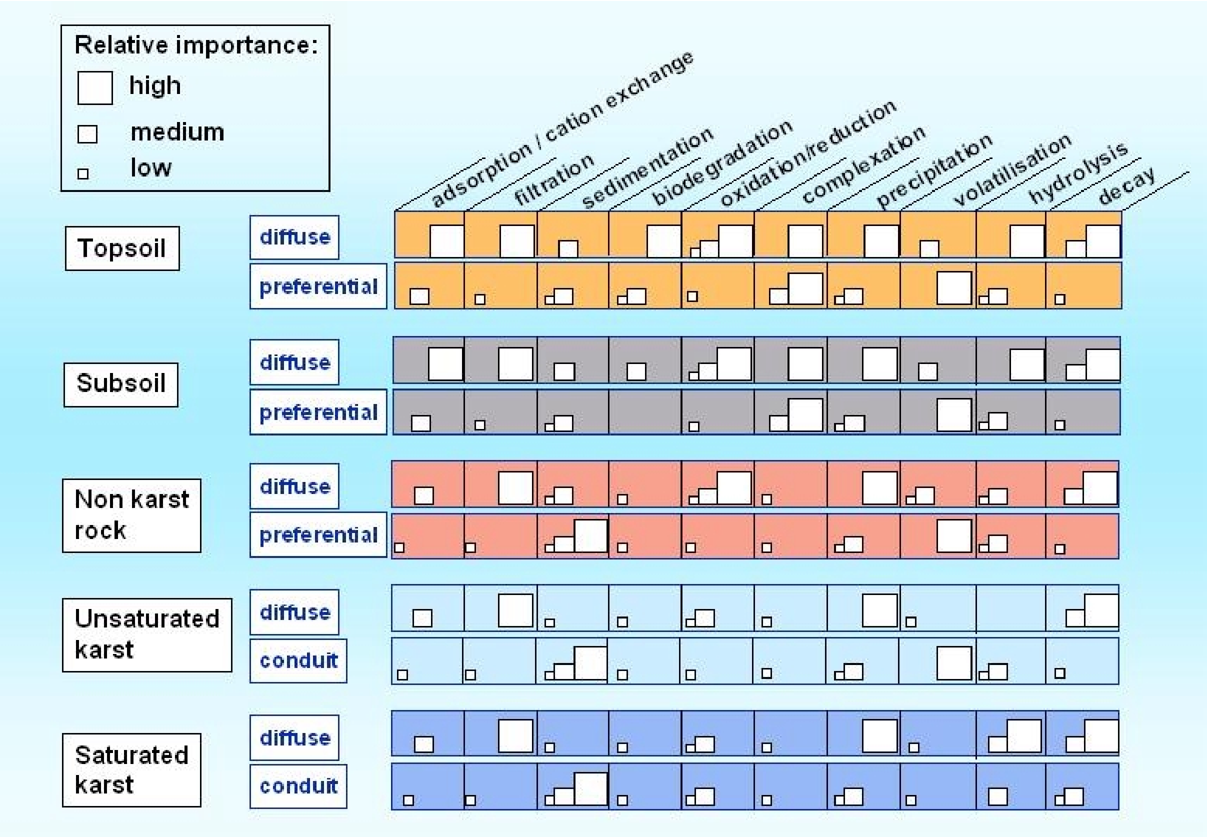

Fig. 11: Key processes affecting transport in karst.

Chemical and biological substances may be affected by one or several of the aforementioned key processes. Each single process can contribute to contaminant attenuation completely, partly or not at all. Thus, there is a wide range of possible migration behaviour from persistent to very degradable and from very mobile to fixed, e.g., some inorganics are sorbable, undergo precipitation or reduction and/or are degraded by decay. Organics are more or less degradable; they may be susceptible to biodegradation and sometimes to volatilisation, whereas filtration, sedimentation and die off normally affect biological particles. The sum of all the active processes, superposed for all the layers through which a contaminant passes between the hazard at the land surface and the studied target, describes the behaviour scheme of a specific contaminant.

Process activity in the subsurface equally requires favourable conditions within the stratum through which the contaminant passes. Most processes take place in unconsolidated deposits, particularly the topsoil and upper subsoil layers. On the other hand, karstified limestone formations generally provide minor specific attenuation capacity, whereas intermediate layers range from process-rich to less affected. Hence, a relatively thin topsoil overlying a limestone formation, that is often the case in carbonate areas, can be very important for karst groundwater protection due to specific processes. Nevertheless, some attenuation processes prefer preferential flow to occur, such as volatilisation and sedimentation. Fig. 11 shows the main processes in each layer with their probable importance based on the layer properties and the fraction of diffuse and preferential/conduit flow.

4.1.5 The working of specific vulnerability assessment

Specific vulnerability mapping deals with the assessment of contaminant and layer properties that determine and influence the previously described processes. It is important that all properties that are relevant for the processes are considered. Each layer offers different conditions for the transport and fate of contaminants due to its own characteristics. Specific attenuation in the subsurface is the superposition of all processes that a contaminant encounters in all layers.

Intrinsic vulnerability assessment cannot account for this diversity of processes and acts only as a general basis for each specific assessment. Specific vulnerability assessment aims at combining the effects of intrinsic and specific processes. Intrinsic vulnerability maps and databases provide intrinsic values. These have to be modified, using a specific weighting factor that adjusts the intrinsic vulnerability figures.

Specific vulnerability mapping sets out to identify and quantify potential contaminant attenuation in the subsurface. Specific vulnerability maps can represent only one particular contaminant or a group of contaminants of similar behaviour, due to layer and contaminant variability. They distinguish areas that are more or less susceptible to the specific contamination, modifying the distribution pattern of intrinsic maps. It is necessary to create an individual map or database for each particular contaminant, which is only valid for that contaminant or group of contaminants. Whereas, intrinsic vulnerability mapping embraces the whole spectrum of possible impact substances, specific vulnerability concentrates on the most important one(s) in the study area. It is therefore, a more appropriate but at the same time a more restricted tool than intrinsic vulnerability.

4.1.6 Contribution of COST Action 620 to specific vulnerability assessment

Some work has previously been done on specific vulnerability mapping (see Vrba and Zaporozec 1994, Magiera 2000). However, appropriate standard methods for the whole contaminant spectrum including the characteristic properties of carbonate environments and karst groundwater are still awaited.

Working Group 2 of COST Action 620 dealt with specific vulnerability as a part of integrated risk assessment. In the following sections, the subsurface behaviour of different kinds of contaminants is shown by defining the crucial layer and contaminant properties and by including information about contaminants in carbonate environments. This enabled the development of an assessment procedure for obtaining a specific weighting factor based on a conceptual model, the "European approach" for specific vulnerability assessment, with a particular emphasis on the special needs of karst environments.

4.2 Physical and chemical properties of layers and related processes

The behaviour (transport and transformation) of contaminants in karst areas, as well as in other hydrogeological environments, is largely dependent on the composition, the thicknesses and the sequence of the layers. The importance of geochemical and biogeochemical processes, responsible for retention, retardation, transformation and degradation of contaminants is therefore strongly linked with the concentration of a particular contaminant on the one hand, and the geochemical conditions of each individual layer on the other.

Of the different layers in karst environments, the topsoil layer has the most important attenuation function. If unconsolidated layers overlie karst, these subsoil sediments can also present retention properties for many contaminants. Where such layers are absent, the epikarst zone, as a particular kind of unsaturated karst, may have important retention properties (e.g. due to soil infillings). The primary karst aquifer flow-paths, with preferential conduit flows and high flow rates, do not enable the retention of contaminants, except by temporal sedimentation. In this part of karst systems, only the physical processes, dispersion and dilution, can reduce contaminant concentration. Depending on the hydrodynamic conditions in a karst aquifer, contaminants may be flushed from the aquifer or, over time, accumulate in the fractures, cavernous voids and the blocs of the karst aquifer.

The main constituents and properties of the topsoil and the unconsolidated subsoil deposits, that influence the specific vulnerability of the karst environment are: organic matter content, clay content and composition, cation exchange capacity, content of Fe, Mn and Al oxides and hydroxides, carbonate content, matrix aperture, as well as pH, redox conditions and temperature. The amount of these parameters in the different layers (Fig. 12) has a significant influence on the key processes. For each layer the key properties differ between the rapid flow component (preferential flow, conduit flow) and the diffuse flow component.

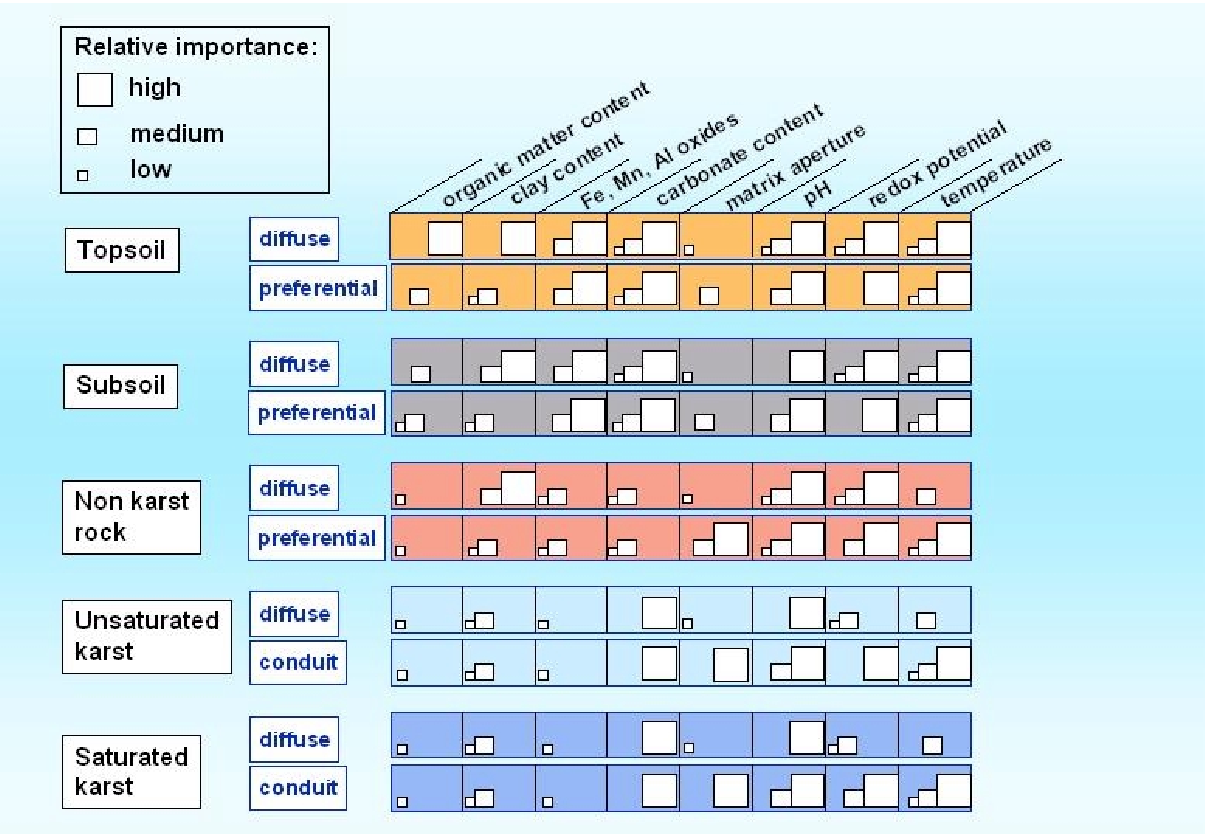

Fig. 12: Key properties affecting transport in karst.

Organic matter is the most reactive component of the soil layer and likewise very often is a major component of overlying fine-grained unconsolidated deposits. It is important because it increases the ability of the soil to retain nutrients (making them available for plant uptake), binds heavy metals (making them less soluble in water by readily forming complexes with heavy metals) and sorbing organic compounds. The large amount of organic matter in these layers provides a valuable medium for microorganisms, which plays an important role in the transformation processes of many organic contaminants such as volatile organic compounds (VOC) and pesticides.

Due to its high reactivity, the organic matter content is the most important constituent for the processes of adsorption, ionic exchange, biodegradation, oxidation-reduction and complexation.

Clay with high amounts in the soil and subsoil sectors is usually a mixture of different groups of minerals. The retention potential of many contaminants depends on the mineral composition present. The vermiculite-smectite group of clay minerals has the greatest ability to adsorb contaminants. Even a small quantity greatly influences the cation exchange capacity (CEC) in soils, otherwise dominated by low charge minerals such as kaolinite. Ion exchange reactions on permanent charge sites in silicate clays are also important in pH buffering.

The clay content is an important parameter for adsorption processes, and to a slightly lesser extent for cation exchange and biodegradation processes.

CEC (cation exchange capacity) is a quantitative measure of the ability of a mineral or other solid surface (e.g. organic matter) to bind ions. CEC is the sum of exchangeable cations that a material can bind at a specific pH. The mineralogical composition of the clay in soils strongly affects its cation exchange properties. The high charge characteristics of soil organic matter enhance retention of cations (Al^{3+}, Fe^{3+}, Ca^{2+}, Mg^{2+}, NH_4^+) and many heavy metals. Organic matter provides 25 to 90% to the cation exchange capacity of the surface layers of mineral soils. There are two main types of binding, by physical adsorption and by chemical adsorption. Physical adsorption occurs when the contaminants in the soil solution are attracted to the soil constituent's surface because of the unsatisfied charges of the soil particles. In this case, ions are held primarily by electrostatic force. Chemical adsorption always includes chemical bonding between atoms: ionic, covalent or coordinate-covalent.

For these reasons, the value of the CEC is an important parameter for cation exchange and adsorption processes.

Fe, Mn, Al oxides in all layers may be present in different mineral forms. Goethite and hematite are the most common Fe oxides in an aerobic environment, while goethite, lepidocrocite and ferrihydrite are common in a reduced soil environment. Gibbsite is the most common among six different Al oxides in soils, while boehmite is less common. Fe and Al oxides have large specific surface areas (FeOx 70-250 m^2g^{-1}, AlOx 100-220 m^2g^{-1}), high point of zero charge (p.z.c.) and a variable surface charge. The adsorption is related to reactive (singly coordinated OH- groups at crystallite edges) rather than the total surface area. Significant correlation is found between the sorption of some heavy metals (Cu, Pb, Zn, Cd, Co, Mn), inorganic anions (PO_4, SiO_3, MoO_4, AsO_4, SeO_4), some organic anions and the content and the mineralogy of Fe and Al oxides (Hingston et al, 1974; McBridge and Wesselink, 1988; McBridge, 1989). As in the case of adsorption, precipitation of Fe and Al hydroxides also removes some contaminants from the aqueous phase. Co-precipitation of heavy metals together with the oxides/hydroxides of Fe and Al is a common process in many soils. In karst regions, due to the aerobic environment, many heavy metals can be removed from the aqueous phase to the solid phase by such processes in each layer.

Manganese oxides are usually minor components in soils, but exert a significant influence on the soil chemical properties. They are present in many different mineral forms (amorphous to poorly crystalline forms). Many authors have found a high sorption capacity of Mn oxides for metal ions. Metal ions are adsorbed in the increasing order Mg < Ca < Sr < Ba < Ni < Zn < Co < Mn < Cu < Pb (Murray, 1975), leading to the accumulation of a relatively high concentration of heavy metals (Child, 1975; Sidhu et al., 1977) and actinides (Means et al., 1978;

Cerling and Turner, 1982). The Mn oxides are also strong inorganic oxidants, which affect the bioavailability and toxicity of particular metals in the aquatic environment (Co, As, Cr) as well as Fe oxides; Mn oxides act as final electron acceptors in oxidizing organic compounds and are consequently dissolved in the process.

Many geochemical studies and experiments have shown the important adsorption function of Al, Fe and Mn oxides, in comparison with other soils and sediment constituents, such as organic matter and clay contents. However, their importance is usually lower.

But, in some European karst regions, the typical soil is "terra rossa", which contains large amounts of Fe and Al hydroxides and oxides, with negligible amounts of organic matter and clay. In such cases, these oxides are essential for the retardation of heavy metals, and their influence is greater than in regions with different pedological characteristics (Durn et al, 1999; Miko, et al, 1999).

Carbonate content (amount of carbonate minerals) is a very effective buffering factor in the soil layer. Its buffering capacity is greater than organic matter or any other soil component. Also, calcareous soils are very highly buffered against acidification. The carbonate content is the factor responsible for precipitation processes, especially for co-precipitation of some heavy metals with carbonates as well as precipitation of phosphorous in calcium phosphate forms (hydroxyapatite), in the zone of groundwater level fluctuation. Also, the well-buffered environment of a carbonate aquifer prevents the long distance transport of dissolved forms of some heavy metals, due to their incorporation in precipitated carbonates on suspended material and fracture surfaces and/or in cavernous voids. Besides it has been established that cadmium has a great affinity to sorption at the calcite surface (Davis et al, 1987). But, such an environment is favourable for the transport of metals, (e.g. uranium), which form stable ionic complexes with carbonate ions.

For the reasons stated, the carbonate content is the most important parameter for precipitation processes.

Matrix aperture includes the most important physical properties of subsurface materials in relation to geochemical interactions and biotransformation processes: the texture and specific surface area of the solids in soils and in layers composed of unconsolidated sediments. Although the particle size of soils influences microbiological activity, the dimension of the matrix aperture (particle size) strongly influences the filtration processes. Small apertures of fractures in the carbonate rock can act as a filter for viscous non-aqueous phase liquids (NAPL) and particulate contaminants as mineral oils and microbes.

pH value is a negative logarithm of the hydronium ion H_3O^+ concentration, but in chemical reactions the term H^+ is often used instead of H_3O^+. Water solutions with pH< 7 are acidic, pH =7 are neutral and with pH>7 are basic (or alkaline). The pH of a system strongly influences what chemical processes will occur in the subsurface environment. The pH affects most other environmental processes in the subsurface: acid-base reactions, sorption reactions, precipitation and dissolution, complexation, hydrolysis and oxidation-reduction reactions. The pH value also induces the changes in salinity, in the cation exchange capacity of clays and in organic matter. In combination with other factors, for example the redox potential (Eh value) of the environment, the pH influences the type of bacteria that will be present and finally affects biotransformation processes. In natural environments, an increase of pH in most cases causes a decrease in Eh.

Thus, variations of pH in all layers mainly affect precipitation, hydrolysis and complexation processes. However, due to the influence on the surface charge of organic matter and clay, the pH of the subsurface environment is also important for processes of cation exchange and adsorption. Lastly, many microorganisms require a particular pH range for their growth and survival.

Eh means the **oxidation-reduction potential (redox potential)** and shows a tendency for the solution to be oxidised or reduced (reversible redox processes). It is the most important influence on biotransformation processes in the soil and the aquatic environment. Usually, this value is linked to the content of oxygen and other oxidising agents in the subsurface environment. Oxidation-reduction processes are governed by the value of the redox potential. Also, Eh values of the subsurface environment, of soil and overlying unconsolidated sediments affect other chemical processes such as hydrolisation, complexation and precipitation.

Biodegradation, by which many organic compounds undergo structural changes in response to biological activity, is very often affected by the redox conditions of a particular environment (layer).

Temperature primarily influences the rate of chemical reactions. For example, the rate of most acid-base and redox reactions and dissolution reactions increase with temperature. But the increase of temperature has the greatest influence on volatilisation processes. Volatilisation has been considered as a significant factor in the decrease of many organic compounds. The rate at which chemicals volatilise from the soil is affected by many factors, the properties of a chemical involved in volatilisation (its vapour pressure, solubility, structural type and number, the nature and position of its basic functional groups) as well as the soil properties (composition and geochemical conditions).

Tab. 2: Matrix for the qualitative relationship between the physical and chemical layer constitution and the process effectiveness (- indicates little or no correlation, + significant correlation, ++ strong correlation).

Key properties \ Key processes	Adsorption	Cation exchange	Filtration	Sedimentation	Biodegradation	Oxidation	Reduction	Complexation	Precipitation	Hydrolysis	Volatilisation	Decay	Die off
Organic matter content	++	+	-	-	++	++	++	++	-	-	-	-	-
Clay content	++	++	-	-	+	-	-	-	-	-	-	-	-
Cation exchange capacity	++	++	-	-	-	-	-	-	-	-	-	-	-
Fe, Mn, Al oxides content	++	-	-	-	-	-	-	-	-	-	-	-	-
Carbonate content	-	-	-	-	-	-	-	-	++	-	-	-	-
Matrix aperture	-	-	++	-	-	-	-	-	-	-	-	-	-
pH value	+	+	-	-	+	-	-	++	++	++	-	-	-
Redox potential	-	-	-	-	++	++	++	+	+	+	-	-	-
Temperature	-	-	-	-	+	-	-	-	+	+	++	-	-

Tab. 2 gives an overview about the influence of the physical and chemical layer properties on related processes with a qualitative judgement of the importance of this relationship.

The layer type plays a crucial role for specific reactions. Although the uppermost layers (soil and subsoil) are generally the most important, each one can show a high activity for one or several processes. Layer parameters that are recognized to be important for contaminant attenuation have to be determined or estimated on the basis of available data. To facilitate this procedure, standard properties may be defined for different layers, which will meet a high percentage of agreement with natural settings found in Europe with variable climatic and geological conditions.

4.3 Properties of contaminants and related processes

The general concept of specific vulnerability deals with the fact that different substances behave in a distinct way in the same geological and hydrogeological environment, due to their physical and chemical properties. These properties determine the affect of one or several attenuation processes. If they are present, then the contaminant concerned is generally affected by the respective process. The following parameters were judged to be key properties for specific process effectiveness, resulting in contaminant retention and transformation: Solubility, partitioning coefficients, viscosity, degradation half-life, radio-active and biological half-life, standard reduction potential, equilibrium constants, vapour pressure, density and particle size.

Solubility is an important contaminant parameter defining the ability of solid material to be dissolved in groundwater. It has an influence on many chemical processes for inorganic and organic contaminants. Solubility ranges from nearly insoluble to entirely miscible. The more soluble the substance, the higher is its mobility potential in the subsurface due to transport in the liquid phase. Low solubility inhibits contaminant mobilisation and favours retention due to sorption processes, but it is not a limiting factor for miscible substances. Environmental conditions such as pH, Eh and temperature have a strong influence on metallic oxides and hydroxide solubility. Under changing conditions, these solutes may re-precipitate.

Partitioning coefficients are strongly related to the solubility of a compound and indicate the affinity being affected by different sorption processes (mainly adsorption and cation exchange) with high values for reactive substances. Substances with a low partition coefficient have a lesser potential to move and to contaminate groundwater. Nevertheless, even reactive contaminants are transported very rapidly if they are adsorbed to a mobile colloid. For low sorbable compounds, the amount lost in the strata medium is small, whereas for high sorbable compounds the amount can be substantial. The distribution coefficient K_d can be deployed for both inorganic and organic contaminants and is the slope of a so-called isotherm illustrating the concentration ratio between the solution and the solid phase of a contaminant. The octanol-water partition coefficient K_{ow} is one measure of how non-polar and hydrophobic an organic compound is. A higher K_d or K_{ow} indicates that a greater amount of material is in the solid phase. Conversely, a lower value indicates that most of the material is in the liquid phase.

A **viscosity** higher than water decreases the advective flow velocity. Furthermore, fine-grained sediments or fine-fissured media may easily filter undissolved highly viscous organic compounds.

The biodegradation half-life of non-persistent organic compounds in the subsurface is commonly expressed in terms of **degradation half-life** (DT50) standing for the period of time required to degrade or transform the half of a chemical mass. Biodegradation in this sense is a cumulative parameter, which includes the biological and chemical decomposition by means of oxidation, reduction and hydolysis. DT50 values range from high for nearly persistent down to low for more degradable contaminants having a lower potential persistence in the environment. Half-lives are different for biotransformation under aerobic and anaerobic conditions.

Radioactive half-life measures the radiation activity of a radionuclide. Its value expresses the time needed for the decay of the half of the nuclides mass into daughter products occurring as an exponential function. The half-life is a physical constant for each nuclide and its related disintegration product. The decay rate is independent of the number of nuclides and thus of the contaminant mass and is not altered by physical environmental conditions.

Biological half-life is a parameter for determining the die-off of microbiology infiltrated in the subsurface. The higher the half-life rate, the better is the survival and the slower the attenuation. Biological half-live times range from days to years, depending on microbiological and environmental conditions.

Standard reduction potential is a dimension for the free energy of redox couples. It is important for several biotic and abiotic transformation processes, but crucial for oxidation and reduction reactions. High values indicate compounds, which tend to reduce; low values indicate compounds, which tend to oxidise.

Equilibrium constants indicate the potential of contaminants to undergo chemical reactions. Chemical equations may be useful to describe the capacity for chemical transformation processes of inorganics, and play the most important role in precipitation. Hence, the equilibrium constant seems to be the most helpful with carbonates.

Vapour pressure indicates the capacity of an organic contaminant to volatilise. Volatile liquids with a high vapour pressure tend to evaporate quickly. Non-volatiles are not scientifically degraded by volatilisation. Henry's law constant between liquid and gaseous phases is another indicator for volatilisation

Density is the crucial factor for contaminant sedimentation in the saturated zone. Particles denser than water may sink to the bottom of an aquifer and be stored there. Remobilisation under changing hydraulic conditions is possible but needs a much higher energy input. This property may also retard non-aqueous phase liquids (NAPL) with densities greater than water (DNAPL). Other substances may settle if they are bound to dense colloids.

Large particle **size** increases the likelihood of retention during flow in the matrix of the medium, due to physical filtration. Other contaminants, besides particles, can also benefit from filtration due to being adsorbed on large diameter colloidal minerals (e.g. clay, carbonates). Furthermore, a large-size particle is more affected by sedimentation than a small-sized one.

Tab. 3 shows the qualitative relationship between the contaminant properties and specific attenuation processes.

Tab. 3: Matrix for the qualitative relationship between physical and chemical contaminant properties and process effectiveness (- indicates little or no correlation, + significant correlation, ++ strong correlation).

Key properties \ Key processes	Adsorption	Cation exchange	Filtration	Sedimentation	Biodegradation	Oxidation	Reduction	Complexation	Precipitation	Hydrolysis	Volatilisation	Decay	Die off
Solubility	++	++	-	-	+	-	-	+	++	+	-	-	-
Partitioning coefficients	++	++	-	-	-	-	-	+	+	-	-	-	-
Viscosity	-	-	+	-	-	-	-	-	-	-	-	-	-
Degradation half-life	-	-	-	-	++	+	-	-	-	+	-	-	-
Radioactive half-life	-	-	-	-	-	-	-	-	-	-	-	++	-
Biological half-life	-	-	-	-	-	-	-	-	-	-	-	-	++
Stand. reduction potential	-	-	-	-	+	++	++	+	+	-	-	-	-
Equilibrium constants	+	-	-	-	-	-	-	+	++	+	-	-	-
Vapor pressure	-	-	-	-	-	-	-	-	-	-	++	-	-
Density	-	-	-	++	-	-	-	-	-	-	-	-	-
Particle size	-	-	++	+	-	-	-	-	-	-	-	-	-

Contaminant properties need to be evaluated only once and can then be used as input information for each specific vulnerability map. Examples of the selected substances, that threaten

European karst environments, are given in later sections where representative contaminant groups are described.

From the process point of view, each contaminant can be identified as sorbable, degradable, volatile, capable of decay, filtration and so on. Another more general classification groups contaminants into three categories, in each of which certain processes may occur:

Inorganics → adsorption, cation exchange, oxidation, reduction, complexation, precipitation, decay

Organics → adsorption, biodegradation (including oxidation, reduction and hydrolysis), volatilisation

Particles (almost entirely microbiological contaminants) → adsorption, filtration, sedimentation, die off

Organic compounds are often shown on a mobility-persistence diagram, plotting DT50 against K_{ow} values, assuming that biodegradation and sorption are the only processes to occur. Fig. 13 shows the wide range of degradation and retardation potential and the clustering of contaminants with a similar subsurface behaviour. Other contaminants may be grouped in a similar way, taking into account their retardation and degradation potential. The lower the mobility and persistence of a contaminant, the more a specific map may differ from an intrinsic map. For a relatively mobile and persistent contaminant, the specific map should be similar to the intrinsic map whereas they should differ significantly for reactive and/or degradable substances.

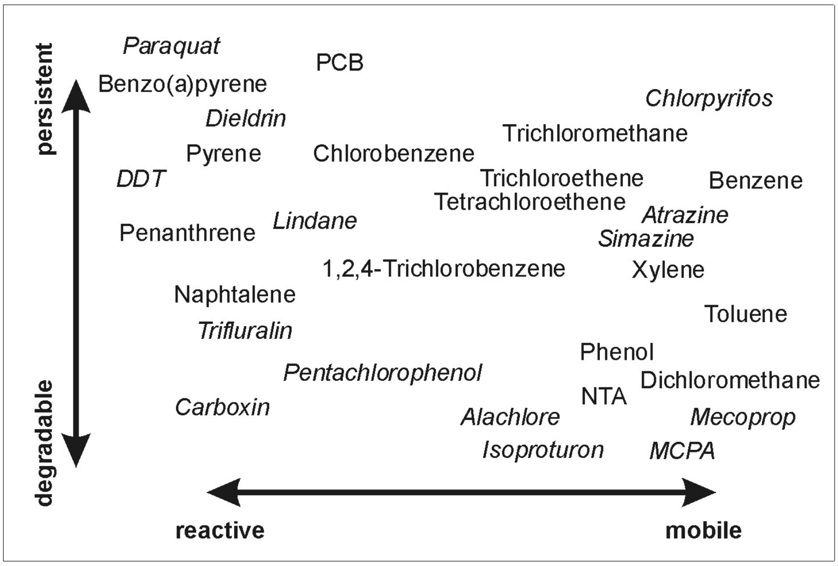

Fig. 13: Qualitative mobility-persistence diagram for selected organic compounds (pesticides in italics), compiled from DVWK (1989), Gustafson (1989), Schwarzenbach and others (1993), Suter and others (1997).

4.4 Contaminants in carbonate-karst groundwater

4.4.1 Inorganic contaminants

4.4.1.1 Background

Inorganic contaminants can have a natural or an anthropogenic origin. Their behaviour and toxicity are affected by several physical and chemical processes. These processes affect the concentration and transport of inorganic compounds in aquifers. The most important reactions are (1) speciation, (2) redox, (3) dissolution/precipitation and (4) sorption. Natural inorganic major ions and heavy metals can be released by rock dissolution and weathering, volcanic gases and changes of pH-Redox conditions within the aquifer. These natural origins rarely produce high contents (except high solubility salts, e.g. evaporites - Swenfurth 1994, and suspended solid materials - Prohic and Juracic 1989). Human activities produce very diverse inputs to karst hydro systems.

Nitrogen

Applications of fertilisers and disposal of wastewaters can result in high contents of nitrate, nitrite and ammonium.

Phosphorus

Used as a plant nutrient, its presence in groundwater results from the application/spillage of phosphated fertilisers, animal manure, silage effluents (Drew 1996), and also organic sewage sludge, solid wastes (Elhatip 1997) and detergents.

Heavy metals

The interaction of metals with inorganic ligands forms different chemical species. Speciation of transition metals is higher than other metals, but Ca^{2+} and Mg^{2+} have a lower tendency to form complexes. Concentration of complexes depends on (1) concentration of metal, (2) concentration of ligand and (3) concentration of other metals in solution.

Mining activities are important sources of heavy metals (oxidation in the mine - Stumm and Morgan 1996, leaching of dumps - Proctor and others 1977; Webb and Sasowsky 1994; Rude and others 1998). Landfill lixiviation, liquid urban wastes (Filipovic 1988), highways and parking lots (Monna and others 1995) produce heavy metals, but the highest contents are found in industrial wastes (Petrovic and Schleichert 1978). As examples, chemical industries produce Cd, Hg, Pb; electroplating: Cd, Pb, Cr, Cu, Zn; metallurgy: Cd, Cr, Cu, Zn; batteries: Cd, Pb; refineries and paint factories: Pb, Cr; fertilisers: Cd; pharmacy, paper, coal, fungicides and pesticides: Hg; gasoline: Pb; dyes, textiles, tanneries: Cr, soap and candles: Zn.

Radioelements

Radioelements now being reported in karst aquifers can have three main origins: fallout from the atmospheric nuclear tests of the sixties (tritium and other beta emitters, KUER 1969), Chernobyl fallout (mainly ^{137}Cs, BAG 1992), and the watch industry (tritium, KUER 1975, BAG 1992). In the latter tritium is used for luminous paints placed onto dials and handles. With this activity, considerable amounts are emitted in the atmosphere by incineration plants and in water by sewage treatment plants.

4.4.1.2 Transformation processes

Several inorganic ions (chloride, nitrate, sulphate, sodium, potassium) are conservative under special environmental conditions (oxidizing and quick flow medium), but significant parameters can control pollutant concentration, if flow conditions are slow enough to enable completion of the reactions (see section 4.4.1.3 transport).

Role of carbonate medium

The karst environment favours high bicarbonate content, which eases precipitation conditions for different solutes. Phosphate precipitates under the species hydroxyapatite (Fetter 1993). Trace metals can behave in two ways: several (e.g. Zn, Cu, Cd, Mn, Co, Fe) are strongly incorporated into calcite, while others are systematically excluded (Sr, Mg, Ba, U, Ra) (Rimstidt and others 1998). So supersaturation *vs.* calcite can remove several heavy metals in solution.

Role of pH and Redox

There is a positive and strong bounding between the amount of adsorption and the pH. The adsorbed mass for metals is maximum for a high pH and minimum for a low pH. As the pH increases, the adsorbed fraction also increases until theoretically, all metals are bounded to the sites of the material. For anions the pattern is a mirror image with maximum adsorption at low pH and minimum adsorption at high pH.

Redox reactions involve electron acceptors and electron donors, and therefore changes in the oxidation state and in the chemical properties of the elements. These changes can act in such a way that elements with a high mobility will remain immobile in the environment.

Thus, it is necessary to estimate the capacity of the system to be able to change its redox state; this is known as “redox buffering capacity” (Heron 1994). This capacity measures both the oxidation capacity (OXC) and the reduction capacity (RDC) of the system, the ability to change from an oxidation state to a more reduced one and vice versa. The OXC of a system depends on the concentration of dissolved and solid electron-acceptor compounds (O_2, NO_3^-, SO_4^{2-} and among the solid phases, Gœthite FeOOH and Pyrolusite MnO_2), and the RDC on the concentration of dissolved and solid electron-donor compounds (mainly dissolved organic matter or DOC and NH_4^+, and secondly Fe^{2+}, Mn^{2+} and S^{2-}). The reduced solid phases are built with metals in reduced forms, as Fe^{2+} carbonates (Siderite $FeCO_3$) or Mn^{2+} (Rodochrosite $MnCO_3$) and solid sulphur species (Pyrite FeS_2). Calcium and calcium-magnesium carbonates frequently predominate in carbonate (karst) aquifers and they will therefore be the key factors which drive the redox state of the system.

The redox state in carbonate aquifers varies from highly oxidized conditions (> 600 mV) to reduced conditions (< -200 mV). Oxidized conditions originate in conduits and fractures where (1) the water drainage is in a turbulent form, (2) the hydraulic connection between the recharge area and the aquifer is fast, and (3) the oxygen concentration is high or even almost saturated related to the atmospheric air (~ 8’5 mg/l at 25 ºC). A direct and fast connection between those recharge areas and the aquifer allows a higher input of rain water and also a higher concentration of dissolved electron-acceptor substances (mainly O_2) which results in a lower vulnerability to contamination. Reduced conditions would be produced (1) in deeper parts of the aquifer, (2) in areas hydraulically disconnected from fast fluxes (e.g. minor fissures or matrix), (3) in aquifers with overlaying clay beds that minimize infiltration or (4) during long periods of nearly-absent recharge. In each of these situations, the arrival of electron-acceptor species is difficult and the oxygen of the water will be depleted, resulting in less oxidizing or even reduced conditions. If the circulation is in areas of slow fluxes all the oxygen can be lost by diffusion, the reduced species would then only be subject to the hydrodispersive processes and thus their occurrence as reduced species will be extended. So well-oxidized carbonated aquifers are less vulnerable to contamination than reduced aquifers.

Together pH and redox have a significant role in that they determine the stability of the species. For example, a lot of metals exist under the solid form in alkaline and oxidizing conditions, and in the ionic form in acidic and reducing conditions. In alkaline conditions, phosphorus is removed from solution (Smith and Schrale 1982). Redox conditions, which are reduced mainly in confined aquifers, generally transform the solid metals into ions. For nitrogen (Reddy and Patrick 1981, Canter 1997), redox conditions alone can only produce ammonia volatilisation:

$$NH_3(aq) + H_2O \Leftrightarrow NH_4^+ + OH^-.$$

This reaction is pH dependant.

Most of the redox-dependant reactions are also controlled by biological conditions:

Ammonification is a biological conversion of organic nitrogen into ammonia. In anaerobic conditions, nitrogen remains in the form of ammonia, whilst in aerobic conditions, the process is followed by oxidation into nitrite and then nitrate.

Nitrification is a biological oxidation of ammonium into nitrate, in two steps, with the help of chemoautotrophic bacteria *Nitrosomonas* $NH_4^+ \Rightarrow NO_2^-$ and *Nitrobacteria* $NO_2^- \Rightarrow NO_3^-$. The general transformation is:

$$NH_4^+ + 2O_2 \Rightarrow NO_3^- + 2\,H^+ + H_2O.$$

The rate of this reaction depends on temperature, pH, alkalinity, inorganic carbon source, microbial population and ammonium concentration.

Denitrification involves more than 10 bacteria genera. This process transforms all species of nitrogen to a more reduced one:

$$NO_3^- \Rightarrow NO_2^- \Rightarrow NH_4^+ \Rightarrow N_2O \Rightarrow N_2$$

This process only takes place in anaerobic conditions in the presence of an available organic substrate:

$$5\,(CH_2O) + 4\,NO_3^- + 4\,H^+ \Rightarrow 5\,CO_2 + 2\,N_2 + 7\,H_2O$$

The denitrification rate is influenced by soil texture, oxygen content, readily available carbon, soil moisture, pH and the presence of denitrifies.

Role of radioactive decay

As its half-life is of the order of 12 years and the fallout from nuclear tests is now low, tritium is no longer a significant source. The Chernobyl accident released short half-life isotopes (^{131}I) which are now exhausted (Bundeskanzleramt 1988), but also ^{137}Cs, with a half-life of 30 years. Whilst some ^{137}Cs has been transported by runoff, some is still present in the soils and cave sediments.

Role of clay content and CEC

Clay, in a karstic environment, is present in overlying soils, in non karstic cover and in karst sediments. Clay plays a double role in the transformation of concentrations of inorganic contaminants: as a substrate for physical adsorption (metals), and as an ionic exchanger (interactions between alkali / alkali earth elements and clays / humic substances).

Nitrogen species can be adsorbed (NH_4^+, NO_3^-) on soil particles, and NH_4^+ exchanged, but both processes are reversible and pH dependant.

Phosphorus is probably submitted to sorption/desorption process mechanisms in the overlying soils (Hardwick 1995). As a heavy metal, ^{137}Cs is adsorbed on clay particles. As for heavy metals, cationic exchange is active and is proved by leaching which is observed during acid rain episodes (Kreutzer and others 1989).

Role of organic matter content

Organic matter, in a karstic environment, is present in both overlying soils and karst sediments. Its role is significant in nitrification and denitrification processes. It has not been reported as important for phosphorus, but plays a role in fixation of As and Se in an aquatic environment (Seyler and Martin 1991). Organic matter can form ligands with heavy metals, including metallic radioelements.

Characteristics of the karst environment

Except for "carbonate environment", which is present throughout the whole hydro system, most of the other conditions discussed in this section are controlled by external factors (confinement, non-karstic cover and soils). Furthermore, the reversibility of processes does not guarantee the permanent protection of water seeping into the limestone.

4.4.1.3 Transport processes

Several characteristics of the karst system play a significant role in controlling the specific vulnerability.

Infiltration

The duality of infiltration changes the input of inorganic solutes in the system. Diffuse infiltration enables a long residence of water in the soil / non karstic cover / epikarst layers. This means that the attenuation processes can have enough time to be effective, before water is gathered into the main drainage network. Conversely, point recharge directly concentrates inorganic content into the drainage network with no retardation occurring.

Hydraulics

The transport of both solutes and particles is significantly conditioned by the contrasting hydraulic ratings of the conductive drains and the low permeability blocks (i.e. resulting in conduit flow and diffuse flow respectively). Drains can occur equally in just the saturated zone or through the whole system from the soil zone to the outlet(s). As a result the drainage distance is not a consideration in the determination of the vulnerability of either the aquifer or its outlets. The transport of particles is important in inorganic movement because inorganic ions migrate as inorganic (inorganic particles, clays), organic (organic matter) or biological form (bacteria, viruses). But if physical and chemical conditions change, solid phases can dissolve again with an increase in the ionic concentrations.

The hydraulic relationship of drains with blocks results in a retardation process for conservative solutes which can be partly stored during flood episodes, and released during the recession. Any such increase in residence time of contaminants may be accompanied by attenuation processes (e.g. precipitation, adsorption).

4.4.1.4 Examples

Several different examples from the literature demonstrate the importance of the environmental conditions in karst groundwater vulnerability.

Phosphorus

The extent to which phosphate is retained within the aquifer is expected to vary, according to the relative importance of diffuse and conduit flow. Smith and Schrale (1982), in their study of Mount Gambier in S Australia, provide an example of phosphate retention. Whereas the annual nutrient load of a cheese factory effluent was 50 t of N, 35 t of K and 12 t of P, the nutrient load in the pollutant plume in the aquifer was estimated as 14 t of N, 14 t of K and only 0.1 t of P. This young karstic aquifer has a predominantly diffuse flow, where phosphorus precipitates.

Even in a conduit flow situation, phosphate can be retained within the aquifer: Wiersma and others (1986) note that in the Door peninsula shallow groundwater system in Wisconsin, with travel times from streams sinks of only a few hours, a seasonal pulse in P in the recharge waters did not produce a corresponding pulse in the spring discharge. Kastrinos and White (1986) carried out biweekly monitoring of both diffuse and conduit flow springs in the carbonate aquifer of central Pennsylvania over one year, but found phosphate to be below the limit of detection (0.02 mgl-1) in all but two samples.

In British caves, Hardwick (1995) found that phosphate concentrations were massively greater than natural backgrounds (rain) and increased following applications of sewage sludge to the overlying field. However, some sites discharged water containing levels of phosphate seemingly unrelated to agricultural applications. They are probably related to multiple adsorption/desorption mechanisms in the soils, to geochemical changes from insoluble to soluble P, and to soil percolation waters flushing soluble P into the cave.

In a general way, phosphate appears as a low mobility contaminant.

Heavy metals

In the case of acidic mine drainage, Fe precipitates along the pathway in limestone and the H+ released by the reaction is neutralised both by exsolution of CO_2 and dissolution of carbonate minerals which release Ca^{2+} and Mg^2+ (Webb and Sasowsky 1994). However, this process was found not to be completed within an 8 km transit: pH remained low and Fe and Al concentrations remained high (Sasowsky and others 1995).

In the case of a roadway runoff, discharged in a drainage well which was well connected to the drainage network, the drainage well contributed large percentages (22% to 75%) of the loads of As, Cu and Pb discharged at the spring (Hoos 1991).

In an example of uncontrolled discharge of untreated domestic and industrial wastewaters into sinkholes, a considerable increase of heavy metals concentrations in groundwater and a deterioration of water quality were observed at the springs (Filipovic 1988)

Heavy metals, despite their easy retention by clays or organic matter, appear to be very mobile in the karst drainage network.

Radioelements

About two weeks after the Chernobyl accident in 1986, analyses were made of drinking water in Vienna, Salzburg and Hallstadt (Bundeskanzleramt 1988). The total beta activity (mainly ^{131}I) increased up to 100 Bq/L.

In the Swiss watch industry area, (KUER 1975, BAG 1992), atmospheric tritium emissions from an incineration plant resulted in 100 Bq/L in precipitation, compared to about 0.8 Bq/L naturally present and about 1 Bq/L that remained from the 1960s nuclear tests. At the sewage treatment plant output, tritium values from about 100 Bq/L up to several kBq/L have been measured. Several karst springs situated some km downstream show tritium levels of about 100 Bq/L.

Nitrogen species

A hydrogeological study conducted in the clay-soil mantle of a limestone terrain in southern Indiana (Iqbal and Krothe, 1995), showed a consistent increase of nitrate at various depths in the unsaturated zone during a one year monitoring period. The increase of nitrate in soil water was attributed to the rapid flushing of inorganic fertilizers from the field after the area received sufficient rainfall in the late fall. The investigation also showed a major movement of nitrate in quick pulses through the unsaturated zone, rather than a slow uniform recharge. "Flushing" phenomena accredited to a karst system and temporal variations caused by hydrological conditions were recognised in many other studies (Felton 1996, Iqbal & Krothe 1996, etc.) As a conservative ion in oxidic karstic environments a decrease in nitrogen content of groundwater is usually caused by dilution.

4.4.1.5 Conclusion

This mobility of nitrogen and heavy metals, and even phosphorus, is worsened by turbidity, which is easily transferred to groundwater and springs, in highly conductive karst hydro systems

4.4.1.6 References

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4.4.2 Organic contaminants

4.4.2.1 Introduction

Background

Organic contaminants may be classified differently according to their analysis, their functionality, their use or their structure.

Tab. 4: List of organic contaminants

Contaminant (Synonyms)	CAS-No.	Formula	Water Solubility [mg/l 20° C]	Density [g/cm^3]	Dynamic Viscosity [mPa s]	Partition Coefficient K_d	Soil-Water Partitioning Coefficient K_{oc} 3)	Octanol-Water Partitioning Coefficient log K_{ow}	Aerobic Biodegradation Half-Life (soil) DT50 [d]	Anaerobic Biodegradation Half-Life (soil)	Vapor Pressure (20-25°) P^0 [kPa]	Boiling point (°C; 1 atm)
Mineral oils (total)	8012-95-1		5 - 250	0.88				1.1-3.3				180-405
Gasoline (benzine)	8006-61-9		120-250	0.72							0.34	80-130
Benzene (benzol)	71-43-2	C_6H_6	1800	0.88	0.65	50	97	2.13-2.17	good	bad	12.7	80.1
Toluene (methyl-benzene)	108-88-3	C_7H_8	526-535	0.87	0.59	130	242	2.65 - 2.69	good	bad	3.78	110.6
Xylenes (dimethylben-zenes; m-,p-,o-xylene)	1130-20-7	C_8H_{10}	175-198	0.86	0.7	170	363 - 588	2.77-3.20	good	bad	0.9-1.2	138-144
Chlorobenzene (phenyl chloride)	108-90-7	C_6H_5Cl	497	1.10		330	318	2.78	good	bad	11.73	130
Volatile Chlorinated Hydrocarbons (VCH)												
1,1,1-Trichloroethane (1,1,1-TCA)	71-55-6	$C_2H_3Cl_3$	1495	1.34	□1.20	0.0012	155	2.5-2.8	extrem. low		13.3	74.1
1,1,2,2-Tetrachloroethane	79-34-5	$CHCl_2CHCl_2$	2962	1.6	1.75		88	2.66			6.67	146.3
Dichloromethane (DCM)	75-09-2	CH_2Cl_2	18000	1.33	0.44		25	1.25			46.53	39.8
cis-1,2-dichloroethene (cis-DCE)	156-59-2	$C_2H_2Cl_2$	800	1.28	0.48			1.86			22.7	60.3
trans-1,2-dichloro ethene(trans-DCE)	156-60-5	$C_2H_2Cl_2$	600	1.27	0.92		39	2.09			35.3	47.5
1,1-dichloroethene (1,1-DCE)	75-35-4	$C_2H_2Cl_2$	2250	1.21	0.36		217	1.66-2.02			78.8	31.7
Tetrachloroethene (PCE)	127-18-4	C_2C_{14}	150	1.63	0.89		303	2.86			2.53	121
Tetrachloromethane (CTET; carbon tetrachloride)	56-23-5	CCl_4	1160	1.6	0.97		232	2.64			12	76.7
Trichloroethene (TCE)	79-01-6	C_2HCl_3	1100	1.46	0.57		152	2.3			7.8	86.7
Trichloromethane (chloroform)	67-66-3	$CHCl_3$	5600-7300	1.48-1.50			134	1.97			21	61-62
6 PAHs												
Fluoranthene	206-44-0	$C_{16}H_{10}$	0.265	1.18		19 000	40000	4.9	aer. slow		$0.7.10^{-6}$	375
Benzo(a)pyrene	50-32-8	$C_{20}H_{12}$	insoluble 0.0038	1.35		282 185	282185	4.33	bad		$0.7.10^{-7}$	495
Benzo(b)fluoranthene	205-99-2	$C_{20}H_{12}$	0.0012			$1.1\ 10^{+6}$	$1.1\ 10^{+6}$	6.57			$0.63.10^{-7}$	357
Benzo(g,h,i)perylene	191-24-2	$C_{22}H_{12}$	0.0003			$1.5\ 10^{+6}$		7.23				500
Benzo(k)fluoranthene	207-08-9	$C_{20}H_{12}$	0.0006			$2.0\ 10^{+6}$	$2.0\ 10^{+6}$	6.84			$0.67.10^{-7}$	480
Indeno(1,2,3-cd)py-rene	193-39-5	$C_{22}H_{12}$	0.062					4.19			$1.0.10^{-7}$	536
6 PCBs (PolyChlori-nated Biphenyls)	1336-36-3	$C_{12}H_{(10-n)}Cl_n$ (n=1,2,.. 10)	slightly soluble	1.18-1.5		300		5.22-10.44	med. to per-sist.			
PCB 28	7012-37-5	$C_{12}H_7Cl_3$	0.14				470					
PCB 52	35693-99-3	$C_{12}H_6Cl_4$	0.11									
PCB 101	37680-73-2	$C_{12}H_5Cl_5$	0.014				561					
PCB 118	31508-00-6	$C_{12}H_5Cl_5$										
PCB 138	35065-28-2	$C_{12}H_4Cl_6$	0.007									
PCB 153	35065-27-1	$C_{12}H_4Cl_6$	0.009									
PCB 180	35065-29-3	$C_{12}H_3Cl_7$	0.004									

Petroleum hydrocarbons (mainly aliphatic and aromatic (HAPs)) are probably the most frequent contaminants in karst environments. They are part of different distillation products such as crude oils, gasoline, kerosene, diesel, heavy oils, or asphalts, and are used in a wide range of applications such as fuel or lubrication.

The most common classes of organic contaminants are chlorinated hydrocarbons, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and pesticides (Tab. 4). Pesticides will be described in section 4.4.3. A more detailed review can be found in Kralik and others (2003).

Behaviour of Non-Aqueous Phase Liquids

One can distinguish between products lighter than water (LNAPL = Light Non-Aqueous Phase Liquid) and those heavier than water (DNAPL = Dense Non-Aqueous Phase Liquid).

One of the main problems is that organics as an immiscible phase can be trapped in conduits and serve as a constant source of contaminant by continuous dissolution in water. For this reason, the contamination by NAPLs of karst springs can last for much longer than that caused by conventional pollutants and can result in a water supply being permanently abandoned.

Moreover, a great deal of organics have a relatively high vapour pressure and can volatilise through the vadose network of the karst, resulting in attenuation in water but an increased danger in the air.

Several attempts to define these processes are reported in Field (1990) and Javandel (1998).

Due to the lack of observations, the problem of transport of NAPLs in fractured media is often approached by theoretical statements, simulations or experiments (Schwille, 1988, Gibert, 1990, Kueper and McWhorter 1991, Birkhoelzer and others).

4.4.2.2 Light Non-Aqueous Phase Liquids (LNAPL)

Background

Mineral oils and gasoline are commonly complex mixtures of more than 200 organic compounds. They are used in enormous amounts for heating, engine fuel and lubricants worldwide. In some industrial applications (e.g. wood preservatives) and at industrial waste sites, DNAPLs are mixed with mineral oils resulting in solutions that are still less dense than water.

In addition to the small fraction of hydrocarbons dissolved in groundwater, gasoline releases significant amounts of aromatic BTEX (benzene, toluene, ethylbenzene, xylenes) as soluble fractions due to their higher water-air partition coefficient. They can reach the water table via the infiltration of capillary water through a zone of residual gasoline, even if no gasoline itself reaches the water table. BTEX in groundwater can therefore be diagnostic of oil spills (Fetter 1993).

Occurrences of LNAPLs in karst

Very few case studies are reported because they are often confidential and not necessarily well documented. Moreover petroleum hydrocarbons are not commonly monitored in karst systems.

A few references can be found in Fels (1999), Rogers and Petrie (1999), Burman (1998) Crawford and Ulmer (1994) and Hötzl (1999). Kranjc (1999) reported investigations of four accidental oil spills in Slovenian carbonate-karst areas. He concluded that it may take a very long time, tens of years, before all the oil is washed out of the underground. In addition, investigations showed that the karst in some areas was more polluted by mineral oils not caused by the identified accidents.

Vulnerability and risk assessment

It is often difficult to predict what happens when NAPLs do penetrate a karst aquifer.

Modelling of the transport of dissolved components in water is common (Bernasconi and Tacher 1990) while multiphase transfer from the surface towards the vadose zone or the movement of LNAPLs or DNAPLs (at the top or at the bottom of water) is more difficult to predict (Palmer 1986).

4.4.2.3 Dense Non-Aqueous Phase Liquids (DNAPLs)

Background

A set of chlorinated organic compounds (Tab. 4) with relatively high vapour pressure, which have been used extensively in industry, fall in this category. Halogenated hydrocarbons which are mostly members of the aliphatic group have a wide range of uses in different industries. Many of these compounds are very effective degreasers and, as such, are being used extensively in plating shops, mechanical shops and a large group of industries dealing with metals. The relative low viscosity of the chlorinated solvents allows relatively rapid downward movement in the subsurface. Mobility in the subsurface increases with increasing density/viscosity ratio.

The low partitioning (Kd, Koc) to soil material exhibited by chlorinated solvents means that soil and rock materials will bind these compounds only weakly. This applies to both the unsaturated and saturated zones. Thus sorption to soil will not significantly retard the movement of a chlorinated solvent, and zones of contamination can grow essentially faster than groundwater can move. Chlorinated hydrocarbon pools have limited surface area in contact with moving groundwater and can persist for long periods of time (Javandel 1998).

Drinking water standards for chlorinated solvents are typically three to six orders of magnitude lower than their solubilities in water (Wolfe and others 1997).

Transformation processes

A sequential dechlorination from PCE to TCE to dichloroethylene (DCE) to vinyl chloride (VC) to ethane occurs. During reductive dechlorination, cis-1,2-dichloroethene (cDCE) is the most commonly formed isomer of DCE. Direct oxidation may serve a vital role in the sequential steps of chlorinated biodegradation by oxidation of lightly chlorinated solvents such as 1,2-dichloroethane (DCA), DCE, and VC to carbon dioxide, chloride and water (Wolfe and others 1997).

1,1,1-TCA can be anaerobically dechlorinated by methane-producing bacteria to form 1,1-dichloroethane, and decompose to give ethanoic acid and 1,1-TCE by abiotic reactions, with a half life of 200-300 days.

Transport processes

Chlorinated organic solvents (1,1,1-TCA, DCM, PCE and TCE) were detected in extremely low quantities (0,01-10 µg/l) in 80 % of 100 precipitation samples collected in 10 stations over the whole of Austria from 1993-96 (Sattelberger and others 2003). Samples exceeding detection limits (0.002-0.2 µg/l) occurred mainly in the cold season (Oct.-Apr.) with a seasonal low from July to September. Only DCM showed relatively higher values during the summer months as well.

The quarterly Austrian monitoring program (1998-2000) of 11 chlorinated hydrocarbons in 180 carbonate-karst springs showed that the detection limit of 0.1 µg/l was exceeded in just 3.6% of the samples for TCE, 2.7% for PCE, 1.6% for CTET and 1.1% for chloroform

(Kralik 2003). Due to the relatively low concentrations and the irregular distributions of the chlorinated hydrocarbons the main source seemed to be infiltration of already contaminated precipitation. Similarly, Fenelon and Moore (1996) found 12 of 58 VOC (volatile organic compounds) in a monitoring campaign of the shallow, partly carbonate-karst, aquifer of the White River Basin (Indiana). The most frequent ones, chloroform, 1,1,1-TCA and CTET, gave slightly higher concentrations in urban environments. In rural areas only 5% of the wells showed VOCs above the detection limits (chloroform, DCM) (Fenelon and Moore 1996).

However, a special study of 22 contaminated sites in the carbonate-karst of Tennessee revealed that maximum concentrations of PCE, TCE, 1,1,1-TCA and 1,2-DCE are in the mg/l range. Five models of DNAPL accumulation were developed: (1) trapping in regolith, (2) pooling at the top of bedrock, (3) pooling in bedrock diffuse-flow zones, (4) pooling in karst conduits, and (5) pooling in isolation from active ground-water flow (Wolfe and others 1997).

Further studies of DNAPLs in karst are given by Hötzl (1989, 1999) and Renner (2002). The latter describes the problem of vapour intrusion into houses above groundwater plumes contaminated with chlorinated solvents which are of particular concern in karst environments in Denver.

4.4.2.3 Polycyclic Aromatic Hydrocarbons (PAHs)

Background

PAHs have no industrial uses but are produced primarily as a result of the incomplete combustion of organic material. The principal natural sources are forest fires and volcanic eruptions, while anthropogenic sources include the incomplete combustion of fossil fuel in houses, power plants, and vehicle exhaust. Industrial plants having considerable PAH emission include coke ovens, gas production and refineries.

The primary concern relating to PAHs is that they are carcinogenic.

Transformation processes

In water, most PAHs are adsorbed onto sediments and suspended solids due to their extremely high Kd and logPow (see Tab. 4) values resulting from their strongly lipophilic character. Volatilisation may be important over periods exceeding 1 month. The lower molecular aromatic hydrocarbons (3-4 aromatic rings) are more water soluble, more easily biodegradable and more volatile. Therefore, the higher molecular aromatic hydrocarbons (5-6 aromatic rings) are more lipophile, less biodegradable and more strongly enriched on particulates in the environment. Most PAHs are susceptible to aqueous photolysis under optional conditions. They are slowly biodegraded in water and are taken up by aquatic organisms due to their lipophilic character (WHO 1996). Carcinogenic PAHs can be encymatically degraded to carcinogenic epoxyd metabolites.

Transport processes

PAHs were detected in only 10 % of 100 precipitation samples collected in Austria. Samples exceeding determination limits (0.003-0.02 µg/l) were taken only in winter and spring indicating that PAHs originate from heating during the cold period (Sattelberger and others 2003). The compounds of lower molecular weight aromatic hydrocarbons such as phenantrene, fluoranthene and pyrene (up to 0.053-0.116 µg/l) were relatively enriched compared to the higher molecular ones.

A similar study of background samples in the Austrian Alps showed a high proportion of less hydrophobic, more volatile PAHs, of lower molecular weight in fir needles compared to less volatile, better lipohilic PAHs with a higher adsorption capacity, higher molecular weight in the raw humus. All sites having above average results were concentrated north of the Central Alps. This area seems to have been subjected to higher input during the last few years and therefore also to long-range transport in the karst areas. Depth profiles at five sites showed that the PAHs are mainly adsorbed at the humus layer (Weiss 1998).

Due to the lack of literature relating to PAHs in carbonate-karst water no definite statements about the mobility of PAHs in such environments can be made. Due to the well known affinity of PAHs to suspended matter and humic substances, the filtrate should be analysed in areas of concern rather than the water itself.

4.4.2.4 Polychlorinated Biphenyls (PCBs)

Background

Because of their low cost, their insulating capacity and their non-inflammable nature polychlorinated biphenyls (PCBs; 209 possible isomers) have been widely used as coolants and lubricants, particularly in dielectric fluids in capacitors and transformers, and in heat transfer and hydraulic systems as well as fungicides in ship paints. Their distribution, persistence and accumulation in the environment and several incidents causing serious health problems among humans and animals have given rise to great concern.

In Europe 6 PCBs are normally analysed for and summed (PCB-Nr. 28, 52, 101, 138, 153, 180). In the US, the concentrations are related to specific Aroclors - Standards.

Due to sorption, generally low concentrations (<300 µg/l) are found. There is a relative enrichment of lower chlorine isomers due to preferential sorption (to solids; preferentially org. C) compared to higher ones.

Transformation processes

In a similar manner to PAHs, PCBs are adsorbed onto sediments and suspended solids due to their extremely high Kd and logPow (see Tab. 4). The log Pow range from 4.5 for Cl1 isomers to approximately 7.0 for Cl6, Cl7 and Cl8 isomers (Feenstra 1992). PCBs, however, seem in many cases to be less strongly adsorbed to soils and sediments than PAHs and therefore the biota–to-soil accumulations factor seems generally much higher for PCBs (Krauss and others 2000). Lower chlorine isomers are preferentially degraded under aerobic conditions. Anaerobic degradation will enrich lower chlorine isomers owing to the dechlorination of higher ones.

Transport processes

PCBs are normally below detection limits in precipitation as is the case for the Austrian Alps (<0.001-3 µg/l). Also in 214 spring samples in the Dachstein karst area (Eastern Northern Calcareous Alps) no samples were analysed as having PCBs above detection level (Herlicska and others 1994).

PCBs in carbonate-karst areas are reported from two sites only:

The Iskra condenser factory (Slovenia) consumed 3.7 million kg PCBs with a waste rate of 8 to 9 per cent in form of waste impregnates, condensers, etc between 1962 and 1985. This waste was dumped at various waste sites within 5 km of the factory. The PCBs pollution problem is related to sinking surficial streams that mix with the regional groundwater supply. PCBs emissions from underground occur mainly through the transport of polluted sediment and suspended particles during periods of heavy flow. Rapid desorption and resuspension of PCBs from the active sediment occurs as well as transport of the more soluble and volatile isomers to the water phase. Highest concentrations were 1 µg/l, which decreased in the mean from 0.38 µg/l in 1986/1988 to 0.1 µg/l in 1995/1997 in unfiltered water samples (Zupancic-Kralj and Jan1994; Polic and others 2000).

Uncontrolled disposal at the site at Lemon Lane in Indiana began at two sinkholes in the mid-1930s and continued until 1964. During the last 7 years of operation of the landfill, capacitors containing PCBs were disposed of. One spring, the Illinois Central Spring (ICS), about 650 m Southeast of the landfill was determined to be the main resurgence of PCBs. Under non-storm conditions, PCBs concentrations at the spring were found to be inversely related to flow (5 – 15 µg/l Aroclor PCBs and 15 – 5 l/s). The peak of PCBs concentrations measured at the spring during storms coincides with the arrival of storm water that had rapidly infiltrated to the subsurface conduit system (up to 300 µg/l). The total mass of PCBs discharged in the peak period during a storm was found to be a function of the storm's intensity. The similar shape of the PCBs concentration curve implies a slug like injection of PCBs to the aquifer from the vadose zone in form of rapid infiltration of uncapped soils of the landfill in the uppermost portion (first 3 to 5 meters) of the bedrock and the epikarst (Krothe and McCann 1996; Cepko and Cann 1999; Krothe and others 1999).

4.4.2.5 References

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4.4.3 Pesticides

4.4.3.1 Background

Because of their extreme heterogeneity, karstic aquifers are exceptionally vulnerable to contamination from surface-derived contaminants in general and from pesticides in particular, due to the very low levels fixed for these compounds in both groundwater and drinking waters. Sinkholes, fractures, swallow holes, and other open conduits enable poorly filtered concentrated recharge to take place. The degree of karstification seems to be decisive, as less karstified carbonate aquifers show behaviours similar to most nonkarst aquifers (HIPPE *et al.* 1994).

Pesticides include hundreds of organic and inorganic compounds used by farmers, institutions and the general public to control weeds, insects, and other pests. As the chemical compositions and therefore the environmental properties of organic pesticides vary greatly it is not possible to generalise about their transport behaviour and about their persistence. Simple assessment procedures are based on mobility (partition coefficient – K_{OC}) and degradation rates (dissipation half-life – DT_{50}). This allows a first ranking of the groundwater contamination risk of pesticides for homogenous flow in porous media (GUSTAFSON 1989) and has also been applied to karstic aquifers (PASQUARELL & BOYER, 1996).

Numerous recent books and papers synthesise the knowledge about pesticide contamination problems of groundwater in general (e.g. BARBASH & RESECK 1996; VIGHI & FUNARI 1994). A study of KOZEL & ANGEHRN (2002) reveals that about 160 pesticide compounds and degradation products have so far been detected in groundwater.

While many authors report pesticide-originated groundwater contamination, few of them specify the type of aquifer they studied. Karst groundwater is too often considered as a low-quality resource, and has been less studied than groundwater in porous media. Some studies mention that the occurrence of pesticides in groundwater can be of concern in karstic areas, but without any data or reference (e.g. SKARK, 1996).

Theoretically the leaching of pesticides in karst areas should depend highly on pesticide use and pesticide characteristics on the one hand, and on the degree of karstification and the importance and nature of the protective cover on the other. Vulnerable zones, where concentrated infiltration takes place, should present a generalised risk for all pesticide compounds to enter the aquifer, whereas migration through well-developed soil profiles without preferential flow and through limestone blocks should retain strongly sorbing and quickly degrading compounds. However, there is a lack of studies and data to verify these first assumptions and to weight the numerous factors involved in the leaching processes of organic compounds in the karstic environment.

4.4.3.2 Transformation

Many pesticides undergo chemical transformation in the subsurface, either with or without microbial involvement. The rates and pathways of these reactions are influenced by a variety of physical, chemical and biological factors. Specific karst conditions (high oxygen content, neutral pH, low organic matter content, strongly varying flow velocities) influence degradation processes (NOVAK, 1999). Recent studies reveal that the frequencies of detection in groundwater for a given compound increase multifold when its metabolites are considered (KOLPIN *et al.* 1998; BARBASH *et al.* 1999).

4.4.3.3 Transport processes

The subsurface migration of pesticide compounds is governed, as with all other solutes, by both advection and hydrodynamic dispersion. In addition, the widespread occurrence of conduit flow in karstic environments is analogous to the preferential flow of solutes through soil, and adds considerable complexity to the movement of pesticides. Migration rates can be much faster than those predicted, which can induce strong contamination peaks at the outlet (spring). On the other hand, the rates at which most pesticides move through the subsurface are slowed by sorption on organic matter and clay. But in karst environments reduced soil thickness and low organic matter and clay contents often limit the retention capacity of the system. The possibility of entry and transport of pesticides adsorbed to particles and colloids via open conduits is specific to karst areas (MAHLER *et al.* 1998). Particularly strongly sorbing hydrophobic compounds, such as paraquat or lindane, may be transported and resuspended in this way especially during storm events.

As karstic systems are very dynamic and monitoring points are scarce, the detection of organic trace contaminants as pesticides is difficult. Analyses performed at springs and wells of different depths in a karstic aquifer of northern Germany showed important seasonal variations of pesticide concentrations in the first 20 m of the saturated zone.

4.4.3.4 Examples

A literature review of pesticide originated contamination in karst groundwater shows the rarity of pertinent studies on the mechanisms and the dimension of this topic. As may have been expected, due to the general high vulnerability of this aquifer type, a large number of pesticide contamination events have occurred in karst waters. But, except for atrazine, only a few papers allow a consistent evaluation of the contamination status. A part of the pertinent information exists, but is not available, as monitoring results rarely refer to the observed aquifer type (KOZEL & ANGEHRN, 2000).

Occurrence of triazines in carbonate - karst groundwater

Widespread use and relatively high persistence and mobility are responsible for the widely reported occurrence of triazines, such as atrazine, simazine and terbuthylazine, in both karstic and non-karstic groundwater. Triazines are largely used as herbicides for agricultural and non-agricultural treatment (vegetation on railroads and roads). Atrazine is the most often detected and the best studied pesticide contaminant of groundwater. It is moderately mobile and it slowly degrades in a chain of metabolites such as desethylatrazine, deisopropylatrazine and hydroxyatrazine. Atrazine-originated groundwater contamination is therefore of real public concern (HEYDEL, 1998; DREW, 1996; JOHNSON *et al*. 2000; LEPILLER & ROUX, 2000; NELL, 1992; GOBBO-BUTTY, 2000; KOZEL, 1997; KRALIK, 1999; HAMIDI *et al*. 1996; LEGRAND *et al*. 1991; VILLINGER, 1987). For example, in Pennsylvania, 40 % of the domestic wells in a carbonate area revealed atrazine contamination, with a maximum of 3 $\mu g\ l^{-1}$ (HALL & LIETMAN, 1991). The metabolites, such as desethylatrazine, can be detected in groundwater, even several years after atrazine application, in the same or even higher concentrations than the parent compound. The ratio between the parent compound and the degradate allows an estimation of the age (PASQUARELL & BOYER, 1996) and/or the origin of the contamination (ADAMS & THURMAN 1991) in some cases.

The Austrian monitoring network based on 240 springs and almost 1300 analyses reported that maximum concentrations of desethylatrazine could rise to double those of atrazine in karst waters (KRALIK 1999).

Occurrence of phenylurea substitutes in carbonate-karst groundwater

Phenylurea herbicides constitute another group of frequently detected pesticides in groundwater. The presence of two main compounds, Isoproturon and Chlortoluron, is reported in several case studies, in both karstic and non-karstic aquifers. PERRIN-GARNIER et al. (1994) showed that Isoproturon concentration declines very quickly in the aqueous phase. After a few weeks, this compound cannot be mobilised further from the sorbing surface. Isoproturon is often detected in groundwater immediately after application, if rapid infiltration processes occur. In highly karstified aquifers, Isoproturon can thus generate short and more predictable contamination than atrazine (BARAN & LEPILLER, 1996). WELTÉ & SOFFIETTI (2000) showed this effect in the Dragon Basin aquifer, where Isoproturon was only detected when heavy rainfalls followed herbicide applications. In the German Münsterland, Chlortoluron had completely disappeared from karst groundwater 3 months after its application (BÖRGER & POLL, 1998). In Southern England, GOODY et al. (2001) studied a poorly karstified chalk aquifer and found out that, in such a system, the impact of phenylurea on groundwater quality was almost negligible, due to the dilution and degradation during the long transit through both unsaturated and saturated layers. However, it was shown that soil might play the role of a pesticide source for groundwater over years, even if it is no longer detectable in significant concentrations in the unsaturated zone.

Occurrence of other pesticides in carbonate-karst groundwater

A wide range of other pesticides is sporadically detected in karstic aquifers. Chlorinated compounds, such as lindane, endrine and dieldrine, occur in considerable concentrations in the Istrian karst (DIKOVIÆ, 1998).

CURRENS (1999) observed high concentrations of alachlor and metolachlor, compounds belonging to the amides and anilids group, with 9.6 and 6.1 $\mu g\ l^{-1}$, respectively, in a karstic spring of Kentucky. In the same state, KEAGY *et al.* (1995) detected metolachlor in karst waters. Alachlor was found in Northern France by DOERFLIGER & MOUVET (2000). In the karstic Dragon Basin aquifer near Paris, it was stated that alachlor behaved as isoproturon with detections only with heavy rainfalls following application (WELTÉ & SOFFIETTI, 2000). PASQUARELL & BOYER (1996) made the same observations in Western Virginia. Another herbicide, metamitron, was detected in France in the Dragon Basin near Paris, after heavy rainfalls (WELTÉ & SOFFIETTI , 2000).

A nematocide, carbofuran, was observed in the Pleasant Grove Spring Basin (Kentucky) at 7.4 $\mu g\ l^{-1}$ (CURRENS, 1999).

Other compounds such as mecocrop (JOHNSON *et al.* 2000) or pendimethaline (KEAGY *et al.* 1995) are also rarely observed.

Monitoring, regulation and attenuation

Monitoring and understanding of pesticide-originated contamination of karstic aquifers is complex. Various parameters specific to the pesticide must be linked to the karst heterogeneity and transport processes assessment, in order to evaluate pollutant behaviour. A realistic vulnerability assessment should take into account aquifer system characteristics, related to organic contaminant application data and specific compound behaviour.

Some countries have reacted to pesticide-originated groundwater contamination, and particularly to the presence of triazines in drinking water supply. Atrazine use is prohibited in Germany and strongly restricted in Switzerland since 1991. Triazine treatments have been prohibited in karst areas in Switzerland since 1998.

The natural attenuation effects as a consequence of such efforts begin to be noticeable. With respect to aquifer type and size, the time necessary to recover an acceptable water quality can vary dramatically (KOZEL, 1997). In the Swabian Alb Upper Jurassic karst aquifer (S-Germany), monitoring was performed after atrazine use was prohibited. It was shown that 10 to 30 years will be necessary for the values to fall below the legal 0.1 $\mu g\ l^{-1}$ limit depending on the aquifer zone (EICHINGER *et al*. 2000).

In NW-Switzerland, in the small karstic Gempen aquifer (watershed of 12 km^2), a similar study was carried out in 1994 (SCHUDEL, 1994). Two springs showed atrazine and desethylatrazine concentrations up to 1.3 $\mu g\ l^{-1}$. After the prohibition of the use of atrazine in the region in 1994, concentrations immediately began to decrease and the desethylatrazine/atrazine ratio rose (KOZEL, 1997).

In the 13 km^2 watershed of the karst aquifer Schöppinger Berg (N-Germany) evidence was shown that atrazine concentrations exponentially decreased in observed wells after atrazine was prohibited in 1991. In the monitored springs, it was not detectable after 3 years, while desethylatrazine was still present (up to 0.28 $\mu g\ l^{-1)}$ in all monitored springs during the winter months (BÖRGER & POLL, 1998).

The study of the Dragon Basin aquifer near Paris showed that both atrazine and desethylatrazine were detectable 5–7 years after the last use of atrazine. In this karstic system, atrazine seemed to be stocked in both soil and unsaturated karst layers (WELTÉ & SOFFIETTI, 2000).

4.4.3.5 References

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4.4.4 Microorganisms

4.4.4.1 Background

The term ‘microorganisms’ is not standardised and comprises a group of non-uniform organisms of considerable physiological variability. One feature they share, though, is a size of <150 µm (Karl 1982). The term ‘microorganisms’ includes parasitical viruses, which are non-independent living forms, bacteria, protozoa and small multi-cell organisms.

In karst water, for instance, which is one of the main sources of drinking water world-wide, investigations mainly concentrate on bacteria.

Bacteria are found in various habitats and are either of autochthonous or allochthonous origin. Autochthonous (or system-immanent) microorganisms, which dominate both the nutrient and energetic flows within their habitat, constitute the natural background of micro-flora, whereas allochthones come from other habitats, such as the soil or wastewater.

Generally speaking microorganisms can be detected by two different methods – either by culture-dependent or culture-independent analyses. Establishing a standardised investigation method for the evaluation of water grade (WHO 1996; Council Directive 98/83EC) guarantees both comparability and legitimacy (Tab. 5).

Tab. 5: Bacteria and protozoa and their significance in karst groundwater (‘higher’ is regarded as relative to the other groups listed here; [1] size of normally transported oocyst).

Bacteria and Protozoa	Synonym	Methods Council Directive 98/83/EC	Density $[g/cm^3]$	Partition Coefficient K_d	Aerobic Biodegradation Half-Life (soil) DT50 [d]	Anaerobic Biodegradation Half-Life (soil) DT50 [d]	Biological Half-life (die-off) $T_{1/2}$ [d]	Particle Size [µm]	Important animal and human reservoir	Council Directive 98/83/EC
Escherichia coli	*(E. coli)*	ISO9308-1	>1				higher	2 - 4	yes	0/100 ml
Coliform bacteria		ISO9308-1	>1				higher	2 - 4	yes	0/100 ml
Enterococci		ISO7899-2	>1				higher	1 - 2	yes	0/100 ml
Pseudomonas aeruginosa		ISO 12780	>1						?	0/100 ml
Clostridium perfringens (including spores)			>1		higher			2 - 4	?	0/100 ml
Colony count (22° C)	*(cfu/ml)*	ISO 6222					higher	1 - 7		100/1 ml
Colony count (37° C)	*(cfu/ml)*	ISO 6222					higher	1 - 7		20/1 ml
Cryptosporidium parvum			>1				higher	5[1]	yes	

Cryptosporidiosis is an important cause of diarrhoea. Cryptosporides are protozoons, their normally transported oocysts (permanent persistent part of the life cycle) have a diameter of about 5 µm. They normally occur in areas of dense cattle and sheep concentration.

In aquatic systems, thus in karst too, microorganisms living in biofilms play an important role. To identify them specific investigation techniques have to be applied. (Flemming and others 2000). Studies have indicated, that the majority of subsurface biomass is attached (Harvey and others 1984; Hazen and others 1991; Mills and Bouma 1997; Lehman and others 2001). Biofilms, which occur everywhere in karst systems, play an important part in this context. They can act as a sort of trap, which catches all sorts of bacteria and which again releases them under certain circumstances (e.g. increased discharge).

4.4.4.2 Transformation processes

Microorganisms intrude into the biofilms and in this way extend their chance of survival. Cysts of parasitic protozoons transform to parasitic cells after consumption of contaminated water by warm-blooded animals or humans. In addition, some bacteria transform to species resistant to chlorination.

4.4.4.3 Transport processes

The transport and fate of microbes within the subsurface is affected by a number of processes that can be divided into two types: those that are related to the characteristics of the organism and those that are related to the environment in which they are located (soil, sediment, unsaturated zone, groundwater (aquifer)). In many cases there is interaction between the two types and they cannot be treated in isolation.

Assessment in relation with its mobility

The most significant characteristics of microbes and the aquifer/soil environment which affect mobility and fate (identified by Robertson and Edberg, 1997, listed in West and others 1998) are discussed below.

Physical characteristics of microbes and aquifer materials

In the case of karst environments preferential flow paths are likely to exist. It is worth noting that colloids (defined as particles which remain in suspension by Brownian motion) have a linear dimension in the range of approximately 0.001–1.0 µm and so are physically similar to microbes. In the absence of research on microbes, colloid behaviour is a useful analogue of microbe movement and the processes which affect colloids can be directly extended to microbes (West and others 1998).

Colloid transport and filtering

In materials where intergranular flow dominates (soils and some aquifers) the peak concentration of colloids in a tracer test (as shown on the breakthrough curve) will be earlier than that of solutes due to preferential flow along pathways with larger aperture. In karstic environments such differences in breakthrough will be much less marked. Filtering effects are only significant when the average particle size exceeds approximately 5% of the average pore size (Harvey and Garabedian, 1991). Bacteria and protozoa are most likely to be affected by filtering (but generally only in the soil cover of true karst environments) whereas viruses, which are typically much smaller (<0.25 µm), are less likely to be so affected (Fig. 14).

Perhaps unsurprisingly, Pekdeger and others (1985) found that filtration constants measured in the field were some 10 times higher in the unsaturated than in the saturated zone. Matthess and others (1991) found that the filtration factor decreases by an order of magnitude if the flow velocity decreases in the same order.

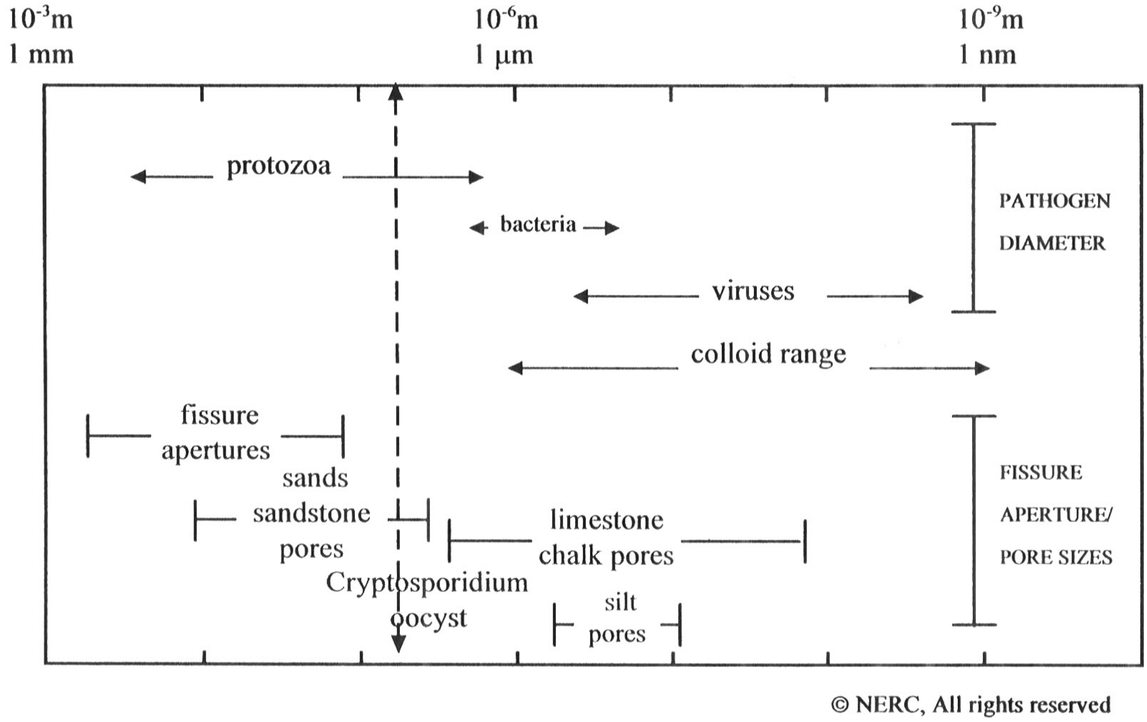

Fig. 14: Diameters of microorganisms and colloids compared to pore sizes and fissure apertures in aquifers (West and others 1998).

Density effects

The larger microbes (bacteria and protozoa) will tend to settle out if flow velocities are small because their densities are greater than water (see Tab. 5).

Inactivation of microbes and time of residence in the subsurface

Pathogens introduced into the subsurface will be transported in a viable form only as far as their life span will allow. The maximum distance that they will move (in a viable state) will be determined by the groundwater velocity and their survival time.

A large number of factors including, temperature, type of organism, water chemistry, organic content of soil and other predatory microbes can influence the half-life of microbes; the most critical factors usually being temperature and moisture content.

Temperature

The survival rate of microbes is generally inversely proportional to temperature with colder temperatures favoured for survival. Where the microbes have been adsorbed however, there is less temperature sensitivity (Liew and Gerba 1980). Otherwise as temperature increases, inactivation is rapid with half-lives halved for every 10 deg C rise in temperature between 5 and 30^0C (Reddy and others 1981).

Moisture content

This factor can play two roles:

1) A reduction in moisture content adversely affects the survival rate

2) More importantly the presence of moisture provides the fluid necessary for transporting the microbes such that at high moisture contents, and ultimately at saturated conditions, diffusion and advection will effect the microbes.

Adsorption

Three types of adsorption mechanism exist: physical, chemical and ion exchange. Viruses are most likely to be affected by adsorption because of their size. Soils/rocks which contain clays are more likely to have a higher sorption capacity due to the shape of the clay platelets and their large surface area (Savage and Fletcher, 1985).

Under most natural pH conditions, microbes suspended in water have a net negative charge, as do most mineral surfaces in the subsurface. Therefore there is a tendency for the microbes to be repelled by the rock matrix and remain mobile.

Hydrophobic materials (those with a low solubility in water relative to organic solvents) tend to associate with organic material rather than water. Many microbes, including some viruses, parasites and bacteria are hydrophobic to varying degrees. This causes them to dissociate within the groundwater and chemically sorb to organic material and coatings in the soil/rock matrix. Bales and others (1993) found that hydrophobic adsorption effects on viruses were greater than electrostatic adsorption. Total organic carbon (TOC) contents as low as 0.0005-0.001% were found to retard virus migration rates by factors of between 15 and 150.

However, changes in groundwater chemistry, organic degradation and other factors can act to reverse sorption and so release microbes. E.g. heavy rain leading to recharge water of different chemical composition can flush out microbes previously sorbed (Fourie and van Ryneveld, 1995 and McCaulou and others 1994).

Predation

It has been demonstrated that the presence of indigenous populations can lead to increasing rates of decline in non-indigenous microbes – e.g. actinomycets in the soil are able to suppress the growth of salmonella and dysentery bacilli (Bitton and Gerba, 1984). Many protozoa have been shown to feed actively upon bacterial populations (Chapelle, 1992). This predation by one population on another will influence survival rates and is a particularly important factor in the soil zone where biological activity is greatest.

Transport in unsaturated zones

Maximisation of residence time in the unsaturated zone has been proposed as the key for removal and elimination of bacteria and viruses (Lewis et al 1982). However, the downward migration of contaminants in the unsaturated zone of karst rocks is generally rapid and complex. Where carbonate rocks exhibit both fractures and a porous matrix, they are called dual porosity systems – e.g. the Chalk. Interchange between fissure water and matrix water is possible and can be a very important mechanism in contaminant transport in such systems. Where downward movement of water and contaminants is slow (0.5 – 1.0 m/year) diffusion into the matrix will result in equilibrium of concentrations in the fractures and the matrix. At higher rates of recharge and where the contaminant cannot physically diffuse into the matrix, movement will be restricted to the fractures and so migration may be more rapid and concentrations of contaminants reaching the water table much higher. For microbiological contaminants, both fracture flow and matrix flow are important. In the Magnesian and Carboniferous limestone aquifers of the U.K. the matrix pores are typically less than 0.5 μm and for the Chalk less than 1.0 μm. Microbes, especially bacteria and protozoa, are therefore effectively precluded from diffusing into the matrix and so will be confined to the fractures.

Lee (1993) investigated the contamination of a water supply well in a karstic environment by *Giardia* spp. and *Cryptosporidium* spp. Rapid infiltration of surface waters to the saturated zone through fractures and connection between the surface and the well was proven. Particle size analysis revealed that the full range of particle sizes found in the surface waters was not however present in the well. There were cut offs at both low and high ranges and it was concluded that there had been adsorption of the smaller particles and straining of the larger ones. The size range of particles that were transported through the system included *Giardia* and *Cryptosporidium*.

Transport in saturated zone

On reaching the saturated zone, the microbial contaminants will be subjected to the same processes described above. Flow will transport the microbes by advection and they will also be susceptible to diffusion and dispersion as a result of aquifer heterogeneity and variation in groundwater velocity. The dispersivity is scale-dependent and increases with increasing flow distance due to the inhomogeneity of aquifer systems.

There is evidently much variation in saturated flow regimes of karst (carbonate) aquifers but, in general, rapid flow regimes predominate. As a conservative approach to a consideration of specific vulnerability to pathogenic contamination, it is recommended that the saturated zone is not considered as providing a significant contribution to the degradation of microbial contaminants.

4.4.4.4 Examples

In this context a study of Thorn and Coxon (1992) should be mentioned. According to them, bacterial contamination, resulting from an insufficient natural protection, has to be considered at any time, even if the pollution is rather low. Although many studies describe a correlation between precipitation events and microbial strains, the results obtained so far are not satisfactory enough and cannot serve as indicators for vulnerability. Thus, the approach has basically been to define karst waters as raw water and not as drinking water.

Panno and others (1997) investigated the sinkhole plain of southwestern Illinois to study the microbial pollution of ground water, focussing on the bacterial pollution these karst regions are exposed to from private septic systems. As early as in 1988 Aldwell and others found that the main sources of pollution (faecal bacteria and/or ammonia) in vulnerable karst areas are septic tank effluent, farm yard wastes and sinking streams. Felton (1996), who performed studies at the outlet of a shallow carbonate aquifer (Inner Bluegrass, Kentucky), the water quality of which showed high faecal pollution, found that “temporal variation in bacterial contamination was not linked to any other variable. The highest faecal contamination corresponds to low flow rates”.

A survey of all studies carried out so far indicates that studies were either aimed at investigating water quality or at obtaining comprehensive information on karst ecosystems.

According to the final research report of the Austrian Calcareous National Park (Haseke 1999) microbiological and ecological investigations of spring water showed, that karst waters are inhabited both “inside and outside” by a far larger number of various microbial life communities than was expected so far. This study particularly emphasises the importance of microbial biofilms and the large diversity of small animal organisms in springs. Studies of geomicrobial processes in waters illustrate that biofilms actively participate in the formation of their environment (Reinheimer 1991). For an investigation of the influence of microbiological processes (formation of CO_2 following the decomposition of organic matter) on limestone dissolution see Mylroye and Balcerzak (1992). In another study, Bottrell and others

(1993) observed indications of processes caused by bacteria in connection with the formation of cave systems.

Since karst regions are important sources of drinking water in Europe, the hygienic quality of karst water is extremely important. Investigations carried out in the Austrian national park "Calcareous Alps" (Haseke 1999) show that those sources, which stem from deeper layers of karst systems or from the areas of the rock karst, which are much more exposed, show a far lower colony count than those stemming from properly karstified areas. In these springs large variations with extreme values of bacterial contamination were observed, depending on how intensely the catchment area was used.

In Alpine karst springs measurements were taken daily over a period of a few years. A considerable growth of *positive E. coli* becomes evident at the start of the warm season (June). Only towards the end of the year (Nov.) does the faecal strain in spring water decrease again.

Seasonal sampling of more than 100 springs in several parts of the Austrian Calcareous Alps (500 – 2000 m altitude) showed that in the winter (Nov. and Feb.) 80% of the spring water are in accordance with the strict limit values of the European Directive 98/83/EC. In spring and summer samples, however, only 12-44% are below these strict limit values. These data indicate that strong changes in temperature and hydrodynamic flow regime (snow melt and strong summer rains) cause significant seasonal changes in pathogen content. In addition, in areas where cattle grazing and touristic activities are common, they are in most cases limited to the summer period (May-September) (Kralik 2001).

Colony counts from many Alpine springs show that due to the stronger karstification the counts in limestone areas are generally higher than in dolomite karst. In Alpine karst covered completely with forest ("green karst") the colony counts are generally higher than in springs with no forest (scrub and grassland), thin soils or even bare rock. This indicates that most of these bacteria from colony counts are coming from soil. These bacteria can survive also in sediments and biofilms within the karst system. However, they are preferentially washed out during or after storms (Kralik, 2001). Traindl and Pavuza (1990) studied the colony counts from various carbonate soils and conclude that rendzinic soils with their high organic carbon content show the highest counts.

Several investigations in Alpine karst showed that many bacteria are attached to organic and inorganic particles. The organic particles seem to be dominant and have a tendency to colloidal composition. This is indicated by high turbidity values without any rise of bacteria numbers but a strong correlation between dissolved organic carbon (DOC) and bacterial counts. This is also supported by some springs where colony counts and UV-spectra (254 nm) show a linear and exponential correlation. UV-spectra (254 nm) measure the content of organic compounds in the water; for some springs, on-line measurements could prove to be an interesting tool for the improvement of microbiological sampling (Kralik 2001).

Perrin et al. (2001) reached a similar conclusion from study of a storm pulse in the limestone karst of the tabular Jura; soil bacteria were already present in the unsaturated zone at the beginning of the storm and faecal bacteria arrived later from the surface. Bacteria act as natural tracers and have a higher sensitivity than classic natural tracers.

Recommendations for vulnerability assessments

Despite the fact that karst regions differ strongly with regard to their hydrological / geological structure and thus react to microbial pollution in various ways, for a comprehensive evaluation of micro biological water quality the following method is recommended:

- Development of a site-specific network of collection sites (springs, boreholes).

- Regular samplings with additional samplings during times of higher precipitation, snow melt and during very low discharges with micro-biological standard programmes (total coliforms, faecal coliforms, faecal streptococci and heterotrophic plate counts (HPC)).

Standard and advanced microbiological techniques are useful in groundwater vulnerability assessments. Bacteria act as sensitive natural tracers. Soil bacteria are present everywhere and will be washed into and through karstified parts of the carbonate system very quickly. In non- or less karstified sequences, however, fewer bacteria will pass or they will die-off during longer passages. Faecal bacteria are an additional indicator for cattle grazing, tourism and a high density of wild life. Advanced microbiological techniques may offer the means to differentiate between these different sources.

As pointed out by Mahler and others (2000) the extreme temporal variability of bacterial contamination in karst water requires an event-based monitoring (storms and dry seasons) of the bacterial quality of groundwater in karst regions.

In this context it is important to stress again, that hygienic-microbiological investigations, which doubtlessly are essential, cannot give any information on the microbial density and activity of biofilms. As mentioned above, for their detection specific investigation techniques have to be applied.

A more detailed review can be found in Zibuschka and others (2003).

4.4.4.5 References

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4.5 Conceptual model for specific vulnerability assessment

4.5.1 Overview

Specific vulnerability is a particular case of vulnerability, which is only valid for an individual contaminant or a group of contaminants. Intrinsic vulnerability treats all substances as having similar transport behaviour as that of water, while specific assessment additionally deals with the differences between particular contaminant behaviour and their specific interaction with the host rock. Specific contaminant aspects are considered in addition to the more general hydrogeological framework of intrinsic vulnerability. The objective is to achieve an integrated intrinsic-specific assessment closely related and compatible to both. This chapter deals with the specific module exclusively as an adaptation to the intrinsic output.

In keeping with the European approach for intrinsic vulnerability, Working Group 2 of COST 620 suggests a “European approach for specific vulnerability”. This approach is based on the principles outlined in section 4.1. It takes into account the physical and chemical properties of layers and the related processes (section 4.2) on the one hand and the properties of contaminants and the related processes (section 4.3) on the other. An important reason for developing a conceptual model for specific vulnerability assessment is that karst aquifers are affected by the various contaminants summarised in section 4.4.

The aim is to create a standardised approach that takes into account the various specific geological and hydrogeological conditions within a catchment and that is applicable to each contaminant. This requires more than intrinsic data. Special features of the karst environment have to be added. The concept is kept general and sufficiently flexible for use in the weighting and rating systems of individual methods. These may reflect regional features or special user interests. The method presented in section 4.6 is one possible implementation of the approach.

4.5.2 European approach for specific vulnerability

4.5.2.1 Specific weighting factor (S factor)

Specific vulnerability assessment requires the creation of an additional factor for correcting the intrinsic vulnerability values (Fig. 15). For the process-based European approach, a specific weighting factor (S factor) represents both layer and contaminant attenuation capacities. The calculation of such a factor has to take into account specific processes. It includes a layer factor and a contaminant factor. The first sets out the potential process in respect to subsurface conditions; the second indicates the potential process of the particular contaminant. If both potentials fit, a process can become effective and can specifically attenuate vulnerability. This means that a contaminant, which due to its nature is subject to a particular process, requires a suitable environment to be affected by that process. Both factors together allow the evaluation of process effectiveness that cannot directly be assessed. While the layer factor treats the question (1) *if* and (2) *to what degree* processes may occur in the subsurface, the contaminant factor aims to confirm (3) if these processes are *relevant* for the contaminant concerned. From specific assessment are derived databases or maps with a specific weighting factor distribution.

In contrast to intrinsic vulnerability, layer properties, which enable specific process activity, do not per se enhance groundwater protection. Only those processes can be considered that are in keeping with the contaminant type concerned. Intrinsic vulnerability assessment omits this second step by assuming from the outset a conservatively behaving contaminant. It is thus a special case of specific vulnerability accounting only for advection, hydrodynamic dispersion and dilution.

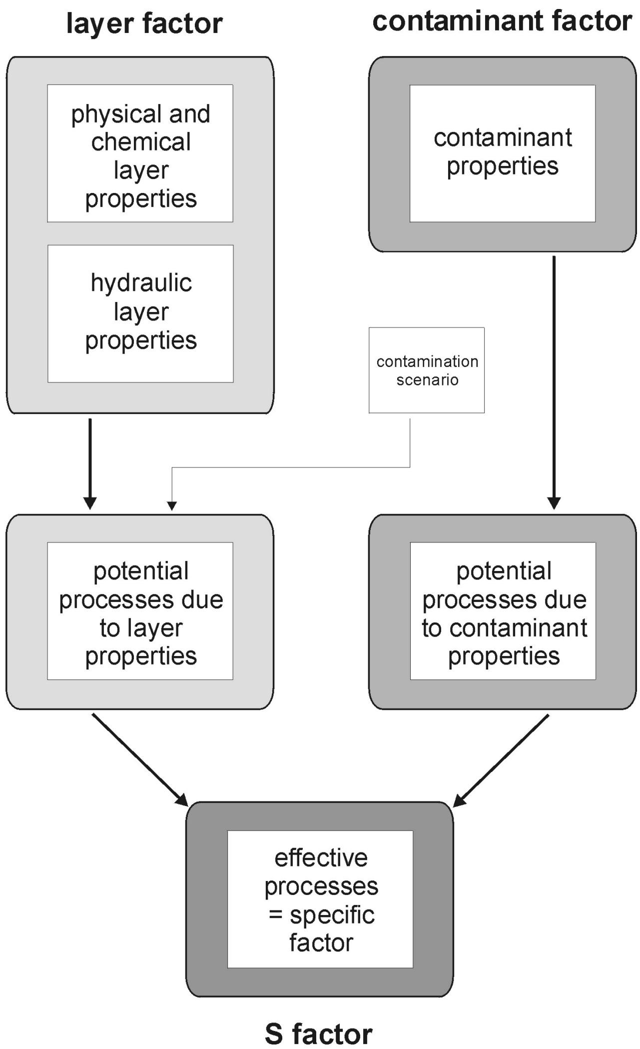


Fig. 15: Schematic procedure for specific weighting factor assessment.

4.5.2.2 Layer factor

The layer factor involves two kinds of properties as internal characteristics of the karst system, subdivided into characteristic layers:

The physical and chemical layer properties describe the composition and thickness of the topsoil, subsoil, non-karst rock, unsaturated and saturated karst layers. Suitable stratum composition is an essential requirement for the occurrence of potential processes. A detailed listing of layer properties is given in section 4.2.

Hydraulic properties are a second important parameter for process potential evaluation. The hydraulic behaviour of a stratum determines the effectiveness of its composition. It determines if a contaminant is really in contact with the medium, permitting attenuation processes. This is only ensured if both the residence time and the contact surfaces are sufficient to enable equilibrium conditions and interaction with the medium.

The effectiveness of layer properties can be limited and largely disregarded, due to preferential flow bypassing the strata. Rapid preferential flow bypasses the layers and thus eliminates their protective function. Only the portion of water and therefore of contaminants that slowly migrates through the layer matrix undergoes significant retardation or degradation. Therefore, the rate of rapid preferential conduit flow is a major limiting factor for layer effectiveness. The fact that, in many carbonate formations, a high percentage of water flows through enlarged fissures and conduits highlights the importance of the amount of such flow. Other important hydraulic properties are flow velocity and permeability.

4.5.2.3 Contaminant factor

The contaminant factor concerns the physical and chemical contaminant properties. It determines if the substance is liable to be affected by a specific process. Each contaminant has its own contaminant factor, split into one or several potential processes. The contaminant factor is not an internal characteristic of a hydrogeological system, but is an external specification of a vulnerability assessment. It determines if suitable layer properties can be transformed into effective attenuation processes. The key contaminant properties are shown in section 4.3.

In contrast to intrinsic vulnerability, specific vulnerability is not independent from the contamination scenario and history. Thus, an active process doesn't automatically mean the involvement of the whole contaminant mass. Some processes only permit a limited amount of this kind of external stress, e.g. due to limited sorption sites. This may lead to a reduction of the layer process potential.

4.5.2.4 Specific source and resource vulnerability maps

The aim is to use the criteria from the intrinsic vulnerability assessment of an area, and to establish additional criteria to enable an adequate integrated vulnerability assessment for specific contaminants.

Specific layer properties are internal characteristics of a karst system and may partly coincide with the parameters for intrinsic vulnerability assessment (e.g. clay content, flow velocity). But, in the context of the S factor they are used for specific process evaluation and not for advection and hydrodynamic dispersion as in intrinsic assessment. Care must be taken that intrinsic assessment doesn't overestimate the importance of for instance, clayey layers. It is important also to keep in mind their specific attenuation potential.

For a common procedure using the European approaches for both intrinsic and specific vulnerability, the S factor has to be linked to the intrinsic overlying layers factor (O factor), in order to obtain groundwater resource specific vulnerability maps. For compiling source specific vulnerability maps (Fig. 16), it has to be linked to both the O factor and karst network factor (K factor). The S factor modifies and upgrades the values of both the intrinsic factors.

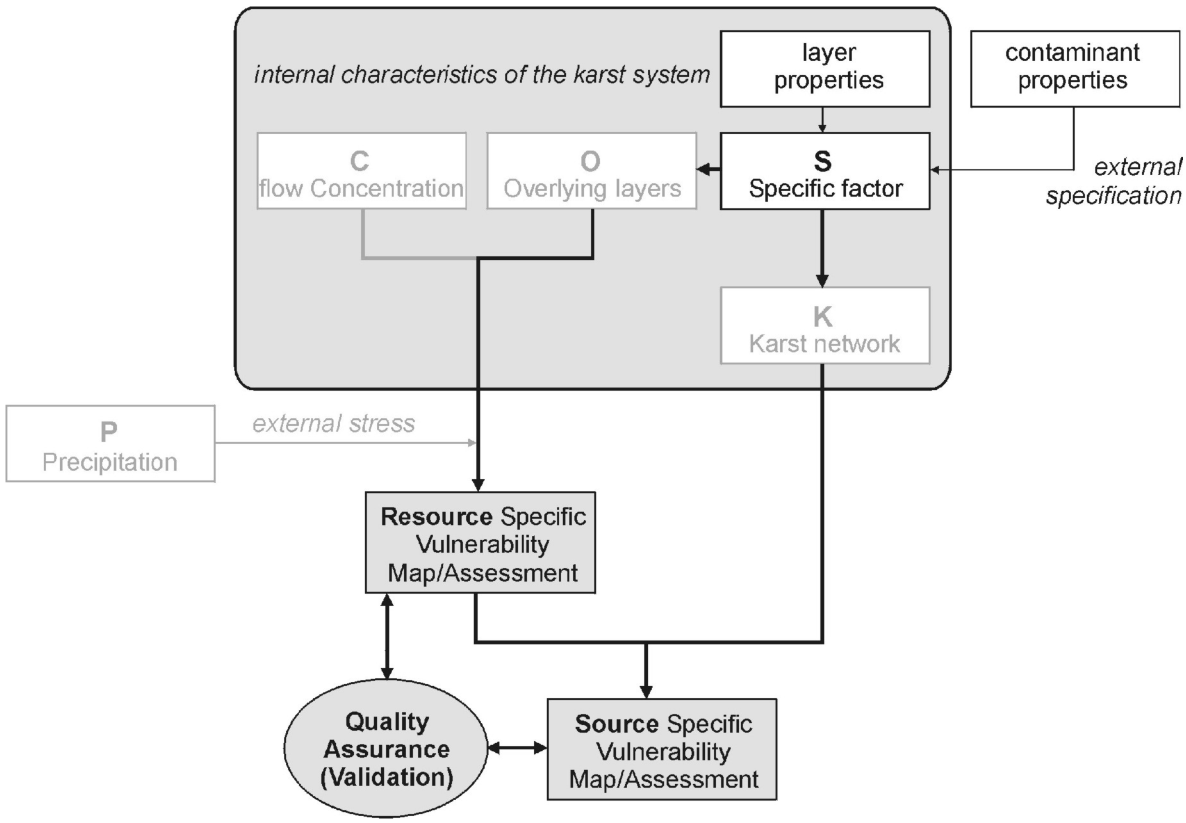


Fig. 16: The specific vulnerability weighting factor (S factor) in the framework of an integrated intrinsic-specific vulnerability assessment as an additional module to the European intrinsic vulnerability approach (see chapter 3) on preparing specific vulnerability maps.

An assumption for an efficient integration of the specific weighting factor into intrinsic vulnerability is the availability of compatible intrinsic databases. They allow an easy intersection and provide the necessary data. Specific vulnerability assessment must be based on a set of intrinsic information.

Specific vulnerability maps can be transformed into risk maps by the inclusion of hazards, the probability of contaminant spills and the value of groundwater resources.

4.6 Specific vulnerability method

4.6.1 Introduction

In the framework of COST Action 620, Working Group 2 has tried to devise a preliminary methodology for specific vulnerability assessment as one possible implementation of the previously mentioned "European approach". It is based on the principle that an additional module has to be integrated into the intrinsic approach for preparing specific vulnerability maps. The requirements for this method are (1) its suitability for specific vulnerability, (2) its applicability to karst environments, and (3) its compatibility with intrinsic vulnerability methods.

The proposed method aims to assess and rank, empirically and qualitatively, the physical and chemical processes relevant for specific contamination, rather than reflect a precise image of its transport and transformation behaviour. Despite these simplifications and limitations, such an approach provides a means to estimate and predict if significant attenuation of specific contamination can be expected in a given groundwater basin.

A systematic and standardised procedure is important for a consistent mapping method and for the comparability of the results. Working Group 2 has provided the layer and contaminant factors of the European approach (section 4.5) with weightings and ratings. This agreed procedure is based on the principles outlined in sections 4.2, 4.3 and 4.4. The output of the European approach, the specific weighting factor (S factor), is then expressed by a specific attenuation index, assessing the groundwater protection taking into account specific processes. A specific attenuation index has to be linked to the protection values of an intrinsic vulnerability method, which represents the intrinsic protection capacity provided by the processes of advection, hydrodynamic dispersion and dilution The systematic procedure demonstrates how the concept of the European approach can be implemented and shows what the results of such a method look like. It is not yet at its final stage of development and still requires upgrading and improvement. It is applicable to each type of potential contaminant and to a wide spectrum of karst environments. Moreover, it also is generally applicable to all other geological settings found in Europe.

Other qualitative methods may be developed from the European approach, as well as quantitative means of assessment. In the latter case, mathematical models may be used to describe the relationship between layer and contaminant properties. These assessments should determine the retardation and degradation effects on contaminant breakthrough curves caused by the different specific processes.

4.6.2 Assessment procedure

4.6.2.1 Procedure scheme

The standardised procedure of the specific vulnerability method aims at determining a positive specific attenuation index for each point in a catchment. Specific weighting factor maps may then show a spatial distribution pattern of specific attenuation for an individual contaminant. Combined with intrinsic protection values, specific vulnerability maps can be compiled.

Tab. 6: A 10-step plan for the implementation of the specific vulnerability method.

Step 1	Choice of contaminant and related processes
Step 2	Evaluation of contaminant process indices
Step 3	Evaluation of layer process indices
Step 4	Fitting of layer and contaminant process indices
Step 5	Process weighting
Step 6	Summation of processes
Step 7	Consideration of layer thickness
Step 8	Consideration of hydraulic properties
Step 9	Summation of layers
Step 10	Specific attenuation classes

The whole procedure stems from the philosophy of a process-based evaluation. This means that it is undertaken separately for each specific process in each existing layer, due to the individual process potential of the strata. One contaminant may underlie several specific processes in a single stratum, each of them inducing certain attenuation. The combination of all the processes in all the layers represents the specific attenuation of a contaminant. This attenuation occurs along its subsurface pathway from the contamination source, where it has been released, until it reaches the observed target. For the resource, the characteristics of each layer down to the groundwater surface, has to be taken into account, namely topsoil, subsoil, non-karst rock and the unsaturated zone of the karst. For a source (spring or pumping well), the saturated karst layer has to be included.

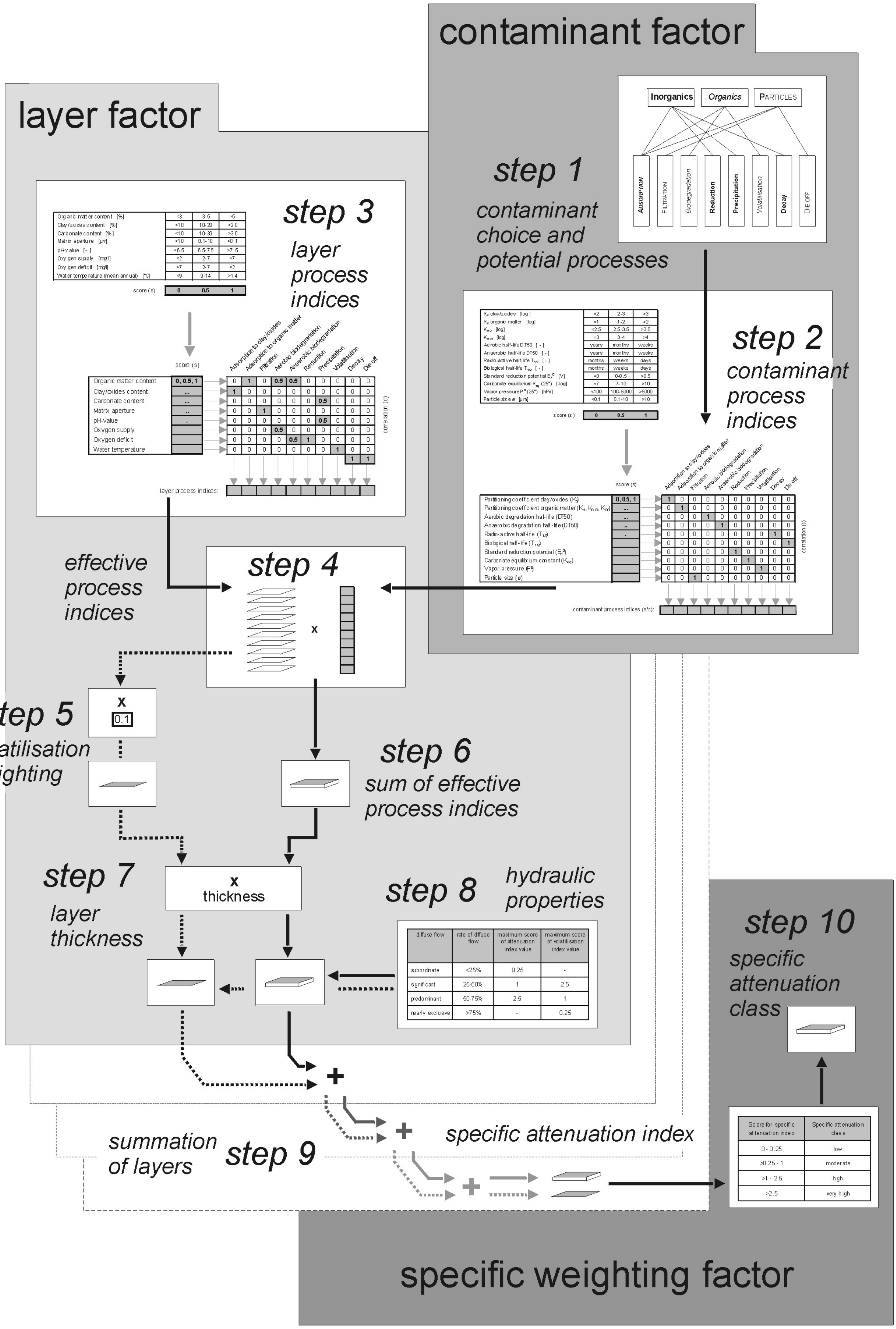


Fig. 17: Procedure for specific attenuation evaluation.

While the saturated karst layer may be very important for intrinsic vulnerability (high advection, high dilution), it plays only a minor role for specific processes, due to their very low contribution in saturated conduit flow. Therefore, specific maps for source protection are unlikely to differ significantly from those for resource protection.

Each specific contaminant requires an individual attenuation assessment. This is undertaken by means of a standard 10-step procedure (Tab. 6). The procedure represents a parametric system model, using matrix and rating systems, each of them divided into a series of intervals. The individual steps, which form the basis of the method, are shown in Fig. 17 and are described in detail in the following sections. The procedure must follow the proposed sequence; any change in step order may lead to misinterpretation and invalidate the approach.

The basic concept of the method stems from three factors: a contaminant factor (steps 1 and 2), a layer factor (steps 3 to 9) and a resulting specific weighting factor (step 10). Firstly, a contaminant is chosen for assessment (step 1). The aim of steps 2 and 3 is the evaluation of the contaminant process indices and layer process indices. That means that each relevant specific process is tested in respect of its effect on a contaminant under the given environmental conditions. The matrices from sections 4.2 and 4.3 are used for this purpose (Tab. 2 and Tab. 3), even though in a modified and simplified form.

The assessment has to take into account that for the occurrence of specific processes the properties of both the layer and the contaminant are important. A process may only occur if both the contaminant index and layer index enable its activation (step 4). Hence, step 4 is the link between the contaminant factor and the layer factor of the conceptual European approach.

The further procedure transfers the separate processes into a common attenuation value (step 6) and integrates the geometric and hydraulic properties of the strata (steps 7 and 8). Due to its particular role, the volatilisation process is treated separately but parallel to the other processes for these steps. As the procedure relates to a single layer only, it must be repeated for each layer present (step 9). The final specific attenuation classes (step 10), representing the specific weighting factor of the conceptual model, are then the link to intrinsic vulnerability values.

The use of GIS is strongly recommended in undertaking the procedure so as to ensure a cost-effective and time-efficient result. For example, layer process indices are easily transferred by means of GIS to other specific assessments of the same area. Thus, intrinsic mapping (e.g. layer thickness or intrinsic protection values) databases are required for this procedure.

Note, that this specific vulnerability method does not integrate the kind of contaminant release (short term – long term, point – diffuse, contaminant mass). Since the reaction capacity of the subsurface is not infinite, the assessment refers to a temporary contamination input of a moderate contaminant amount. A second point is the convention that for parameters, which are not stable in time, the maps should be valid for mean storm events, i.e. mean bad hydrological conditions.

4.6.2.2 Step 1: Choice of contaminant and related processes

The choice of a specific contaminant depends on the current land use of the area to be mapped. For example, specific vulnerability assessments for agricultural areas could include microorganisms, nitrate and/or pesticides, but this combination of potential contaminants is unlikely to be appropriate for an industrial or urban area. The contaminant choice results in a general reduction of possible specific processes, depending on the nature of the substance (Fig. 18):

- Adsorption, reduction, precipitation and decay for inorganic contaminants;
- Adsorption, biodegradation and volatilisation for organic compounds;

- Adsorption, filtration and die off for microbiological organisms.

Thus, several processes were classified as being relevant. For simplicity, only these relevant processes are taken into account and included in step 2 (Fig. 19). Besides, if one of them is known not to be active it can be eliminated in the first step so as to simplify the rest of the assessment. The evaluated processes should also be tested in respect of their real attenuation effect (e.g. some metals are more mobile in a reduced state).

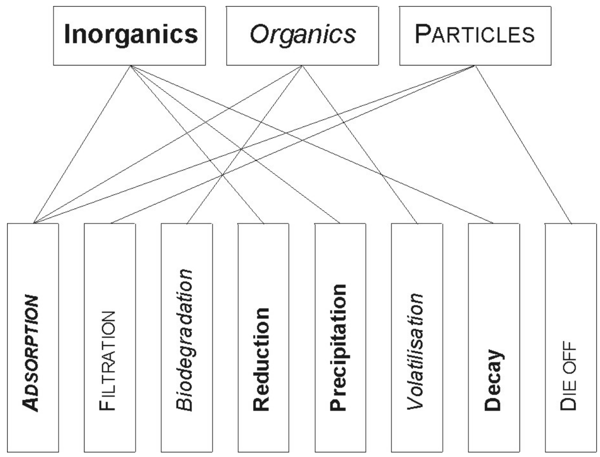


Fig. 18: Selection of processes relevant for inorganic, organic and particle contaminants.

At the present stage of development of the proposed method the following processes are excluded: Cation exchange is very similar to other sorption reactions, both from the contaminant and layer perspective, and is thus included in the adsorption process. Oxidation and complexation are not taken into account because most inorganics are more mobile in an oxidised form or as a complex. Furthermore, contaminants coming from the land surface are mostly already in an oxidised state. For organics, oxidation and complexation procedures as well as hydrolysis are included in biodegradation. Sedimentation of particles only takes place in the conduits of the saturated zone, but with a very low remobilisation limit. A long-lasting settling of adsorbed contaminants is already taken into account within the adsorption process.

4.6.2.3 Step 2: Evaluation of contaminant process indices

Once a specific contaminant has been chosen and the possible related processes established, the potential of these processes has to be ascertained. Each process is regarded as dependent on a particular contaminant property. Fig. 19 contains a matrix of processes and contaminants, with one key property for each process, identified as affecting a given contaminant. For example, biodegradation is considered as being dependent on degradation half-life and filtration being dependent on particle size.

	score (s)	Adsorption to clay/oxides	Adsorption to organic matter	Filtration	Aerobic biodegradation	Anaerobic biodegradation	Reduction	Precipitation	Volatilisation	Decay	Die off
Partitioning coefficient clay/oxides (K_d)	0, 0.5, 1	1	0	0	0	0	0	0	0	0	0
Partitioning coefficient organic matter (K_d, K_{OW}, K_{OC})	...	0	1	0	0	0	0	0	0	0	0
Aerobic degradation halt-life (DT50)	...	0	0	0	1	0	0	0	0	0	0
Anaerobic degradation half-life (DT50)	..	0	0	0	0	1	0	0	0	0	0
Radio-active half-life ($T_{1/2}$)	.	0	0	0	0	0	0	0	0	1	0
Biological half-life ($T_{1/2}$)		0	0	0	0	0	0	0	0	0	1
Standard reduction potential (E_H^0)		0	0	0	0	0	1	0	0	0	0
Carbonate equilibrium constant (K_{eq})		0	0	0	0	0	0	1	0	0	0
Vapor pressure (P^0)		0	0	0	0	0	0	0	1	0	0
Particle size (ø)		0	0	1	0	0	0	0	0	0	0
contaminant process indices (s*c):											

correlation (c)

Fig. 19: Matrix for the relationship between contaminant properties and processes for calculating contaminant process indices. Scores to be introduced are taken from Tab. 7.

The key contaminant properties enable the calculation of the contaminant process indices, which establish the potential processes. The dependence on only one key parameter is expressed in the matrix by a correlation of 1 for the relationship between the contaminant properties and the specific processes. The absence of influence by other parameters is shown by a correlation of 0. Due to this simple kind of correlation (0 or 1, only one key parameter per process), the index for a potential process is the result of the correlation c multiplied by the property scores for the particular contaminant. For contaminant process indices, values between 0 and 1 may be calculated, whereas 0 means that the key property value is generally insufficient for the related process. A value of 1 indicates a high potential for the process, while 0.5 is an intermediate value.

Tab. 7: Intervals of contaminant properties with assigned scores. Scores for the contaminant concerned have to be introduced in Fig. 19.

K_d clay/oxides [log]	<2	2-3	>3
K_d organic matter [log]	<1	1-2	>2
K_{OC} [log]	<2.5	2.5-3.5	>3.5
K_{OW} [log]	<3	3-4	>4
Aerobic half-life DT50 [-]	years	months	weeks
Anaerobic half-life DT50 [-]	years	months	weeks
Radio-active half-life $T_{1/2}$ [-]	months	weeks	days
Biological half-life $T_{1/2}$ [-]	months	weeks	days
Standard reduction potential E_H^0 [V]	<0	0-0.5	>0.5
Carbonate equilibrium K_{eq} (25°) [-log]	<7	7-10	>10
Vapor pressure P^0 (25°) [hPa]	<100	100-5000	>5000
Particle size ø [µm]	<0.1	0.1-10	>10
score (s):	0	0.5	1

The contaminant property scores are the required input data for the generation of process indices. Tab. 7 provides guidelines for contaminant property scoring, with several intervals of property values. Assigned scores are 0, 0.5 and 1, representing unfavourable to favourable property values. Contaminant properties values may be obtained from the scientific literature. If no data is available, for safety a score 0 is assigned. Contaminant process indices are assigned only once, and are universally valid both for the ongoing assessment, and for any further specific vulnerability assessment in any region. If several substances have exactly the same input score scheme, they are comparatively, affected by the same processes. In that case (contaminant group of similar behaviour), a single assessment is valid for each of these contaminants and the resulting map can be used for each of them.

4.6.2.4 Step 3: Evaluation of layer process indices

The evaluation of the potential processes from the viewpoint of the layers is analogous to the contaminant process indices, but is a bit more complex. On the one hand, the relationship between layer properties and processes is regarded as less simple (Fig. 20), and on the other hand, the required data is more difficult to assess. However, only significant parameters are taken into account, even though other properties may have a certain additional influence. Interactions between different processes are ignored.

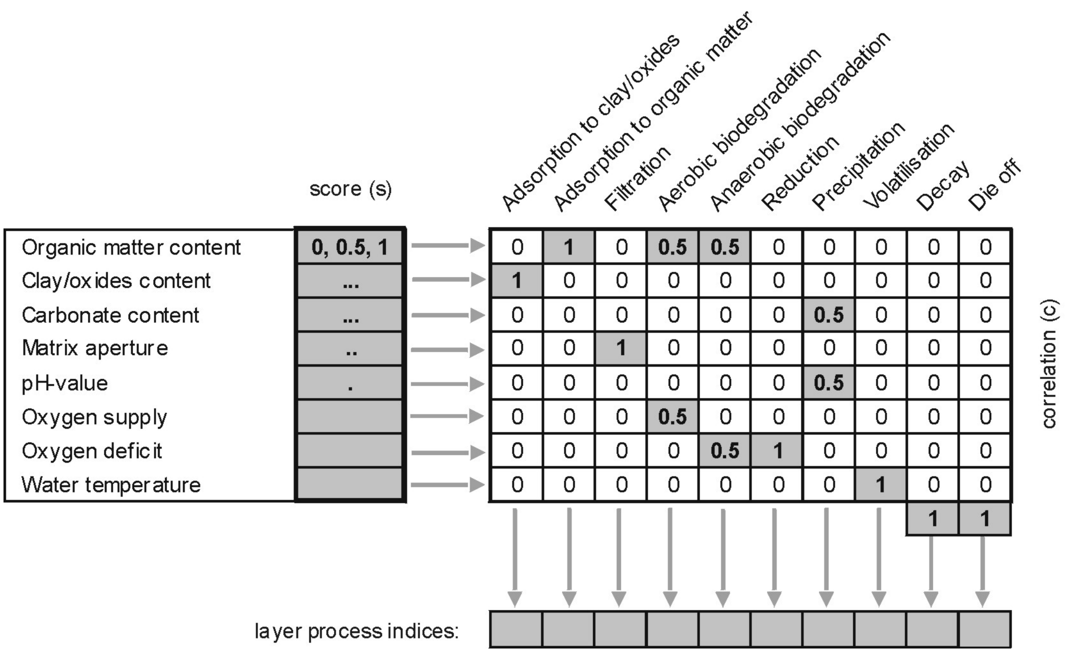


Fig. 20: Matrix for the relationship between layer properties and processes for calculating layer process indices. Scores to be introduced are taken from Tab. 9.

Specific processes are either independent from layer property scores (correlation = 0) or related to one (correlation = 1) or two of them (correlation = 0.5). A correlation of 1 means that there is only one key parameter, whilst 0.5 indicates that two equally important, internally interacting key properties are both essential for process activity. The differing correlations require different calculations for obtaining layer process indices, reflecting the diversity of layer property significance (Tab. 8). Decay and die off do not appear to depend on specific layer properties, so they always get a layer index of 1. Calculated layer process indices may range from 0 to 1.

Tab. 8: Calculation of layer process indices based on the correlation (c) between scores (s) for layer properties and processes.

(1)	no related property	layer decay index layer die off index	1
(2)	one single related property	layer adsorption indices layer filtration index layer reduction index layer volatilisation index	s^*c
(3)	two related properties	layer biodegradation indices layer precipitation index	$s_1{}^*c_1 + s_2{}^*c_2$ $= 0$, if s_1 or $s_2 = 0$

Layer property value intervals for obtaining input scores for the matrix have been defined based on the geological conditions found in European carbonate environments (Tab. 9). Each parameter must be assessed over the whole catchment and for each layer, attributing a score to each point or polygon of identical characteristics (measured or estimated on available data). Also a single representative score for an entire layer may be possible. Overall, the mapped scores should be somewhat underestimated so as not to imply too high an attenuation potential (especially in uncertain cases).

Tab. 9: Intervals of physical and chemical layer properties with assigned scores. Scores for the layer concerned have to be introduced in Fig. 20.

Organic matter content [%]	<3	3-5	>5
Clay/oxides content [%]	<10	10-20	>20
Carbonate content [%]	<10	10-30	>30
Matrix aperture [µm]	>10	0.1-10	<0.1
pH-value [-]	<6.5	6.5-7.5	>7.5
Oxygen supply [mg/l]	<2	2-7	>7
Oxygen deficit [mg/l]	>7	2-7	<2
Water temperature (mean annual) [°C]	<9	9-14	>14
score (s):	**0**	**0.5**	**1**

Knowledge about a contaminant reduces the need for field mapping. Depending on contaminant characteristics, several processes become insignificant and the number of layer parameters needed, decreases. If conversely, the mapping is done for the purpose of having a single database available for all contaminants, then all layer properties must be known.

4.6.2.5 Step 4: Fitting of layer and contaminant process indices

After steps 2 and 3, all processes are standardised to a maximum index of 1 both for contaminants and layers. Thus, both factors are equally weighted so that all processes possess the same relative importance. Although contaminant process indices provide a single global value for each process, the layer process indices may show a spatial distribution over the catchment and can be displayed as process index maps.

In the subsurface, a specific process only occurs, if both layer and contaminant enable its activation. That means that for both, a certain process potential is necessary. The objective of step 4 is to check how well the process indices of the layer and of the contaminant fit together and to identify those processes, which affect a contaminant. Thus, only layer and contaminant indices of the same process are combined.

Multiplying the process index of the layer with the matching process index of the contaminant provides the required value for a process, the effective process index (Fig. 21). Since a process index, in the context of this method, ranges between 0 and 1, the effective value can range from absent (0) to complete (1). A value 0 means that either the layer or the contaminant is regarded as insignificant for process occurrence (none or below a certain limit allocated in the framework of steps 2 and 3). This results in the elimination of the process in the assessment (for this layer only), due to no layer-contaminant interaction.

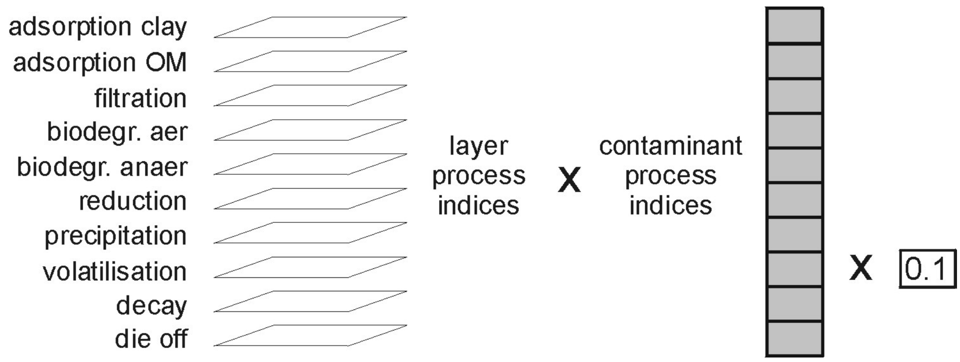


Fig. 21: Fitting of layer process indices and contaminant process indices. A weighting factor of 0.1 is used for the volatilisation process.

4.6.2.6 Step 5: Process weighting

Processes may be weighted one with another, taking into account their different contribution to attenuation. For instance, it can be argued that adsorption is usually much greater than reduction. However, the philosophy of this method is the equal weighting of all specific processes. Each process is valued as being able to contribute in the same degree to attenuation. Differentiation between the processes has already been made in the course of the layer and contaminant property rating.

The one exception is the volatilisation process, which plays a special role in the assessment. All other processes need slow laminar flow conditions, occurring in the diffuse matrix flow, with a substantial contact between the contaminant and the medium in order to be effective (see step 8). However, volatilisation is regarded as related to turbulent preferential flow in open voids, due to the fact that its reaction surface is the contact surface with the air instead of with a layer. Therefore a weighting factor of 0.1 is used for the volatilisation process, reducing its impact by a factor of ten with respect to the other processes (Fig. 21). This is done to lessen the risk of overestimating a process that has a much lower residence and reaction time.

4.6.2.7 Step 6: Summation of processes

The effective processes may complement each other, leading to an increase in the overall effects on contaminants. The sum of the individual effective process indices represents the specific attenuation as a whole.

Volatilisation is omitted from the summation of processes as it belongs to another flow system (preferential flow instead of diffuse flow). It will be included later after hydraulic layer properties have been considered.

4.6.2.8 Step 7: Consideration of layer thickness

The critical determining factor of the effectiveness of a protective layer is its thickness. The thicker the layer, the more effective is the interaction between the contaminant and the medium. A long pathway through a thick stratum extends the transit time and the contact surface, both necessary for contaminant affection. The previously described assessment values must be referenced to the layer thickness. This is done, by multiplying all the specific processes by the layer thickness, which results in an attenuation index. The spatial distribution of layer thickness may be directly taken from the already available intrinsic vulnerability data.

4.6.2.9 Step 8: Consideration of hydraulic properties

Step 8 deals with the hydraulic properties of the layer, which not only determine intrinsic processes, but also strongly influence specific contaminant affection. Two main flow systems may be differentiated in a karst environment, diffuse flow and preferential flow. Both flow types occur often simultaneously, but have rather different consequences for water flow and contaminant behaviour. Preferential flow occurs in the conduits of both unsaturated and saturated karst layers, but also is frequently present in overlying layers. This is due to a steep vertical gradient towards the karst drainage system on the one hand, and weathering, vegetation and animal influences on the other. Preferential flow is characterised by generally high flow velocities (order of magnitude: meters per hour), while diffuse flow travels slowly through the layer matrix (order of magnitude: meters per day or week). Water following preferential flowpaths migrates through a stratum bypassing its fine-pored matrix, without significant residence time and layer contact. Thus, contaminant bearing water flow short-circuits the protective function of the layer and hence reduces the effectiveness of most attenuation processes.

Diffuse flow is recognised to be an essential parameter for process effectiveness, providing sufficient availability of reaction surfaces. Therefore, a parameter "rate of diffuse flow" is introduced. The rate of diffuse flow, by this definition, is the *portion* of recharge, which takes a diffuse pathway through the layer matrix. This corresponds neither to the proportion of matrix porosity nor to the area through which the diffuse flow passes, which even in karst are both very high in comparison with preferential flow values.

Tab. 10: Ranges of "rate of diffuse flow" and assigned maximum score values.

diffuse flow	rate of diffuse flow	maximum score of attenuation index value	maximum score of volatilisation index value
subordinate	<25%	0.25	-
significant	25-50%	1	2.5
predominant	50-75%	2.5	1
nearly exclusive	>75%	-	0.25

Thus, the rate of diffuse flow may represent the portion of contaminant affected by the identified processes. This major simplification allows the estimation of the effectiveness of the process, with respect to hydraulic layer properties. In this context, the rate of diffuse flow is used to define a simple model of maximal attenuation index values. The achievable attenuation index score is dependent on the value of the rate of diffuse flow. Diffuse flow may range from “subordinate” (rate <25%) to “significant” (25-50%) and “predominant” (50-75%) up to “nearly exclusive” (>75%). Each category refers to a maximal value in attenuation effectiveness of the layer concerned. The attenuation index must not exceed the maximum value assigned to the diffuse flow value, even if the previously calculated score is higher. Tab. 10 shows intervals of possible attenuation values. For instance, a layer showing predominant rate of diffuse flow gets a maximum attenuation index score of 2.5 points. If there is no limitation due to preferential flow, the attenuation score is completely valid.

The consideration of hydraulic properties requires additional work on the evaluation of the volatilisation process. Volatilisation cannot be merged with the other processes due to different flow preferences (preferential flow for volatilisation, diffuse flow for the remaining processes). Therefore the previous step needs to be done separately for volatalisation, but in reverse order: high rates of diffuse flow lead to low volatilisation effectiveness (see Tab. 10).

The rate of diffuse flow is a parameter, which is difficult to assess and impossible to measure. For this reason it must be dealt with indirectly by broad parameter intervals, and not as a strict yes/no-criterion. The basic idea is to have a means to avoid an overestimation of attenuation effectiveness, instead of trying to predict exact affects of contamination The objective is to avoid thick bypassed layers, being assessed as making a big contribution to contaminant attenuation.

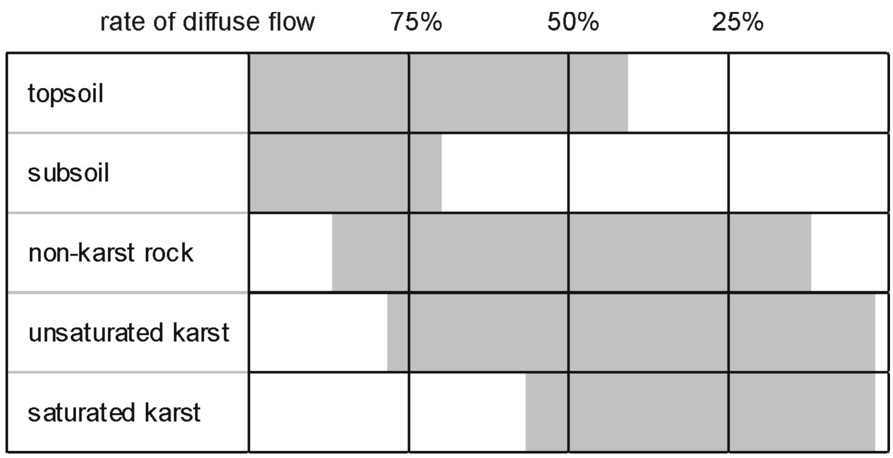


Fig. 22: Typical "rate of diffuse flow" in the different layers.

The range of diffuse flow may be estimated by means of existing data from intrinsic vulnerability mapping (hydraulic conductivity, dual porosity, layer thickness, geomorphology, layer type, karst development, hydrographs, infiltration tests, etc). Fig. 22 gives an overview of probable diffuse flow rates in different layer types.

4.6.2.10 Step 9: Summation of layers

Steps 3 to 8 are co-requisite for every layer of the system. The results can be added by superposing the different layers. For source vulnerability assessment, the saturated karst layer also must be included, although no major specific attenuation is expected for saturated groundwater flow.

In a few cases, the summation of layers may lead to an overestimation of the attenuation index. If several layers with major preferential flow are added, they may produce an attenuation index, which does not correspond to a major bypass. In such a case, the final score must be obtained following the procedure set out in step 8. The value is then based on the rate of diffuse flow of the least bypassed layer. The volatilisation index requires the same effectiveness check in respect of preferential flow and must not exceed the value for the layer with the highest preferential flow rate.

After the summation and the hydraulic effectiveness check, the volatilisation index can be added to the attenuation index for the other processes.

4.6.2.11 Step 10: Specific attenuation classes

The specific attenuation index is listed as a positive protection value due to the activation of specific processes. It results in a lowering of specific vulnerability compared to intrinsic vulnerability. The index is subdivided into four specific attenuation classes, which represent different score intervals (Tab. 11). The classes may be predicted in the following conditions:

Low specific attenuation: Specific processes do not occur or only to a minor degree, due to either low/absent contaminant or layer process indices, or very unfavourable flow conditions. Specific effects are considered as negligible.

Moderate specific attenuation: Specific processes are only slightly active. This may be due to low process indices, low layer thickness or limited values because of unfavourable flow conditions. In spite of noticeable attenuation, a given contaminant is unlikely to be significantly attenuated by specific processes.

High specific attenuation: Specific processes are significantly active. This requires favourable process indices, sufficient layer thickness and favourable flow conditions. A contaminant is likely to be significally attenuated compared to a conservative substance.

Very high specific attenuation: Specific processes are strongly active. High process indices are combined with sufficient layer thickness. No significant limitations due to flow conditions have to be taken into account. A contaminant is strongly affected by retardation and/or degradation.

Tab. 11: Classification for the specific attenuation index.

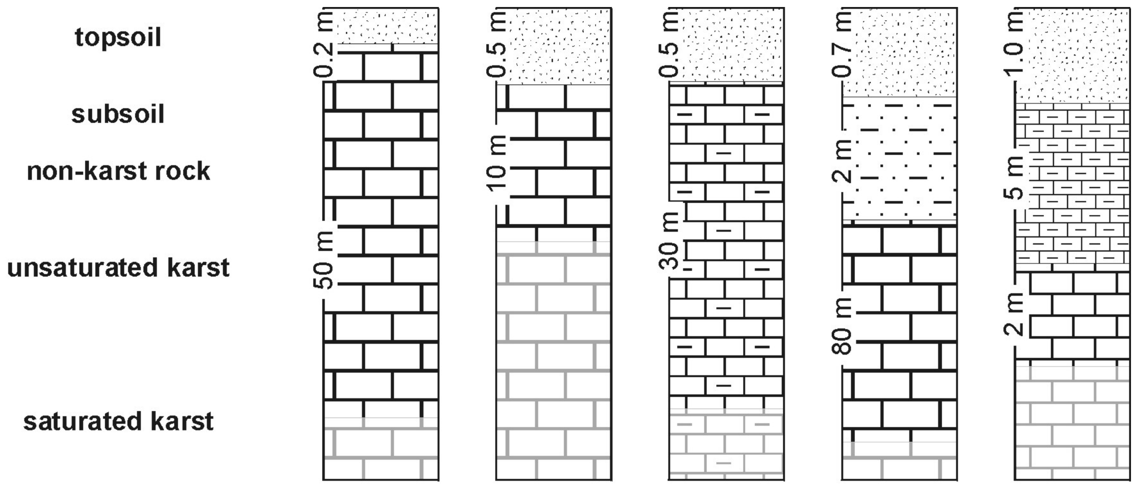
Score for specific attenuation index	Specific attenuation class
0 - 0.25	low
>0.25 - 1	moderate
>1 - 2.5	high
>2.5	very high

The attenuation index graduation is constructed in a non-linear way in order to allow for the importance of relatively thin but effective layers in karst environments. Another reason is to avoid high attenuation classes being reached too easily. The philosophy underlying this classification is the assumption that, under "perfect" conditions (both high layer process index and high contaminant process index), a process can moderately attenuate a released contaminant, which traverses a layer of at least 1 m (10 m for volatilisation). A high attenuation is suggested for a layer thickness of 2.5 m (25 m for volatilisation). Specific attenuation classes are associated with the hydraulic effectiveness of the system, allowing less flow bypass for increasing attenuation classes.

Attenuation used in this context is an umbrella term for retardation and degradation effects. However, this method cannot provide quantitative data about contaminant transit time or loss of contaminant mass as would be the case for a physically based approach.

4.6.3 Examples

An application of the empirical method to natural conditions allows for plausibility testing. The results of an assessment reflect the method quality with respect to experiences about contaminant migration in karst systems. Fig. 23 shows some hypothetical examples illustrating the specific attenuation diversity for several contaminants and geological settings.



Specific attenuation classes for resource protection

Lead	moderate	high	high	very high	very high
Zinc	moderate	moderate	high	high	high
Iodine 131	moderate	moderate	moderate	moderate	moderate
Nitrate	low	low	low	moderate	moderate
Tetrachloroethene	high	moderate	moderate	high	moderate
Benzene	very high	moderate	high	very high	moderate
Naphtaline	high	high	high	very high	high
Benzo[a]pyrene	moderate	moderate	moderate	high	high
Atrazine	low	low	low	moderate	moderate
Lindane	moderate	moderate	high	high	high
Isoproturon	low	moderate	moderate	moderate	moderate
Cryptosporidium	moderate	moderate	high	moderate	high
E. coli	moderate	moderate	high	very high	very high
Enterovirus	moderate	moderate	high	high	very high

Fig. 23: Specific attenuation classes for different contaminants in exclusively karst settings.

The method was checked in the field at two test sites, one in the Swiss Jura Mountains and the other in the Betic Cordillera of southern Spain. These applications are described in part B of this volume. They illustrate the general contrast between intrinsic and specific maps, as well as the differences in several specific maps. The confidence level of hypothetical or real examples represents an initial method validation. Nevertheless, additional validation tools are necessary in future in order to adequately test the proposed approach.

4.6.4 Limitations

The proposed method at its present stage of development is based on many assumptions and simplifications, most of them already mentioned or are obvious from the procedure description. Nevertheless, it seems necessary to enumerate them in order to show the limitations of the approach and to avoid misuse and misinterpretation. Note that the present module refers to specific processes only (neither advection, nor hydrodynamic dispersion or dilution), although the use of internal parameters such as layer thickness or hydraulic properties may suggest otherwise.

Uncertainties undoubtedly derive from system limitations. A qualitative assessment is unable to reflect the actual subsurface behaviour, so conceptual simplifications are inevitable. The rating and weighting are focused on the most important processes and parameters using a qualitative judgement. Parameters of minor importance are discounted for practical reasons, so many such potential parameters have not been taken into account. For example, interactions between different processes were omitted. Thus the method only takes into account those influences, which contribute to significant attenuation results. It is neither intended nor realistic to seek to fully represent the complexity of specific processes. In order to minimise the consequences due to uncertainties in mapping, the ranges for mapping parameters have been kept relatively broad.

Hydraulic property evaluation is also based on greatly simplified considerations, with a discrete differentiation between diffuse and preferential flow. The absence of residence time evaluation for matrix flow introduces uncertainties, especially for decay and die off processes, even if general residence time is provided for through intrinsic assessment. The fact that several of the processes are partly reversible is not very important, due to the associated retardation effect.

Some of the input parameters are not stable in time. Seasonal and climatic variations of temperature, pH value etc., and also of hydraulic characteristics lead to dynamic input parameters, causing a disagreement as regards static maps.

In certain cases, the process activity may not assist groundwater protection. For instance, reduction of individual inorganics increases their mobility, as well as contaminant adsorption on colloids. Another important aspect is the production of metabolites (biodegradation) or daughter elements (decay) representing a secondary hazard. They may be more dangerous than the original substance (e.g. vinylchloride from tetra- or trichloroethene) and should theoretically be subjected to a second specific assessment with a new contaminant property input. Such possible effects have to be checked in the framework of step 1.

Specific attenuation or vulnerability classification does not predicate the danger of breaching groundwater limit values at the target concerned. This danger depends on the contamination scenario, which is not specified in the method, as well as on the toxicity of a contaminant. Those aspects have to be carried out in a comprehensive risk assessment.

4.6.5 Link to intrinsic vulnerability

Although the specific module possesses its own significance, intrinsic vulnerability provides the basis of an integrated specific vulnerability assessment. The link between intrinsic and specific modules is thus a final, but essential, step. Intrinsic methods, used as the basis for the specific module, should be compatible with the intrinsic European approach. This helps to facilitate the combination and to obtain more conclusive results.

The link to an intrinsic assessment is obtained by upgrading the protective cover function (O factor), by using the calculated specific attenuation index to represent the specific weighting factor (S factor). For source protection, the S factor must be combined with both the O factor and the K factor, the latter taking into account the karst network of the saturated zone. Due to the particular features of each intrinsic method, no standard procedure for combining the intrinsic and specific modules can be provided. This in turn leads to a slight difference in importance of a specific module relative to an intrinsic one, depending on each intrinsic method.

The specific attenuation index classification is in principle designed for integration with a five-category intrinsic protection classification or intrinsic vulnerability classification respectively (very low to very high). A low specific attenuation is regarded as insufficient to upgrade a protection class (downgrade a vulnerability class). A moderate specific attenuation index raises the protection by one class (e.g. high to very high). Likewise, a high specific attenuation raises the protection level by two classes and a very high specific attenuation by three classes (e.g. low to very high). The influence of specific attenuation varies slightly with regard to the linked intrinsic method. Specific vulnerability maps (integrated intrinsic and specific modules) reflect both intrinsic and specific assessments. However, it is unlikely that a situation exists where very low intrinsic protection occurs along with very high specific attenuation (except for volatilisation). Specific assessment is likely to produce a moderate change by increasing already existing tendencies. Nevertheless, a relatively thin soil cover overlying the karst layer, as it is very common in carbonate environments, may already be sufficient to provide effective specific attenuation (see Fig. 23).

This kind of combination provides a balanced weighting of both modules taking into account that both intrinsic and specific protection can cause contaminant attenuation on their own. The intention is to illustrate the independence of the specific module. However, if the specific module is based on quantitative retardation and degradation evaluation, a link by multiplication with intrinsic travel time and concentration attenuation values seems appropriate.

Two intrinsic vulnerability methods were used for integrated specific vulnerability assessment in the framework of COST Action 620 (see part B of this volume). The COP method was used to produce specific vulnerability maps of the Sierra de Líbar in southern Spain. The specific attenuation factor was linked to the O score of the COP method, taking into account its special rating system. Then, the modified O score was treated in the same way as in intrinsic assessment, namely multiplication by the C and P factors. The Vaulion basin in the Swiss Jura Mountains was mapped using the VULK intrinsic vulnerability method. The resulting specific attenuation maps directly lower the intrinsic vulnerability classes due to the absence of the C and P factors, which have not yet been included in the VULK mapping method at its present stage of development.

4.6.6 Concluding remarks

A module for specific attenuation evaluation is presented, since the strategy of COST Action 620 was to separate intrinsic and specific vulnerability approaches. The objective is to combine it with an intrinsic vulnerability method, while benefiting from the intrinsic work. The method should be able to deal with the complexity of contaminant subsurface behaviour, yet be simple enough to be suitable for all contaminant types. Many simplifications and limitations had to be accepted to reach this goal. However, the examples and field tests show that the developed procedure provides reasonable results.

The proposed procedure aims to minimise the risk of the overestimation of specific attenuation by the use of three different filters: Firstly, the potential of the underground to allow specific processes and secondly, the potential of a contaminant to be affected by those processes, are taken into account. This ensures significant effects only are utilised, what tends to underestimate process effectiveness. Thirdly, the parameter "rate of diffuse flow" plays the role of a filter, by preventing the overestimation of the protective function of flow-bypassed layers.

The highest specific attenuation normally belongs to the topsoil and to some unconsolidated subsoil deposits. But the method also takes into account the specific process potential of other layers in relation to favourable contaminant properties. Finally, several contaminant types may show similar attenuation values, although derived from completely different processes in a different layer.

Another option for implementing the European approach guidelines is to divide the retardation and degradation processes. These effects could be evaluated by means of a simple flow and transport model, allowing the use of physically based process equations. Such a model could also be an appropriate tool for integrating specific vulnerability into a comprehensive risk assessment by simulating contamination scenarios (see section 1.3 in part B: the VULK model).

The present procedure is not a final method, but a preliminary test of the feasibility of the European approach for specific vulnerability assessment. Besides ongoing development and improvement, further mapping examples involving new contaminants and different geological settings are still needed. Furthermore, the use of validation tools is clearly envisaged. One possibility is computer modelling, as previously mentioned. Other techniques are artificial and natural tracing for simulating the behaviour of specific contaminants.

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5 Hazard Mapping

5.1 Vulnerability in a Risk Framework

Defining hazard and risk is difficult, firstly because it is only in recent times that the terminology has started to be formalised, and secondly because in the literature these concepts are usually being considered in relation to human health and safety. Also, hazard and risk are overused terms with a wide variety of definitions. Moreover, both terms are often used simultaneously in English and French, for instance. Other usage sometimes confuses risk with probability or chance of an event occurring. Therefore, in order to be useful, the terms 'hazard' and 'risk' have to be defined unambiguously. Generally, definitions available in literature do indicate that neither of the terms is intended to provide an absolute measure, but should rather be understood as a means of relative measure or comparison. These initial observations resulted from the preparatory work carried out by WG0 'Definitions' at the outset of this Cost Action.

As explained in more detail below, WG0 also advised that maintaining a closely defined distinction between hazard and risk becomes particularly useful when considered in relation to the *origin-pathway-target* model, as a framework in considering the prevention/ minimisation of contamination and the protection of groundwater.

A Royal Society (London) Study Group (1992) formally defined an **environmental hazard** as "an event, or continuing process, which if realised, will lead to circumstances having the potential to degrade, directly or indirectly, the quality of the environment". Consequently, a hazard presents a risk when it is likely to affect something of value (the *target*).

In the ***origin-pathway-target*** model, the risk of contamination of groundwater depends on three elements:

- the hazard posed by a potentially polluting activity (equivalent to *origin*)
- the intrinsic vulnerability of groundwater to contamination (equivalent to *pathway*)
- the potential consequences of a contamination event (the *target* is the groundwater)

Within this framework, the power and value of the vulnerability concept is best achieved when vulnerability assessment and mapping are used as a component of the process of achieving EU and/or national environmental objectives and groundwater protection. Achieving environmental objectives (for surface water, groundwater and ecosystems) and groundwater protection requires an assessment of the risks posed from existing and future human activities, together with responses to the risk in the form of measures designed to mitigate and manage the risk, and achieve the environmental objectives.

In a karst groundwater protection context, **risk assessment** requires:

- Identification of potential hazards. For existing hazards, this can be achieved by hazard mapping.
- Analysis of the potential impact of hazards on groundwater. This requires details on the contaminant concentration and quantity. A system for rating and weighting hazards can assist this process.
- Information on the hydrogeological characteristics of geological materials beneath hazards, which influence contaminant movement and attenuation. This is shown by

the vulnerability of the groundwater, either by means of vulnerability maps or vulnerability assessments.

- Information on the value of the groundwater. However, karst groundwater is considered to have a high value, and therefore this factor is not considered further in this report.

Risk management is based on an analysis of these elements followed by a response to the risk indicating guidelines on the acceptability of potential hazards, investigation requirements, monitoring requirements and, where appropriate, the likely planning or licensing scenarios. Vulnerability is a critical factor in influencing the level of response.

This Chapter focuses on the preparation of hazard maps, and follows a 7-step work plan (Fig. 24), starting from an Inventory of Hazards and leading to the eventual production of Hazard Maps. Chapter 6 provides a brief review of approaches to integration of hazard analysis and mapping with vulnerability assessment and mapping to assist in the risk assessment and management processes.

Step 1 Definition and Inventory of Hazards
Step 2 Hazard Data Requirements
Step 3 Rating and Weighting of Hazards
Step 4 Graphical Interpretation
Step 5 Mapping Techniques
Step 6 Data Evaluation
Step 7 Production of Hazard Maps

Fig. 24: Work plan for the preparation of hazard maps

5.2 Professional Needs and Applications of Hazard Maps

Several of the meetings held by the members of Working Group 3 also included site visits to both public and private agencies that are involved with the production of hazard and risk maps, with the purpose to investigate the different professional needs and applications of these maps. As a general outcome, the meetings highlighted the need for explicit recommendations, which would recognize and address legal, decision making, planning, scientific and consultancy viewpoints and perspectives. From an application point of view, the maps are considered essential tools for both national and local entities with relevant responsibilities at the planning and decision-making level. At the consultancy level, applications appeared to be concerned mostly with the production of site-specific hazard and risk maps as an integral component of environmental impact studies. Important questions arose on the professional needs from a legal perspective: e.g. does the law require these maps to be produced and in what context; does the law specify what these maps should consist of and which if any specific methodology should be used to produce the maps.

The European Water Framework Directive (2000) calls for the "need for common aspects to be described and regulated in relation to vulnerability mapping and pollution sources". In this regard, the concept and approach to the mapping of hazards, as developed by WG3 in Cost 620, clearly advocates a pro-active approach which would undoubtedly prove more cost effective than relying on remediation. Consequently, the hazard assessment methodology developed here is believed to be more useful in comparison to investigating "pollution sources".

Other observations, which were voiced at the meetings, included the apparent reluctance for authorities to commission maps, which explicitly show the term "risk", so as to avoid land speculation, loss of land value among other reasons. Most importantly perhaps, the main concern that was put forward by each and every organization, whether public or private, was that hazard and risk maps must be simple if they are to serve as efficient tools in planning and decision-making processes. Undoubtedly, the meetings provided a strong impetus to continuously refine our understanding of the professional needs and understanding of hazard and risk maps.

5.3 Hazard Analysis and Mapping

5.3.1 Step 1: Definition and Inventory of Hazards

5.3.1.1 Definition

In the context of groundwater contamination, a hazard is defined as a potential source of contamination resulting from human activities taking place mainly at the land surface. Many human activities directly affect our environment. However, the main impacts are considered to emanate from the handling of harmful substances, through their production, transport, storage and disposal as well as their use and application in a wide range of activities. Endangering the environment and, in our case, placing the groundwater at risk, can result from any such type of targeted activities, often aggravated by unintentional measures or worse still by accidents that may lead to a release of toxic and harmful materials into the environment.

A hazard assessment considers the potential degree of harmfulness for each type of hazard. It is determined by both the toxicity and the quantity of harmful substances, which may be released as a result of a contamination event. Of course, the likelihood of a contamination event, in which case contaminants are actually released into the environment, depends on many factors, including for example the age and/or condition of industrial plants and ancillary facilities. In particular, safety as well as security measures can be used effectively to reduce the likelihood of contamination.

5.3.1.2 Hazard types and hazard classification

In relation to groundwater, a hazard could be a human activity, an installation or even a physical area. Such a characterization does not necessarily comprise a fixed location, e.g. as in the case of transport of a toxic liquid from one place to another, nor does it include a time restriction. It could be just for a short period, temporarily or more or less permanent. Thus, in order to attempt a classification of hazards, a systematic registration of different hazards needs to be based on clear criteria.

Possible criteria could be the time and duration a hazard is posed, the type of human activity or the nature of the harmful substances, based for example on a more general classification due to their gaseous, liquid or solid state or due to their specific chemical composition.

From a groundwater protection point of view, it is typically very important to know where a hazard may occur. Less important would be to know when activities that may pose a threat occur, but rather to have precise information on the location or areas associated with activities representing a hazard to groundwater. Using this criterion provides the opportunity to locate the different hazards on maps, and thus enables the hazard information to be compared and related to other spatially distributed data, such as the hydrogeological properties of the underlying rock sequence. Hazards can be of point like dimension, linear or aerial extent. There is no sharp boundary between these three types, but commonly rather a question of the mapping scale.

Consequently, it is proposed that the hazards should be classified according to the type of land use. A general differentiation of the land use on a local or regional scale distinguishes between three main categories, i.e. infrastructural, industrial and agricultural activities. These main categories are proposed as Level I Categories of Hazards in the Hazard Inventory (Tab. 12).

The proposed Level II Categories (Tab. 12) make use of additional criteria, which distinguish between hazards according to the main source (solid or liquid contaminants) of possible groundwater contamination, or else refer to types of industrial or agricultural activities with their corresponding spectrum of possible pollutants. A further subdivision into Level III Categories is shown in Tab. 13.

It is not the intention to make as many subdivisions as possible in our hazard inventory. Some environmental agencies provide inventories in which several thousands of specific branches of different hazardous activities are listed.

The main aim of the hazard inventory proposed here is to cover all the various hazards that are considered relevant to groundwater and to allow, through a reasonable subdivision, the mapping, evaluation and assessment of the hazards in an economically feasible and practical manner. To deal with hazards that may be very specific to the area under investigation, the user of these guidelines is encouraged to extend the list of hazards shown in Tab. 13.

The main impacts resulting from these activities and endangering the karst systems are:

- emission of air pollution;
- discharge of waste water and non aqueous organic liquids;
- storage and disposal of solid waste;
- excavations in connection with mining, foundation and construction work;
- distribution of fertilizers and pesticides.

Tab. 12: Classification of hazards for groundwater protection purposes

Number	Level I Categories of Hazards	Level II Categories of Hazards
1	Infrastructural development	
1.1		waste water
1.2		municipal waste
1.3		fuels
1.4		transport and traffic
1.5		recreational facilities
1.6		diverse hazards
2	Industrial activities	
2.1		mining (in operation and abandoned)
2.2		excavation sites
2.3		oil and gas exploration
2.4		industrial plants (non-mining)
2.5		power plants
2.6		industrial storage
2.7		diverting and treatment of waste water
3	Livestock and agriculture	
3.1		livestock
3.2		agriculture

5.3.2 Step 2: Hazard Data Requirements

Assessing the potential degree of harmfulness for each type of hazard requires information on the following:

- process or nature of activity (production, storage, etc.);
- type of harmful substances;
- amount of substances which can be released;
- age and status of installations and plants.

The collection of information on the various hazards is likely to be based on the combined use of:

- extraction from topographic maps;
- evaluation of aerial photos;
- collection of data from archives and agencies;
- field surveying;
- direct inquiries with companies, etc.

The various data required to perform a detailed hazard assessment have been grouped in Data Collection Cards. Such cards have been elaborated for each of the fifteen types of hazards listed as Level II Category of Hazards in Tab. 12. Since the assessment is performed in a GIS environment, information about the precise location (including map coordinates) and land-take (aerial extent) is required for all hazards. The latter implies that only hazards that can be georeferenced are taken into consideration.

An easy-to-use software, developed by Massimo Civita and Giuseppe Sappa as a specific activity of WG3, which can feed directly into common GIS systems, facilitates the collection of data on hazards and thus the eventual production of hazard maps. The PC programme generates a spreadsheet type of file in which all data pertaining to a specific hazard are stored in a single row. Transfer to a GIS is therefore assured directly (Excel file importation) or alternatively after first saving the spreadsheet as an ASCII text file. A more detailed description of the software is provided with the CD-Rom included with this publication.

As shown by the software coversheet (Fig. 25), each of the 15 hazards listed as Level II hazards in Tab. 12 are considered, and a specific data collection sheet is available for each of these hazard types. Fig. 26 provides an example of such a data collection sheet for wastewater. The information to be filled in each data collection sheet has been grouped according to the following criteria:

- Identification of the nature of the activity;
- Localization of the activity by topographic coordinates;
- Characterization and quantification of the solid and liquid waste production;
- Quantification of the water demand.

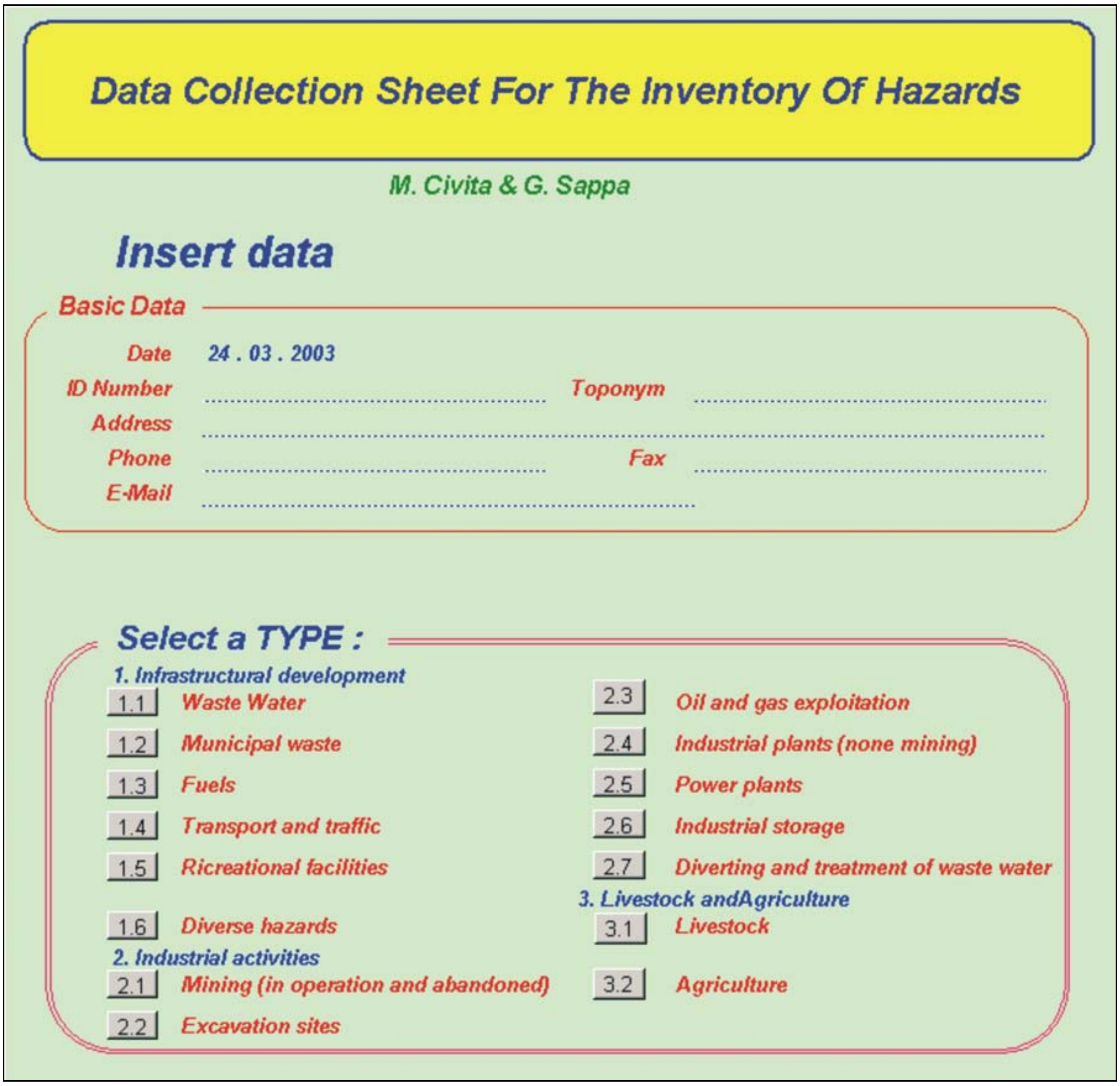

Fig. 25: Cover Sheet of Software Programme

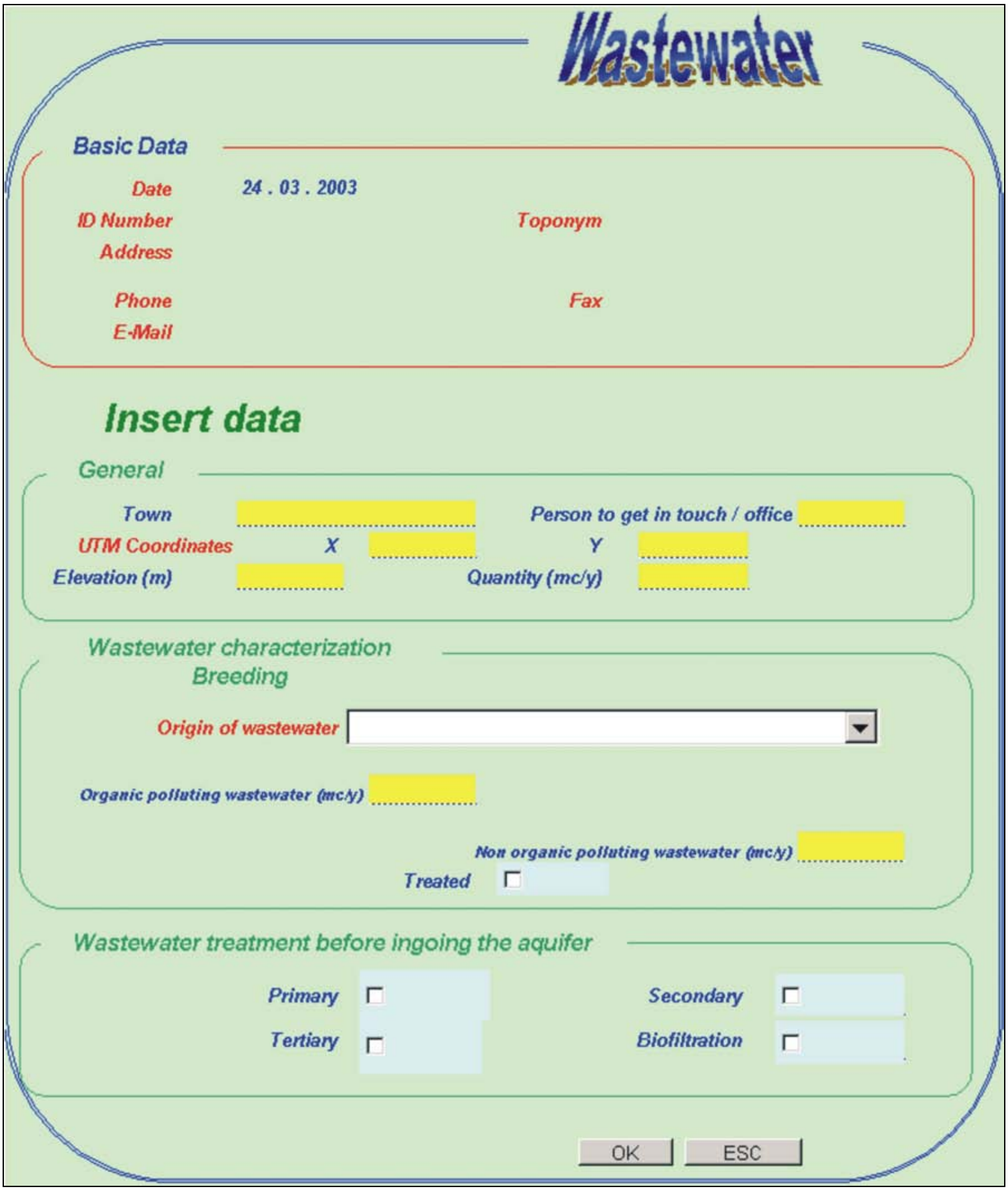

Fig. 26: Example for a Data Collection Sheet generated by Software Programme

The organization of the data collection sheets has deliberately been kept very simple. Each of the sheets begins with a section that is used to fill information about the type and geographic location of the activity. The next section of the data collection sheet requires data about the specific nature of the activity. Thus, this second section is directly used to characterize the activity that is being investigated. The third and final section is almost identical for each of the data collection sheets and deals with the collection of data concerning fuel use and storage.

The software programme is started with the sheet "***insert***" and the user is prompted to choose a specific kind of activity from a list of possible activities, which are also numbered. By clicking on the relevant, numbered button next to the name of the desired activity, the software goes directly to the insert sheet, where once again the user is prompted to fill the data pertaining to the activity in the relevant boxes. On completion of data entry, the user should click the *OK* button, positioned at the bottom of any of the sheets, so that the data previously entered are automatically saved. The software then saves the data in an ordered list, whereby each activity becomes identified by a numerical code. In doing so, the numerical code relates the hazard with its various properties to a geographic information system.

This means that the software has been purposely designed to allow the direct cartographic representation of data collected. Thus, not only maps showing simply the location of hazards but also maps representing the potential degree of harmfulness of these hazards are easily produced. Indeed, as can be seen from the classification of hazards proposed in Tab. 13, it is possible by any GIS software to represent the different hazards according to their intensity, providing an easy way to arrive at a hazard assessment map.

5.3.3 Step 3: Rating and Weighting of Hazards

5.3.3.1 Concept of rating and weighting

The possible impact of a hazard on groundwater and groundwater quality can vary considerably. Referring to the hazards listed in Tab. 13, a quantitative comparison of the potential damage or deterioration of the groundwater (and of the ecosystem in general) may differ over several magnitudes if an absolute scale system were to be used. One only has to reflect about the short-term bacteriological effect of a septic tank overflow compared to a large release from an organic solvent tank, or even with a long-term leakage from a hazardous or perhaps nuclear waste site. Even if only economic issues were considered, a direct comparison would be very difficult. Indeed, the difficulties of doing so would become even more numerous if human health and environment conditions are also included in such an evaluation and assessment.

Yet another aspect has to be considered as well: in order to come up with a reliable assessment of a hazard, all the necessary details of the influencing factors have to be determined. These include the amount and harmfulness or toxicity of the possibly involved substances as well as an assessment of all factors influencing the likelihood that a release may occur, e.g. state of preservation, maintenance or security measures. Apart from the fact that some of the data would be not directly available, it would require enormous efforts and manpower to collect all the data pertaining to the different hazards that are needed for a quantitative and systematic evaluation and assessment.

In contrast to such an absolute quantitative approach, groundwater protection schemes are rather simple schemes based on an easily comprehensible number of classes, commonly referring to associated codes of good practice. This has resulted from the experience that such a system obtains much more acceptance, as it is more understandable, of clear logic and leading to a fair solution for all affected groups. Hence, WG3 focused on using the same, conceptual approach with regard to hazard assessment and hazard mapping. This means that from the outset, the aim was to have a subdivision in rather few classes, which should permit a fast assignment and a good reproducibility of the main hazard categories as well as to guide the mapping of hazards (and hazard assessment) in a logical and clear manner, e.g. by advising on appropriate symbols to depict the different hazards.

For these objectives to be reached, the following steps are necessary in the hazard assessment computation to arrive at a classification of the hazards:

- establishing of a weighting system which allows comparison between the different type of hazards within a relative assessment scheme, and is suitable for the formulation of possible groundwater protection zones or measures;

- establishing of a ranking procedure for hazards of the same type, which fits the above mentioned criteria;
- developing of an assessment scheme for the determination of the likelihood that a release of contaminant may take place in connection with a hazard; and
- defining of a mathematical algorithm for the calculation of the potential degree of harmfulness for each hazard by considering the weighting and ranking coefficients as well as a probability term to represent the likelihood of a contamination event.

The data collection cards (paper format) or sheets (software programme) for the evaluation of each hazard (see Step 2) have to be adapted to such a rather simple assessment scheme. On the one hand, it should be sufficiently extensive to permit a reliable assessment of the harmfulness, while on the other hand great care should be taken to avoid the collection of data that are not necessary for the assessment. This is important in order to save both time and energy and thus to maintain a procedure that remains economically viable.

5.3.3.2 Weighting Procedure

It has already been mentioned that it is nearly impossible to establish quantitative methods to arrive at an absolute measure of the degree of harmfulness. The main criteria for weighting different hazards concern the toxicity of relevant substances associated with each type of hazards as well as their properties regarding solubility and mobility. They determine the weighting coefficient or the "harmfulness of a hazard to groundwater (H)". Three parallel approaches were considered by WG3 to estimate the harmfulness of each hazard type.

As a first approach, the hazard inventory list was sent to the national teams responsible for the test areas with the invitation to judge the importance of each type of hazards by assigning figures between 0 and 10 according to their relevance in the respective test areas.

As a second approach, preliminary weighting factors were calculated with a formula, which is used in connection with the Italian vulnerability assessment scheme SINTACS (Civita et al. 2000). This scheme is based on the EU Directive on toxic sources of contamination. A "contamination index (DCI)" is calculated for each hazard by adding up the potential "environmental impacts caused by specific human activities (ICEI)", which are considered the most important parameters to assess the degree of harmfulness for groundwater. The different environmental impacts are grouped according to the EU classification of waste hazards and the water demand generated by the human activities. Thus this second approach considers: special and dangerous substances, organic and non-organic wastewater, water demand and large surface contamination. The environmental impact associated with each of these causes of environmental impact is assigned a score ranging from 0 to 3 according to general experience on environmental impact. A further description of the second approach is provided in the Annex of this report, which also provides a discussion on the weighting values obtained by this second approach.

In a third approach, six experts were asked to assign weights to all the hazards appearing on the inventory list. They were asked to do so independently from the previous approaches and only according to their experience and the general toxicity. It was requested that values should be distributed with a range from 0 (not harmful) to 100, indicating extreme harmfulness. It was advised from the outset that this range should be understood to represent a kind of logarithmic scale, bearing in mind the very wide range of harmfulness that can be associated to the different types of hazards.

Finally, using the results obtained from each of the three approaches, a weighted average was calculated for every hazard type. The latter values are given as a general recommendation from Cost 620 when judging the potential degree of harmfulness of different types of hazards. The weighting values are listed in Tab. 13 and vary between 10 and 100. If it is necessary to extend the list of hazards provided by Cost 620, it is possible to follow the second approach to assign the new hazards with an appropriate weighting value.

5.3.3.3 Ranking Procedure

For a comparison between hazards of the same type, once again all the different factors influencing the degree of harmfulness have to be considered. According to the general definition of the hazard categories (Tab. 12), the hazardous substances involved within each individual category are more or less the same or can be considered to be from the same group. Therefore the differences in harmfulness within each hazard category will be mainly due to the variable quantity (Q_n) of harmful substances, which can be released and further seep into the underground.

Given the high priority that was assigned to the toxicity of the harmful substances in the average hazard weighting values, the ranking procedure should neither lead to a drastic minimization nor excess overvaluing within a same category of hazards. Indeed, a relatively small amount of highly toxic substances or alternatively a high amount of low toxic substances should not in itself lead to wide deviation from the average weighting value previously assigned to a particular type of hazards. Thus, in order to maintain a fair balance with the average weighting values, it is recommended that these weighting values should be changed only slightly by multiplying them with a ranking factor between 0.8 and 1.2 in order to indicate low or high amounts respectively of toxic substances compared with the general average.

Furthermore, for the ranking within a same hazard type category, it makes sense to leave it to the respective specialists whether to introduce a gradual variation between 0.8 and 1.2, or to limit the subdivision just to three classes, i.e. low (0.8), medium (1.0) and high (1.2).

5.3.3.4 Likelihood of groundwater contamination

Apart from the type and quantity of harmful substances associated with a hazard, the technical status, level of maintenance, surrounding conditions and security measures are important factors when assessing the probability that a real contamination of groundwater may occur. This likelihood has to be taken into account whenever the total, combined hazard level, expressed as the hazard index (HI) is to be estimated. Clearly, a quantitative determination of the likelihood of contamination differs strongly between the different hazard types. A real quantitative estimation might be very difficult to achieve, especially for industrial hazards but also for infrastructural hazards.

To avoid time consuming work without any guarantee for real success, a reduction factor R_f is recommended. This coefficient provides an assessment of the probability for a contamination event to occur. If no information on the above mentioned factors is available, then $R_f=1$. Otherwise, positive information concerning the reduction of the likelihood can be used to reduce the Hazard Index. Theoretically the reduction factor may range from 1 to 0. In a situation where the value is set to zero, it follows that it is considered that there is no risk of groundwater contamination, while a factor of 1 means that there are no reasons known to reduce the likelihood of an impact to the groundwater.

The reduction factor should be applied very carefully especially with regard to an increasing amount of reduction. The reduction factor is not a linear correction term, but it should be understood as a negative exponential term, which should be used for instance with the square root of the reduction value in the calculation formula. If such a restriction is not foreseen in the calculation formula then only small deviations from 1 should be used for the reduction factor. Otherwise such reductions could easily lead to a minimization of the effects of hazards with high toxic potential.

5.3.3.5 Calculation of the Hazard Index (HI)

The hazard index describes the degree of harmfulness of each hazard. For its calculation the following formula is recommended:

$$HI = H \cdot Q_n \cdot R_f$$

Whereby HI is the hazard index, H is the weighting value of each hazard as assigned in Tab. 13, Q_n is the ranking factor (0.8 to 1.2) and R_f the reduction factor.

As explained earlier, the square root of the reduction values should be inserted in the above formula, and in all instances only small deviations from 1 should be used for the reduction factor.

The possible range of the hazard index HI runs from 0 to 120 scores. For the subsequent interpretation of the hazard index values, a subdivision of no more than five or six classes is recommended.

Tab. 13 (first part): Hazard Weighting Values and Map Symbols

No.	Hazards	Weighting Value	Map Symbols		
			Marker	Line	Shade
1	**Infrastructural development**				
1.1	Waste Water				
1.1.1	urbanisation (leaking sewer pipes and sewer systems)	**35**			1
1.1.2	urbanisation without sewer systems	**70**			1
1.1.3	detached houses without sewer systems	**45**	1		
1.1.4	septic tank, cesspool, latrine	**45**	2		
1.1.5	sewer farm and waste water irrigation system	**55**	3		9
1.1.6	discharge from an inferior treatment plant	**35**	4		
1.1.7	surface impoundment for urban waste water	**60**	5		9
1.1.8	runoff from paved surfaces	**25**	6	1	
1.1.9	waste water discharge into surface water courses	**45**	7		
1.1.10	waste water injection well	**85**	8		
1.2	Municipal Waste				
1.2.1	garbage dump, rubbish bin, litter bin	**40**	9		8
1.2.2	waste loading station and scrap yard	**40**	10		8
1.2.3	sanitary landfill	**50**	11		8
1.2.4	spoils and building rubble depository	**35**	12		8
1.2.5	sludge from treatment plants	**35**	13		
1.3	Fuels				
1.3.1	storage tank, above ground	**50**	14		
1.3.2	storage tank, underground	**55**	15		
1.3.3	drum stock pile	**50**	16		
1.3.4	tank yard	**50**	17		11
1.3.5	fuel loading station	**60**	18		
1.3.6	gasoline station	**60**	19		
1.3.7	fuel storage cavern	**65**	20		
1.4	Transport and traffic				
1.4.1	road, unsecured	**40**		2	
1.4.2	road tunnel, unsecured	**40**	21		
1.4.3	road haulier depot	**35**	22		11
1.4.4	car parking area	**35**	23		11
1.4.5	railway line	**30**		3	
1.4.6	railway tunnel, unsecured	**30**	24		
1.4.7	railway station	**35**	25		
1.4.8	marshalling yard	**40**	26		
1.4.9	runway	**35**	27	2	
1.4.10	pipline of hazardous liquids	**60**		4	
1.5	Recreational facilities				
1.5.1	tourist urbanisation	**30**	28		2
1.5.2	camp ground	**30**	29		2
1.5.3	open sport stadion	**25**	30		3
1.5.4	golf course	**35**	31		3
1.5.5	skiing course	**25**	32		3
1.6	Diverse hazards				
1.6.1	graveyard	**25**	33		10
1.6.2	animal burial	**35**	34		10
1.6.3	dry cleaning premises	**35**	35		
1.6.4	transformer station	**30**	36		
1.6.5	military installations and dereliction	**35**	37		13
2	**Industrial activities**				
2.1	Mining (in operation and abandoned)				
2.1.1	mine, salt	**60**	38		7
2.1.2	mine, other non-metallic	**70**	39		7
2.1.3	mine, ore	**70**	40		7
2.1.4	mine, coal	**70**	41		7
2.1.5	mine, uranium	**80**	42		7
2.1.6	outdoor stock piles of hazardous raw material	**85**	43		6
2.1.7	ore milling and enrichment facilities	**70**	44		
2.1.8	mine waste heap and dirt refuse	**70**	45		6
2.1.9	ore tailings	**70**	46		6
2.1.10	mine drainage	**65**	47	5	
2.1.11	tailing pond	**65**	48		6
2.2	Excavation sites				
2.2.1	Excavation and embankment for development	**10**	49		
2.2.2	gravel and sand pit	**30**	50		12
2.2.3	quarry	**25**	51		12

Tab. 13 (second part): Hazard Weighting Values and Map Symbols

No.	Hazards	Weighting Value	Map Symbols		
			Markersymbol-Number	Linesymbol-Number	Shadesymbol-Number
2.3	Oil and gas exploitation				
2.3.1	production wells	**40**	52		
2.3.2	reinjection wells	**70**	53		
2.3.3	loading station	**55**	54		
2.3.4	oil pipline	**55**		4	
2.4	Industrial plants (none mining)				
2.4.1	smelter	**40**	55		4
2.4.2	iron and steel works	**40**	56		4
2.4.3	metal processing and finifhing industry	**50**	57		4
2.4.4	electroplating works	**55**	58		4
2.4.5	oil refinery	**85**	59		4
2.4.6	chemical factory	**65**	60		4
2.4.7	rubber and tyre industry	**40**	61		4
2.4.8	paper and pulp manufacture	**40**	62		4
2.4.9	leather tannery	**70**	63		4
2.4.10	food industry	**45**	64		4
2.5	Power plants				
2.5.1	gasworks	**60**	65		4
2.5.2	caloric power plants	**50**	66		4
2.5.3	nuclear power plant	**65**	67		4
2.6	Industrial storage				
2.6.1	stock piles of raw materials and chemicals	**60**	68		
2.6.2	containers for hazardous substances	**70**	69		
2.6.3	cinder tip and slag heaps	**70**	70		5
2.6.4	non hazardous waste site	**45**	71		5
2.6.5	hazardous waste site	**90**	72		5
2.6.6	nuclear waste site	**100**	73		5
2.7	Diverting and treatment of waste water				
2.7.1	waste water piplines	**65**		5	
2.7.2	surface impoundment for industrial waste water	**65**	74		9
2.7.3	discharge of treatment plants	**40**	75		
2.7.4	waste water injection well	**85**	76		
3	**Livestock and Agriculture**				
3.1	Livestock				
3.1.1	animal barn (shed, cote, sty)	**30**	77		
3.1.2	feedlot	**30**	78		
3.1.3	factory farm	**30**	79		
3.1.4	manure heap	**45**	80		
3.1.5	slurry storage tank or pool	**45**	81		
3.1.6	area of intensive pasturing	**25**			14
3.2	Agriculture				
3.2.1	open silage (field)	**25**	82		
3.2.2	closed silage	**20**	83		
3.2.3	stockpiles of fertilisers and pesticides	**40**	84		
3.2.4	intensive agriculture area (with high demand of fertilisers and pesticides)	**30**	85		14
3.2.5	allotment garden	**15**	86		14
3.2.6	greenhouse	**20**	87		14
3.2.7	waste water irrigation	**60**	88		14

5.3.4 Step 4: Graphical Interpretation

Commonly, the graphical interpretation of hazard data is obtained from a map, which shows spatial information such as their location and extent (size, shape), together with descriptive information, which are the map features or attributes. It is obvious that in some cases the actual size of a hazard cannot be presented due to its small dimensions. In such instances, the shape as a spatial information is lost and only the descriptive information remains as an attribute shown on the map. As a thematic map, the hazard map therefore shows the distribution and location of different kinds of hazards with a common defined attribute. The hazards are represented on the map by means of symbols or signatures of different colour to indicate their potential degree of harmfulness as derived from the calculation of a Hazard Index (see Step 3).

The accuracy of the map with respect to the location of the hazards is obviously dependent on the quality of the original sources that were used to determine their coordinate information. Thus, the scale at which the information is collected should be the same or more detailed as the eventual scale of the output map. While clearly dependent on the size of the area under investigation, the scale of the hazard map should also be chosen in agreement with the scales that were chosen in respect of the hydrogeological and vulnerability maps.

As discussed in detail in Section 7.3, it is recommended that the mapping be performed using a Geographic-Information-System (GIS), while the eventual production of high quality output maps is then obtained from Computer Aided Cartography (CAC) software.

5.3.4.1 Database requirements

A specified structure is needed for the graphical interpretation of the data stored in a GIS. Whereas raster data models contain features as a matrix of cells in continuous space, vector data models include points, lines, and polygons. To interpret the data in vector data models, the spatial nature of the data for the different hazards, depending on the scale of the graphical interpretation, has to be known. Most of the geographical information systems store the data in specialized file structures and every layer or subclass can only include one type of shape (e.g. either points, lines, or polygons). Thus it is generally expected that the hazards would be stored separately according to their spatial properties. It is also recommended that the hazards should be arranged in separate layers according to the hazard inventory list provided in Tab. 13 (e.g. layer 1.2.3 is used to include all sanitary landfills and layer 1.4.5 all railway lines).

Within one layer the data are organised in tables, in which the rows correspond to each of the individual hazards and the columns to their corresponding attributes. For the calculation and storage of the Hazard Index (HI), it is recommended that all required coefficients be entered in separate columns in the attribute tables. This means that in addition to the weighting value, ranking factor and reduction factor, it is recommended that the Hazard Index as well as the Hazard Index Class be stored in separate columns. As discussed in more detail below, this data structure permits an easy approach to display the different hazards with appropriate symbols, and in the colour corresponding to the relevant Hazard Index Class.

Finally, it is recommended that the output map should also show some other important attributes like the name and a short description of each hazard, thus providing additional information of a descriptive nature.

5.3.4.2 Map layout

a) Symbols

The symbols and signatures for the hazards are given in the Symbol-Sets (Fig. 27 and Fig. 28), where all the hazards included in the inventory list (Tab. 13) are represented according to their spatial properties. Both point and polygon signatures are offered for some hazards since the choice of representation may depend on the scale of the map.

b) Colours

As shown in Tab. 14, the colours representing the potential degree of harmfulness of the different hazards are assigned according to the resulting Hazard Index.

Tab. 14: Hazard Index and Hazard Index Classes

Hazard Index	Hazard Index Class	Hazard Level	Colour
0 - 24	1	no or very low	blue
> 24 - 48	2	low	green
> 48 - 72	3	moderate	yellow
> 72 - 96	4	high	orange
> 96 - 120	**5**	**very high**	**red**

Starting from blue, representing no or a very low hazard level, red is the obvious colour to depict a very high hazard level. In between the colours follow the colour sequence of the rainbow: green, yellow and orange. Any areas that have not been mapped should remain in white on the map.

In cases where overlapping symbols of different spatial properties occur on the map which have been assigned with the same colour, it is recommended that a white mask is set around either or both of the overlaying symbols to highlight these.

Fig. 29 and Fig. 30 provide an example of an unclassified and a classified hazard map respectively so as to illustrate the above recommendations, while Fig. 31 presents a graphic presentation of the proposed mapping procedure.

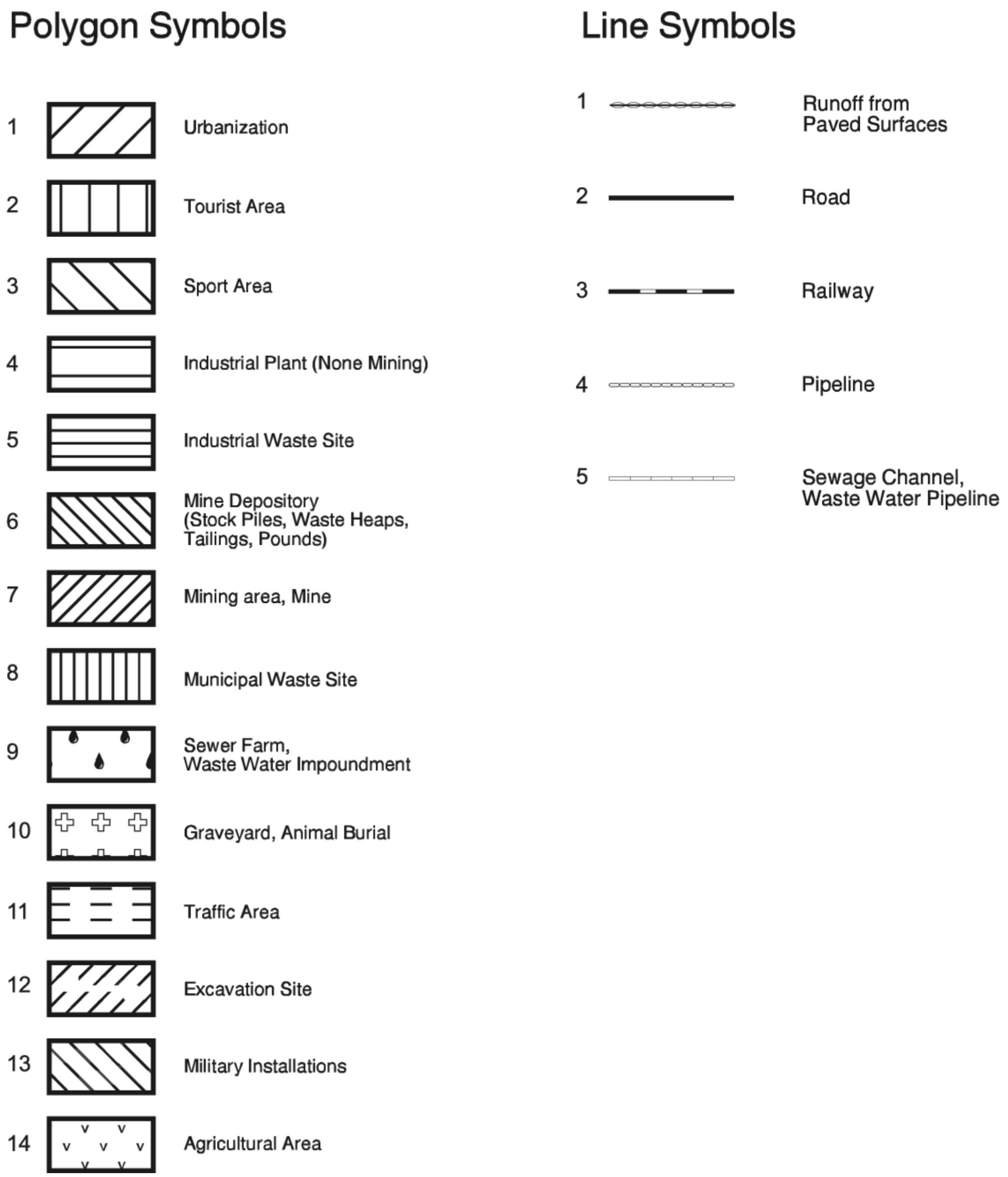

Fig. 27: Polygon and Line Symbols for Hazard Mapping

Fig. 28 (first part): Point Symbols for Hazard Mapping

Fig. 28 (second part): Point Symbols for Hazard Mapping

Fig. 29: Example hazard map - unclassified

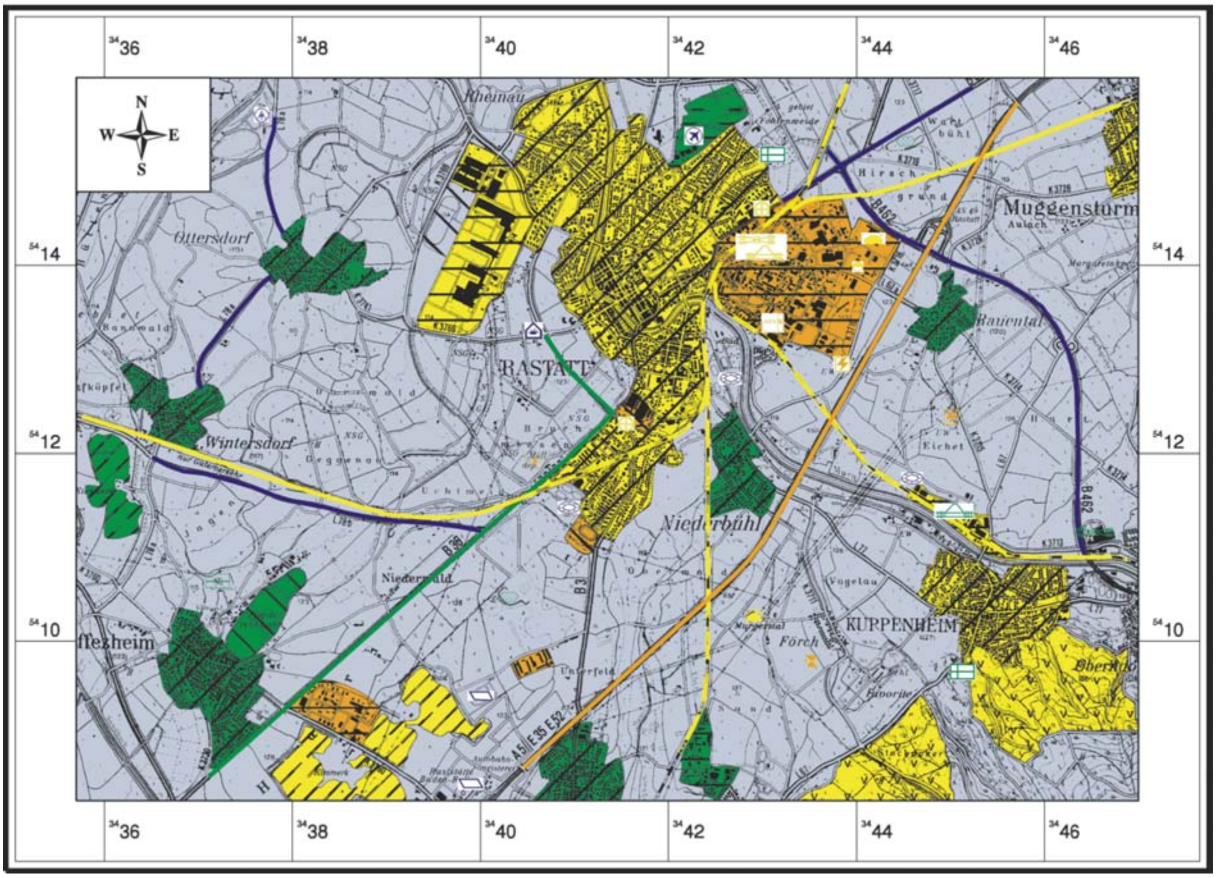

Fig. 30: Example hazard map - classified

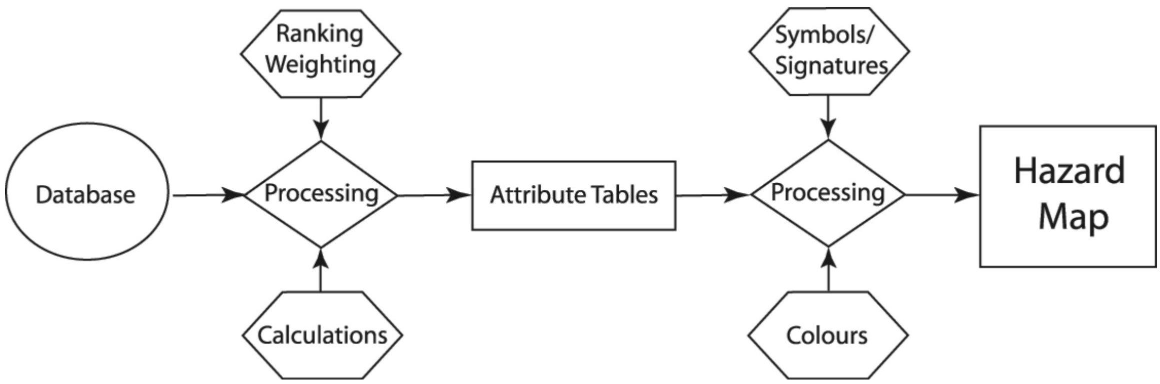

Fig. 31: Schematic illustration of the mapping procedure

5.3.5 Step 5: Mapping Techniques

The acquisition of the data required for hazard mapping as outlined in Step 2, may be obtained not only from ground surveys, but also from aerial surveys and satellite imagery. It is recommended that the integration of such data for the purpose of hazard mapping be performed using a Geographic-Information-System (GIS), while the eventual production of high quality output maps is then obtained from Computer Aided Cartography (CAC) software.

A review of currently available platforms and sensors and their usefulness for the purpose of hazard mapping is presented in Section 7.2. This is followed by an in-depth discussion on the use of GIS in Section 7.3.

5.3.6 Step 6: Data Evaluation

There is no unique interpretation of the term 'Data Evaluation', which can in fact be linked to several distinct tasks, including:

- Data quality
- Data integration (coupling of models and GIS)
- Output evaluation; and
- Data sensitivity

Clearly, each of these tasks should be performed not only in relation to the hazard maps, but also in respect of vulnerability and risks maps. Chapter 7.4 discusses the tasks that are associated with demonstrating the reliability of the maps obtained.

An exhaustive inventory of potential sources of error in groundwater vulnerability maps as drawn up by the National Research Council (1993) can be found in Chapter 7.3. It can be ascertained that these potential sources of error are equally relevant to the production of hazard maps.

It is evident that any map will only be as good as the data that were used for its construction. Assessing an overall 'margin of error' by establishing an estimate for each of the potential sources of error would not appear feasible or economically possible. Yet, the introduction of hazard attributes that will permit a rigorous data quality control and assurance is strongly recommended. In this regard, it can be especially useful to introduce attributes so as to permit a distinction between measured, statistical, extrapolated and estimated hazard data as well as attributes which will help appraise the spatial and temporal variability of each hazard.

5.3.7 Step 7: Production of Hazard Map

The final step in the Work plan (Fig. 24) concerns the actual production of the hazard map. A detailed description of the hazards maps prepared for several test sites in Cost 620 is presented in Part B. These include both 'unclassified', i.e. showing only the relevant geographical location information, and 'classified' (based on a hazard index computation) hazard maps. As could be anticipated, the test sites are not characterised by any substantial number or any large variety of hazards. Nonetheless, it is possible to draw some further conclusions on the results obtained.

It can be noted that several aspects are directly linked or influenced by the scale of the maps. To start with, the less detailed the scale, the higher the possibility that different hazards become associated with one location on the map. Wherever such overlap occurs on the map, it is recommended that the activity with the highest potential harmfulness to the groundwater (i.e. the highest hazard index) be shown. A further observation in relation to the scale effect may be noted from the discussion on the ranking factor (Q_n) for some of the test sites. Several approaches are presented which enabled the authors to differentiate degrees of harmfulness of hazards within a same category. Evidently, given the rather small surface areas associated with most of the test sites, the recommendation would be to establish ranking values on a regional scale first before applying these to the test sites.

It should also be borne in mind that Cost 620 purposely restricted the possible effect of the ranking factor (Q_n) in the computation of the hazard index (HI). If not, a 'larger' farm could arrive at a hazard index equal to that of a nuclear waste facility.

The same consideration applies to the reduction factor (R_f) for which it is recommended that the square root of the reduction values should be used in the computation of the hazard index (HI), and in all instances only small deviations from 1 should be used for the reduction factor.

Arguably, the test sites did not lend themselves to be the most appropriate case studies to demonstrate the Work plan to produce a Hazard Map as developed by Cost 620. Nonetheless, it is hoped that the presentations on the hazard maps prepared for the test sites have added further illustrations of the Cost 620 approach.

5.4 References

CIVITA M. (1994): Le carte della vulnerabilità all'inquinamento: teoria e pratica, Pitagora Editor

CIVITA M., DE MAIO M. (2000): Sintacs R5, Pitagora Editor

USEPA (1998): Guidelines for ecological risk assessment

ZAVATTI A., TACCONI P. (1999): Kind of hazard sources coming from human activity – 3rd Italian Meeting on the protection of groundwater, Parma October 1999

6 Risk Assessment

6.1 Risk Definition

Risk assessment and risk management techniques are increasingly used in a broad range of hydrogeological activities, including groundwater resource and quality management, or prioritisation of groundwater remediation (NEPC 2000, NRC 1993, U.S.EPA 1996). In particular, groundwater protection schemes are usually based on the concepts of contamination risk and risk management (ADAMS & FORSTER 1992, DALY & WARREN 1998). As outlined already at the beginning of Chapter 5, the definition of the term 'risk' is difficult, due to the different interpretations given to this term in the literature and even more so by its overuse in every day usage.

In the general understanding risk means the probability in a certain timeframe that an adverse outcome will occur in a person, a group of people, plants, animals or ecology of a specified area that is exposed to a particular dose or concentration of a hazardous agent. Various harmful substances, for example artificial chemical compounds, heavy metals, and radioactive substances, are spreading out in local and regional scale and polluting our environment. The risk associated with these harmful substances depends both on the level of toxicity and on the level of exposure. Their existence in the environment is a risk to all forms of life, and it is therefore necessary to evaluate the 'environmental' or 'ecological' risks posed by these substances. Typically, risk analysis focuses on natural resources, like air, water and soil, which are essential to all life forms.

In order to obtain a clear understanding of the specific application of risk assessment to groundwater, a preparatory working group (WG0) was established at the beginning of COST 620 with the task of providing unambiguous definitions for risk and related terms.

Risk: A term used to denote the probability of suffering harm from a hazard. With regard to groundwater, it refers to the possible contamination as a result from a hazardous event. It embodies both probability and consequences, defined as the likelihood or expected frequency of a specified adverse consequence on groundwater. Risk is not intended as an absolute measure but as a means of relative measure or comparison. This relative measure can be defined by the product of the probability of an event times the consequential damage.

Following the origin-pathway-target model (see Section 2.1.2) the risk of contamination of groundwater depends on three elements:

- the hazard posed by a potential polluting activity (equivalent to origin);
- the intrinsic vulnerability of groundwater to contamination (equivalent to pathway);
- the potential consequences of a contamination event (the target is the groundwater).

Risk assessment: An evaluation process for estimating the potential impact of a chemical, biological or physical agent on groundwater. The risk assessment or risk analysis identifies the existing or potential hazards and exposure pathways of contamination that need to be addressed in order to provide the basis for taking action to ensure groundwater protection. The assessment of groundwater has to take the following into account:

- the likelihood of an impact;
- the intensity of a potential impact with which it affects the groundwater (impact intensity);
- the sensitivity of groundwater with respect to the impact (groundwater sensitivity).

In connection with the third point, factors relevant to the environmental and economical value of the groundwater resource such as the current and likely future uses have to be included in the assessment procedure. Combining point one and two of the above list and investigating the third point separately, risk analysis may be operated in two stages:

- risk intensity assessment;
- risk sensitivity assessment.

Both are essential components to of a 'total' risk assessment, which forms the basis for sustainable risk management. These are highlighted by the questions shown in Tab. 15.

Tab. 15: Risk assessment: main questions of risk intensity assessment, risk sensitivity assessment and risk management

What can go wrong? *Hazard identification and identification of outcomes* How likely is it to go wrong? *Estimation of probability of these outcomes* How far can the hazardous impact reach the target? *Estimation of possible impact reduction (or estimation of the vulnerability)*	**risk intensity assessment**
What would happen to the target if it does go wrong? *Evaluation of the sensitivity of the target against the impact (consequence analysis)* Is the risk acceptable and can it be reduced? *Evaluation of the damage with regard to environmental and economic value*	**risk sensitivity assessment**
TOTAL RISK ASSESSMENT	
What decisions arise from risk assessment? What control measures are needed to minimise the risk?	**risk management**

Risk map: A method of summarising the result of total risk assessment with regard to the spatial distribution of risk. Given the multi-component assessment procedure, special maps may show only partial results of the whole procedure:

- Risk intensity map: shows the result of the risk intensity assessment. It involves the combination of the hazard map and the vulnerability map.
- Risk sensitivity map: gives the result of the risk sensitivity assessment. It depicts the sensitivity of the groundwater against a certain impact under consideration of the economical and ecological value of the resource.
- Risk map or total risk map or ecological risk map: represents all aspects of the assessment procedure.

Risk management: is based on the analysis of land use and subsurface contaminant load as well as of the aquifer vulnerability and is followed by a response to the risk. This response includes the assessment and selection of options and the implementation of measures to prevent or minimise the probability of a contamination event and its consequences, should it occur. This includes, for instance, land-use practices directing developments towards lower risk areas, suitable building codes that take account of the vulnerability and value of the groundwater, installation of monitoring networks and special operational practices.

6.2 Assessment concept

6.2.1 General approach

The conventional origin-pathway-target model for environmental management is used for expounding the concept of groundwater risk assessment. This model is already explained in detail in Chapter 2.1.2 because it is also used as the basis for both vulnerability and hazard mapping. For the risk assessment, an operational cost efficiency analysis of the expected damages supplements the conventional risk evaluation, which is based just on the concentration of harmful substances in the target (SCHOLLES 1997, COLOMBO 1992, LEESON et al. 2002).

In a two tiered approach, firstly the intensity of a hazardous event on the specific target (impact intensity) has to be estimated by considering all possible impairing factors in connection with the hazard and along the pathway to the target (risk intensity assessment). In a separate, second step the consequences of such an impact on the target with respect to its sensitivity to changes coming from outside (groundwater sensitivity) needs to be evaluated by taking into consideration its ecological and economic value (risk sensitivity assessment). Finally, both steps may then be combined according to a certain mathematical procedure (FEDRA & WEIGKRICHT 1995, KOVAR & NACHTNEBEL 1993) to complete the risk assessment (Fig. 32). In order to distinguish this last step from the two intermediate steps, it could be termed total risk assessment or ecological risk assessment (SCHOLLES 1997). Although in many risk analyses only step 1, that is the risk intensity assessment, is carried out, it should be clear that for further application, and particularly with regard to risk management, a complete risk assessment including risk sensitivity assessment is necessary.

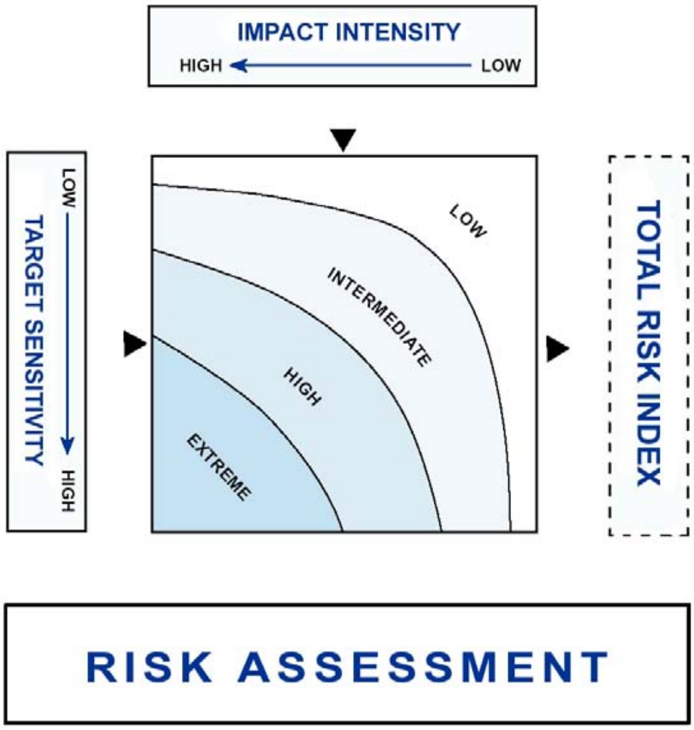

Fig. 32: Risk assessment scheme showing the two main steps of the analysis (risk intensity assessment and risk sensitivity assessment) leading to a quantitative or semi quantitative approach (total risk index).

Attention may be drawn also to another aspect of risk assessment. Some of the influencing or controlling factors are static or relatively static, others are very dynamic and sometimes just represent a moment in time. This makes the risk assessment much more complex. For long term risk assessment in particular, we normally refer to a static condition by averaging dynamic processes over time. However dynamic processes can also be included, but these then require time related functions for the evaluation of the relevant dynamic processes (FEDRA 1998 AND 2002, MAGIERA 2002). In more complex multidynamic processes the inclusion of numerical models in risk analysis may help to satisfy the time depending conditions of different processes (KOVAR & NACHTNEBEL 1993).

6.2.2 Risk intensity assessment

Risk intensity assessment for groundwater has a twofold task (FORSTER & HIRATA 1988, LOBO-FERREIRA 2000, NEPC 2000, DALY & DREW 2000):

1) dealing with the possible origin of contamination and the likelihood that a release of the contaminant into the environment occurs (described as hazard assessment in Chapter 5);
2) estimating all the processes which can lead to a reduction of the contamination before it reaches the groundwater surface or the special part of groundwater defined as the target (this is consistent with the definition of vulnerability as described in Chapter 2 and 3).

The origin of contamination could be any hazard in the relevant catchment, the combination of hazards represent the potential contamination loading (COLOMBO 1992, DALY & MISSTEAR 2002, ROSEN & LEGRAND 1997). Determining the degree of harmfulness will require a number of factors to be considered as described in Chapter 5. This will be based initially on a review of all the relevant site history including historical maps and plans, planning information, records of incoming and outgoing chemicals (to estimate possible contaminant loss), details of site processes and the knowledge and experience of workers or local residents. Detailed information about the anthropogenic activities, including knowledge on the state of the industrial plant, its maintenance and the processes and history of incidents on a site is important as it leads to an understanding of the likelihood of release, source geometry, release mechanism and likely contaminants so that the probability of an hazardous event can be estimated. Site investigation data are essential for understanding the degree of harmfulness as they give an indication of the distribution (laterally and vertically) and the concentrations of the contaminants over the period of monitoring.

For the possible reduction of the contaminants on the way from the source to the potential receptor, the pathways or routes have to be defined (FORSTER 1987, MAGIERA 2002, U.S.EPA 1993). It seems appropriate to start with the development of a conceptual hydrogeological model of sufficient accuracy to describe the migration or flow of contaminants through the underground to a potential target (receptor). The soil zone, the unsaturated zone and the saturated zones, and the entire key flow and transport processes therein need to be considered. Pathways along which flow normally occurs should be examined, but potential bypass mechanisms must also be taken into account. This requires that information on soil, geology, hydrology and hydrogeology have to be collected and assessed. The goal is to come up with a quantitative or at least semi quantitative estimation, the *risk intensity index*, which is a comparable coefficient describing the portion (or concentration) of contaminants reaching the target.

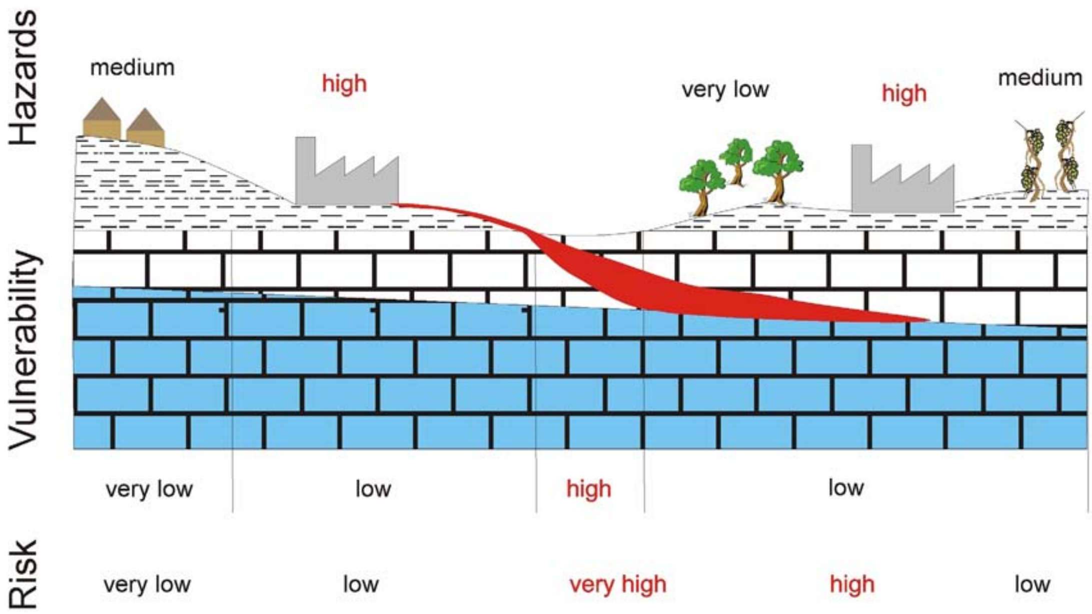

Fig. 33: Risk assessment of groundwater considering only the superimposed effects of hazards and vulnerability according the procedure of risk intensity assessment.

Fig. 33 illustrates the various possible interactions of the hazard and vulnerability distribution on risk intensity assessment. The risk intensity index for groundwater contamination depends not only on vulnerability but also on the existence of significant pollutant loading entering the subsurface environment. It is possible to have high aquifer vulnerability but low risk index, if there is no significant pollutant loading; and conversely to have a high risk index value in spite of low vulnerability, if the pollutant loading is exceptional or if there is the possibility of bypassing in less vulnerable areas.

6.2.3 Risk sensitivity assessment

For COST 620, the potential target of the contaminant is the karst groundwater or in general the groundwater, which represents the endangered receptor. But precisely what do we mean by groundwater? By analogy to the same issue arising when discussing vulnerability we have to ask what is the reference location within the groundwater system that is of interest. This leads back to the interpretation of *targets* at risk, where different possibilities could be differentiated, including for example the whole groundwater body of an aquifer, part of it, wells or springs, or groundwater at the discharge point. In the vulnerability concept (cf. Chapter 2) a distinction is made between source vulnerability (e.g. the water of a spring or a well capture) and the resource vulnerability (the whole water body of an aquifer).

Risk is defined as the probability of a hazardous impact multiplied by the consequential damage. The impact is attributed to a particular hazard. In case of point like hazards (e.g. a leaking septic tank) the possible contamination will normally only affect a part of the groundwater body. The contamination of the groundwater will start where the contamination reaches the saturated zone, and will be distributed further downstream according the flow direction and a certain hydrodynamic dispersion effect. The consequential damage, which by definition has to be included in the risk assessment, depends however on the extension of the plume and the distribution of the contaminants in the plume. In the case of low attenuation or negligible degradation, the plume will reach the discharge point (spring) or discharging area with a relatively high concentration. If on the other hand, there is a strong attenuation in the aquifer, it will only be possible to follow the plume over a limited distance before the concentration values drop below certain levels or even disappear due to detection limits. Therefore the target at risk generally consists of only a part of the whole groundwater body, and within a contaminated plume, the risk assessment varies with the concentration levels.

The above examples demonstrate that flow condition (flow quantity, flow velocity, flow direction) is the most important factor determining the sensitivity of groundwater to a contamination impact (U.S. EPA 1993, MORRIS & FOSTER 2000). The sensitivity is influenced more or less by all factors, which might minimize the harmfulness of the contaminants with regard to the target. The most important sensitivity parameter is the attenuation ratio, which depends on the available amount of groundwater and the flow velocity. Other sensitivity factors could include biodegradation or the chemical milieu of the water, which might support precipitation of the supplied contaminants. With regard to the extension of the plume, the flow direction is decisive. Both flow velocity and flow direction underline therewith the functional dependency of the sensitivity from space and time.

Similarly to the discussion on risk intensity assessment, the risk sensitivity assessment may also deal with dynamic processes. However, including such processes in the evaluation will complicate the assessment procedure. Nevertheless, approaches can be achieved by using non-steady state flow or transport models (FEDRA 2002, TIM ET AL. 1996, SCHOLLES 1997, U.S.EPA 1996). With regard to the long-term prediction of the risk, it seems acceptable to use averaged steady state conditions as first approaches.

Risk sensitivity assessment does not only enquire about the effects of the impact on the target, but also has to analyse the resulting damage: e.g. what reduction in value of the target is caused by the impact? For groundwater we have to consider both the ecological as well as the economical value (SCHOLLES 1997, U.S.EPA 1996, ZAKHAROVA 2001). This includes firstly the damage to the ecosystems involved and the accompanying utility loss in terms of anthropogenic use value and secondly the remediation cost for the removal of the contaminants. The literature provides ample approaches on how to calculate the total cost for such damage from an economical point of view. However, due to the uncertainties that are typically associated with these types of calculations, especially with regard to the ecological damage, groundwater risk assessment frequently just refers to a certain classification of the groundwater body, according to a more or less qualitative subdivision on the basis of recent or future utilisation of that groundwater (SCHOLLES 1997).

Examples for groundwater use and value determination can be also found in the literature and in some regulatory guidelines. For example, the Australian Water Quality Guidelines (NEPC 2000) define the environmental value as the value or use of the environment which is conducive to public benefit, welfare, safety or health and which requires protection from the effects of pollution. Six environmental values are presented: aquatic ecosystems; aquaculture and human consumers of food; agricultural water; recreation and aesthetics; drinking water; and industrial water. For each environmental value, a set of guideline criteria is presented.

The U.S. EPA (1993) has proposed three classes:

- class I : groundwater of high value, which are irreplaceable sources of drinking water and/or are ecologically vital,
- class II: groundwater which is defined as current and potential sources of drinking water and water having other beneficial uses,
- class III: groundwater that is unsuitable for human consumption because of salinity or widespread contamination from multiple sources.

For a more detailed value determination, consideration of the following eight factors is required (U.S.EPA 1996): quality, quantity, current public water supply systems, current private drinking water supply wells, likelihood of future drinking water use, other current or reasonable expected groundwater use in review area, ecological value, and public opinion of use and value of groundwater.

Overall, the purpose of the use and value determination is to identify whether the groundwater in the risk assessment should be considered as high, medium or low value groundwater.

6.2.4 Aggregation of risk components

Groundwater risk depends on many influencing factors, each of which can be considered to be responsible for a certain element of the risk. It follows that the final aim of a risk analysis is to come up with a total risk assessment, and hence enabling decision-makers to reach a decision e.g. on the type of protection measures that would be most appropriate. On the other hand, for a more detailed interpretation by experts, it might be useful to give not only the values of the total risk assessment, but to show also the results of the intermediate steps. As mentioned in section 6.2.1, the multiple factors influencing the risk can be subdivided between those, which contribute mainly to the intensity with which a certain impact approaches the groundwater, and others, which govern the sensitivity, i.e. how groundwater reacts to such an impact. To characterise this relationship, three risk indexes are recommended (Fig. 34):

- Risk intensity index (RII): provides a relative measure for the intensity of the hazardous impact resulting from the likelihood of the hazard and the vulnerability of the pathway.
- Risk sensitivity index (RSI): provides a relative measure for the sensitivity with which the groundwater reacts to the impact and the resulting damage expressed in terms of ecological and economical values.
- Total risk index (TRI): summarises the effects of all influencing factors analysed in the risk assessment procedure.

The aggregation of the different risk components caused by the multiple factors is one of the most difficult problems and needs special strategies (SCHOLLES 1997, FOSTER 1987, FOSTER & HIRATA 1988, DALY & MISSTEAR 2002). With regard to the analysis and prognosis of the ecological or economical effects there exists great uncertainty. The aggregation is made difficult partly because of the completely different dimensions or measure scales, and also because of their different weights in a total risk assessment. In general, certain weighting factors for each influencing component are used, which are multiplied by the evaluated or estimated intensities of the components. Examples and recommendations are presented in Chapter 3, 4, and 5 for the relevant factors and indicators, which determine the risk intensity caused by hazards and vulnerability.

The problem of aggregation still remains for the total risk assessment, where the results of risk intensity assessment and risk sensitivity assessment have to be merged. As in the previous investigation steps, separate assessment of the two main components, their classification and finally their aggregation have to be performed in a given sequence. A rather simple system is recommended for the classification, for instance in just three classes "low" – "medium" – "high" or, possibly extended to five classes by adding "very low" and "very high". It does not seem advisable to extend classification to more than ten classes.

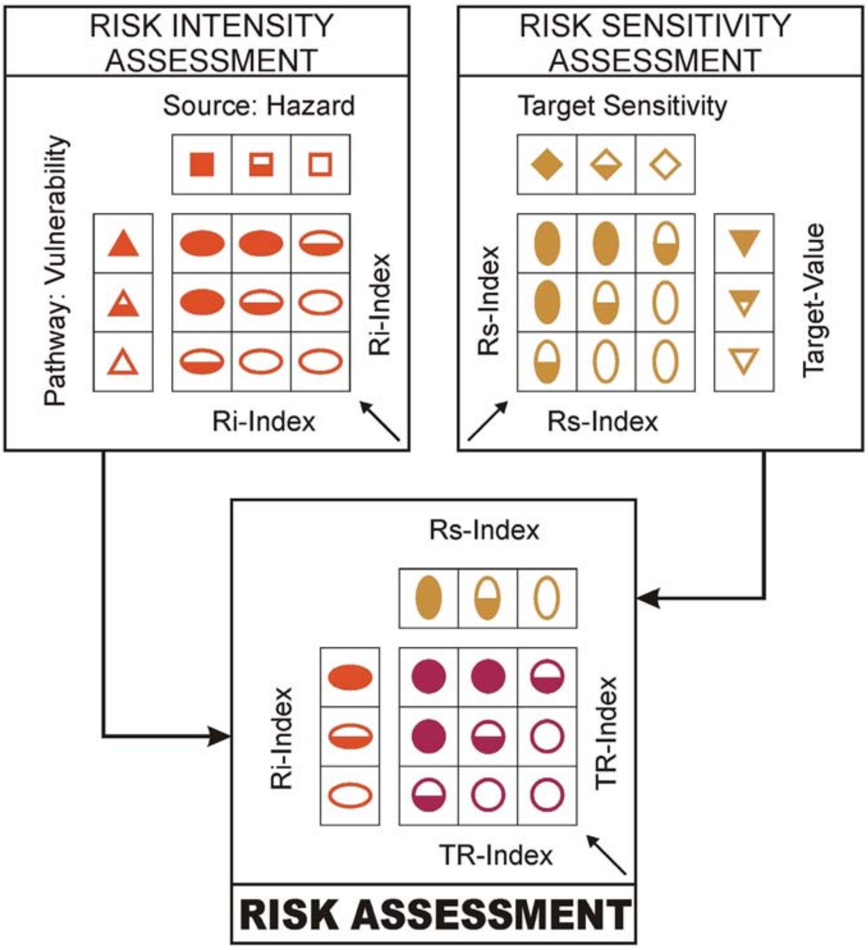

Fig. 34: Schematic procedure of risk aggregation by simple matrix technique (SCHOLLES 1997). The figure shows the link between the results of risk intensity assessment and risk sensitivity assessment leading to a total risk assessment with their respective characteristic indices.

6.3 Risk Mapping

6.3.1 General Remarks and Mapping Techniques

Thematic maps are important tools to display spatial geo-referenced data as they are used in risk assessment. Risk assessment is information intensive, yet one of the major aims is to disseminate this information to decision makers, experts, water agencies or companies, potentially affected persons, etc. Therefore, there are major advantages in using thematic maps to present the information in a concise and clear manner.

As demonstrated in Chapter 5, the scale and resolution affects also present important issues for risk mapping. The selection of a suitable scale is most important and depends on the objectives, for which the map shall be used. Local hydrogeological and hence risk conditions may cause variations in local groundwater assessment which are not evident at the regional or national scale. In many instances, small hazardous or special geological features that are not mapped on a regional scale can pose great potential risk. The recommended map scales are as follows:

- 1:100.000 to 1:300.000: larger regions or countries
- 1:25.000 to 1:100.000: regulatory activities and general groundwater management tasks

- 1:5.000 and 1:10.000: specific groundwater management problems, like delineation of groundwater protection zones, remediation works or emergency planning

Modern information technology provides very efficient tools to support mapping techniques (FEDRA 2002, FEDRA &WEIGKRICHT, KOVAR & NACHTNEBEL 1993, TIM ET AL. 1996). Geographic information systems (GIS, cf. Chapter 7) are one of the most powerful tools for spatial analysis, but their capabilities for complex and dynamic analysis are limited. Early GIS applications are often based on static geo-referenced data, and simple overlay and buffer analysis. Simulation models, on the other hand, are useful tools for complex and dynamic analysis, but however often lack the spatial analysis function that GIS is offering. Therefore the integration of GIS and simulation models in the case of dynamic factors should support more powerful and easy to use risk information systems.

Fig. 35: System of risk maps showing the hierarchical structure of the different type of maps.

The three main types of risk maps as defined in Section 6.1 are presented in Fig. 35. The complete risk assessment map shows the distribution of risk classes defined by the range of the TRI (total risk index). Frequently, this map will be the only printed risk map, which will made available to decision makers, land use planers, water utilities and to the interested public. The supporting maps in the logic sequence of the assessment procedure are the risk intensity map (RII) (together with its two basic maps, the hazard and the vulnerability map) and the risk sensitivity map (RSI). All these maps may be accompanied by single factor maps, such as geology, thickness of unsaturated zone, groundwater flow direction, time of breakthrough and others. These additional maps are mainly produced for all those involved directly in groundwater management tasks.

6.3.2 Mapping procedure

Risk assessment requires collection, processing and analysis of large volumes of environmental and technical information. Fortunately modern information technology provides tools to support these activities.

<u>Risk intensity maps for groundwater resources</u>

MORRIS & FOSTER (2000) defined groundwater pollution risk "as the probability that groundwater in the aquifer will become contaminated to an unacceptable level by activities on the immediately overlying land-surface". This approach uses the interaction between the subsurface contaminant load and the aquifer pollution vulnerability at the location concerned, described in Section 6.1 as “risk intensity assessment”. The approach is frequently used, as it is a rather simple way to assess the risk intensity (Fig. 36). It is also used in the work of COST 620 for the test sites in Engen and Sierra de Líbar (see Part B). These maps show the risk of groundwater pollution of each hazard in relation to resource protection. The decisive risk index is the probability that contaminants with a certain amount and concentration (intensity index) reach the surface of the groundwater. If it refers to one specific contaminant or one group of contaminants, the map may be called ‘specific risk intensity map’ or in the case of a selected contaminant such as nitrate, it is called ‘nitrate risk intensity map’ (Fig. 36). The groundwater and the aquifer characteristics are not included in this type of risk assessment.

The calculation of the risk intensity index may consider the hazard and vulnerability indices, according the matrix concept shown in Fig. 34 (FOSTER 1987, MAGIERA 2002, DALY & MISSTEAR 2002). Another possibility is to combine the effects of the intrinsic vulnerability and the hazard by using a mathematical approach. The advantage of this procedure is that smooth values (not classes) can be used, resulting in infinitely variable risk values. However, this second approach suggests a very high precision can be achieved, while it is unlikely that the information available in the database would permit such a precision. In the test site studies of Engen and Sierra de Líbar the risk intensity values were determined with a simple equation:

$$RII = 1/HI \cdot \pi \qquad (1)$$

RII = risk intensity index
HI = Hazard Index
π = PI-factor (index for intrinsic vulnerability).

As the values of the hazard index and the PI-factor are developing in the opposite direction with higher risk, the reciprocal hazard index was used to ensure non-ambiguous risk values (Fig. 37). The classes were built considering the maximum and the minimum risk values as well as the favoured number of classes. This approach favours the assignment of assessed groundwater to higher risk classes in contrast to the matrix system (cp. Fig. 34). It implicates very high risk everywhere where vulnerability is high even when the hazards are very low and also high risk everywhere where hazards are high and vulnerability is low. The indices can be rearranged so that low risk classes are overemphasised compared with high risk assignment. This means that the way, how the parameters are arranged in the diagram of Fig. 37 gives certain scope to subjective preferences.

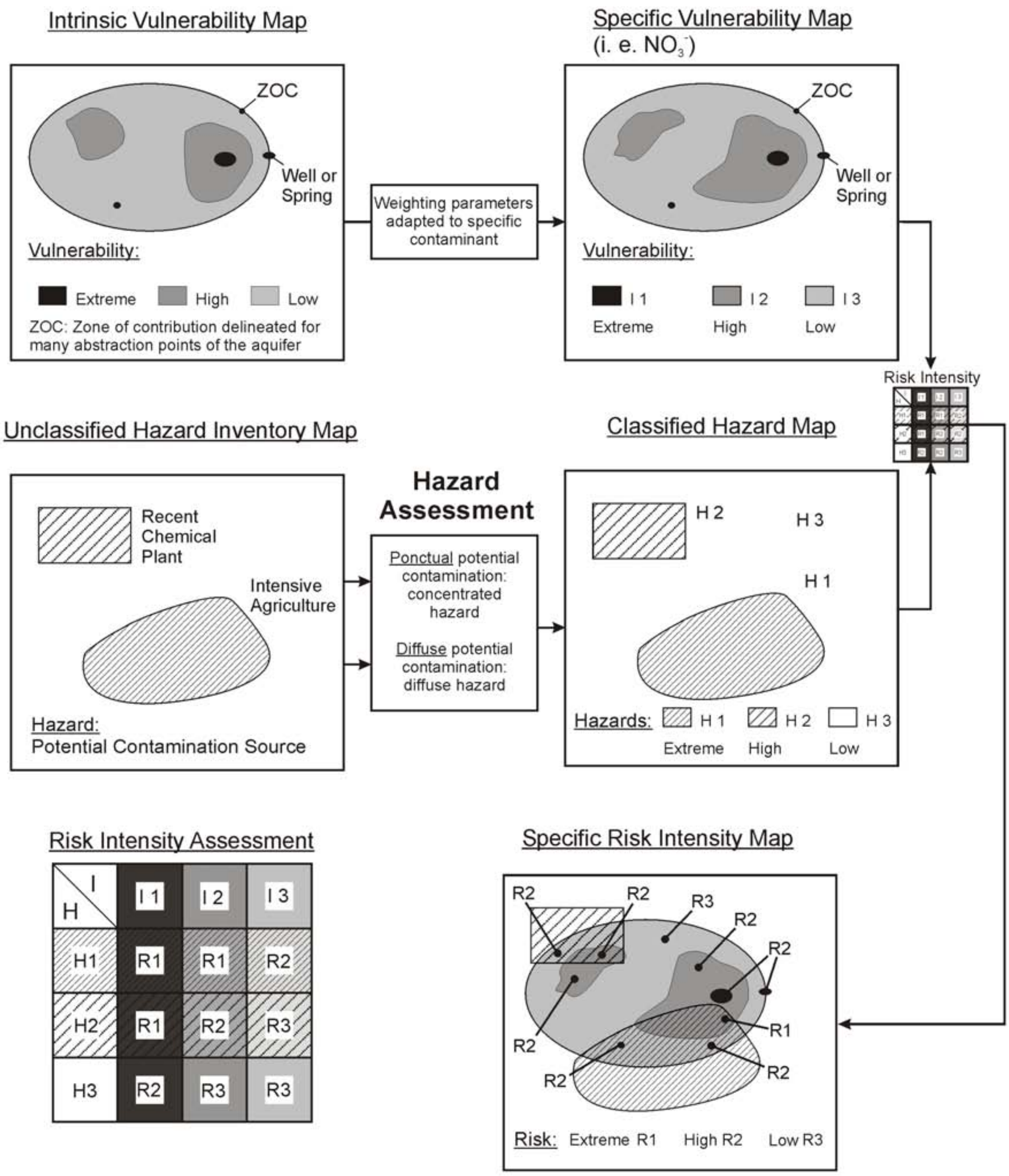

Fig. 36: Step by step approach to develop a specific risk intensity map for groundwater resources (after an unpublished draft from Cyril Delporte).

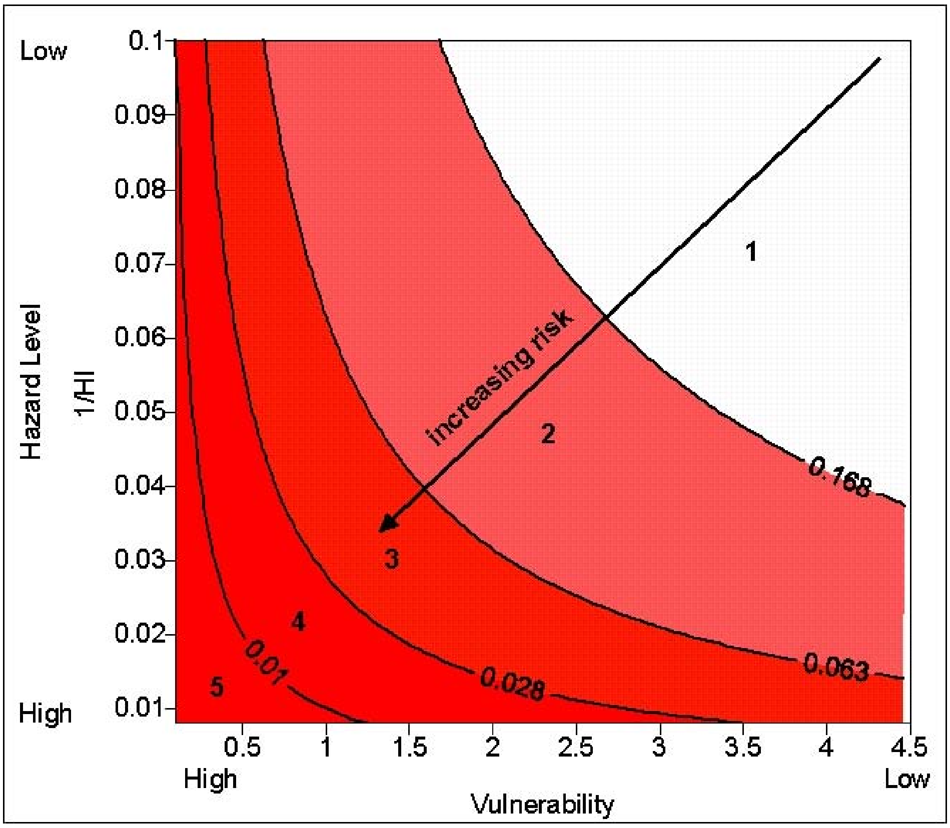

Fig. 37: Diagram of risk intensity index calculated after Equation (1). Five different classes were assigned to build risk classes.

Another example of a similar system for drawing the risk intensity map was developed by Peter Malik and Jaromir Svasta from the Slovakian Group in COST 620 (Fig. 38). The proposal makes easier the aggregation of the two base maps, the hazard and the vulnerability map (Fig. 38, a and c). To this end the point and linear hazards are transferred in aerial hazards by creating buffers around their original forms (Fig. 38, b). In case of additional overlapping of two adjacent hazards by this procedure, the hazard index is summed up for areas of intersection to a maximum value of 120 scores. For the aggregation itself the two different point counting systems are transformed in relative scales (0 – 100 %). The final risk intensity numbers for each point are achieved by simple multiplication of the two relative values (Fig. 38, d). Due to the fact that the resulting risk values can vary over several magnitudes a logarithmic scale is proposed for the risk intensity classification.

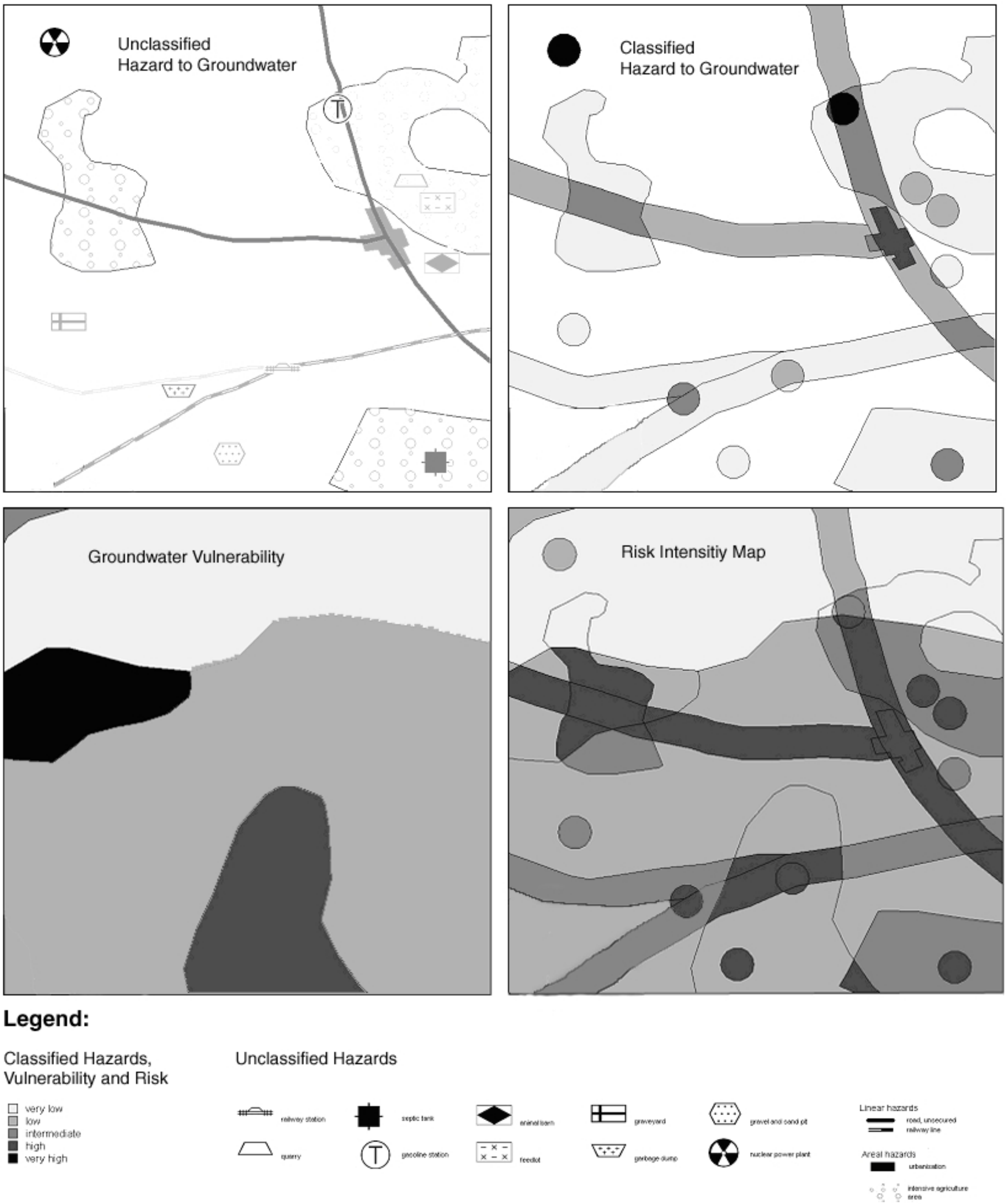

Fig. 38: Example of the construction of a risk intensity map by integration of a hazard and vulnerability map.

<u>Risk intensity maps for specific targeted groundwater</u>

In chapter 6.2.3, it was already mentioned that, particularly in the case of point like hazards, the possible contamination frequently affects only a part of the groundwater body. The contamination of the groundwater will start where the contamination reaches the saturated zone, and will be distributed further downstream according to the flow direction and a certain dispersion effect. The consequential damage in this instance should refer only to the contaminated part and not to the whole resource.

Therefore it is recommended that the risk assessment be restricted only to those parts of the groundwater, which are actually endangered by a possible impact. For preparing such maps, knowledge of flow conditions are necessary. For a detailed study of the possible spreading process of the contaminant, the use of a transport simulation model offers an optimum solution. However this presupposes a very detailed knowledge of the hydraulic conditions and parameters. Knowing the groundwater flow direction and assuming stable input conditions, the extension of the plume might be estimated in a more usual way by assumption of flow velocity and flow dispersion (BURGER & SCHAFSMEISTER 2000). A schematic example of such a risk map, shown in Fig. 39, still follows the definition given to a risk intensity map, because it doesn't consider primary attenuation by mixing of the contaminant with the groundwater body nor includes any valuation of the harm caused to the groundwater. The assessment of the latter conditions would lead to an intrinsic sensitivity of groundwater and finally to a total risk assessment.

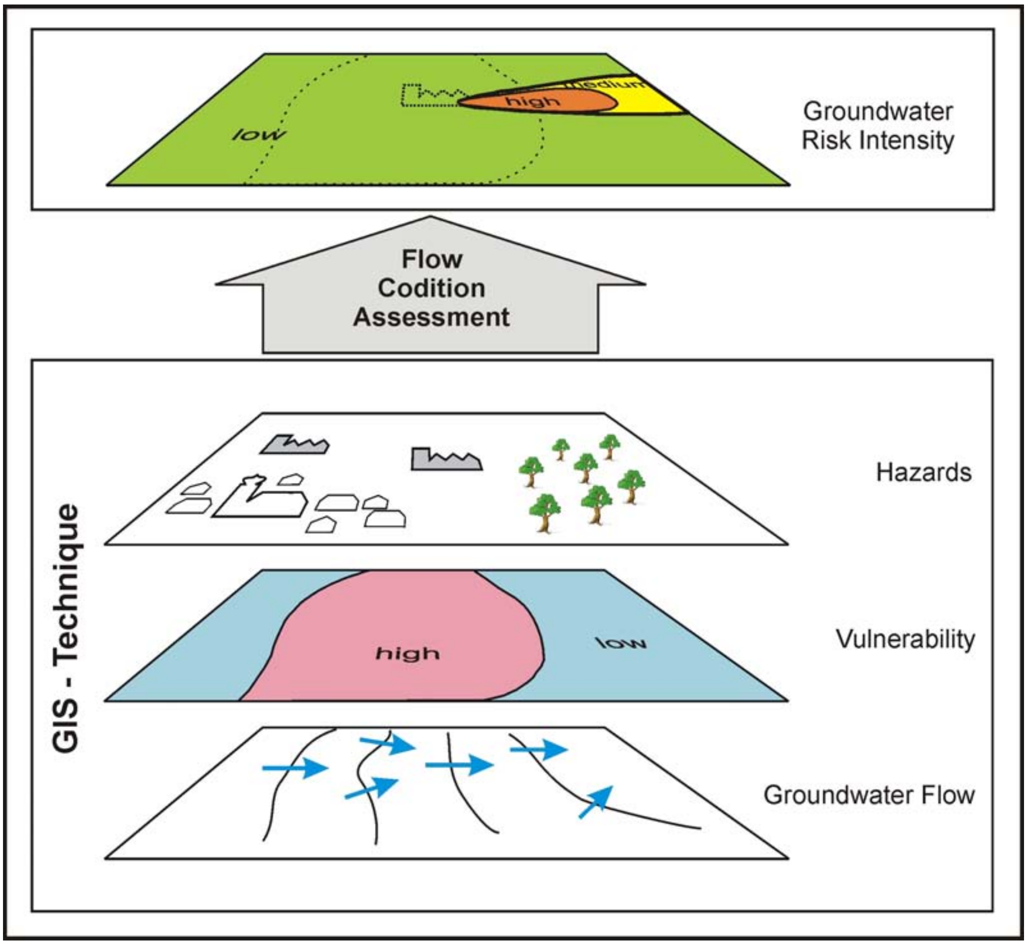

Fig. 39: Construction of a risk intensity map considering the lateral spreading of the contaminant plume.

Total risk map

In Section 6.2.4 an overview of the main three types of risk maps was given. COST 620 was dealing with groundwater vulnerability and hazards causing impacts on groundwater. These two components lead to the evaluation of the risk intensity. Examples of risk intensity maps are illustrated in part B of this volume.

Total risk assessment needs the inclusion of the groundwater risk sensitivity under aspects of ecological and economical valuation. This was not the topic of COST 620, therefore no examples for total risk assessment can be given from work of this COST action. It should be emphasized that this does not mean an ignoring or a certain unimportance of total risk assessment, quite the reverse valuation of groundwater and its inclusion in risk assessment is essential for a sustainable groundwater management. Risk evaluation of groundwater should therefore not stop with risk intensity, but should include a total risk assessment.

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7 From Data Collection To Map Validation

7.1 Overview

As stated in Chapter 1, the intention of COST 620 was to develop an approach to vulnerability mapping that would be sufficiently flexible to be able to be adapted for use in all types of karstic environments. The development of meaningful intrinsic or specific vulnerability maps is reliant on the availability of appropriate data. The data needs for vulnerability map development are extensive; as a result data from numerous sources, differing resolutions and often-uncertain quality are used. Such diverse data sources of uncertain quality can impact on the reliability of the final map, this places a strong requirement for inclusion of appropriate validation methods within the mapping process.

When vulnerability maps are being produced, it is essential to be able to estimate their reliability – the degree to which the map corresponds to the actual degree of protection/vulnerability afforded to groundwater in the area. Validation of vulnerability maps is always difficult but particularly so as they relate to karstic areas in which there are marked three dimensional variations (heterogeneities) in hydrogeological attributes and in which the relative intensity of hydrogeological processes are high.

The principles of and differences between the validation and verification processes, as they relate to the production of vulnerability maps, are outlined in this chapter, as are the key map "validation" tools.

It is not the intension of this chapter to discuss routinely employed basic data collection or evaluation techniques, as these are considered to be the "everyday tools" of the hydrogeologist. However, a section is included that summarises the collection of remote sensing data and its potential use in the production of both hazard and vulnerability maps.

The collection, collation and "interpretation" of the data necessary in the production of a vulnerability map requires significant effort for anything but the smallest geographical areas. As a result this chapter includes an overview on the use of Geographical Information Systems (GIS) processing techniques in the production of vulnerability maps.

7.2 Data collection using remote sensing

In the past few years, remote sensing techniques have been gaining acceptance as a useful tool to employ in various political decision-making processes such as; EU agricultural subsidies, coastal pollution monitoring and flood and forest fire surveillance. With respect to the preparation of landuse maps, it is particularly worth noting the CLUSTERS (Classification for Land Use Statistics: Eurostat Remote Sensing programme) nomenclature developed by EUROSTAT in the framework of its Remote Sensing and Statistics Programme.

However, with no dedicated hydrological satellites in orbit, groundwater analyses and particularly hazard, vulnerability, and risk assessments do not as yet form part of this process. Several studies have been undertaken to identify observational requirements in the field of Hydrology and Water Management, and the degree to which these can be addressed by the existing remote sensing platform/sensor combinations (Barrett, 1996). The very recent improvement in the resolution of satellite images has broadened the application possibilities of groundwater relevant information, which can be derived through various systems and methods, which are continuously being developed. The use of satellite images for more accurate hazard mapping has recently also become possible through data fusion with aerial photos and digital terrain models (DTMs), and the integration of cartographic features with GIS data.

Platforms and sensors to use for hazard mapping - Choosing the appropriate remote sensing system (i.e. platform and sensor), depends greatly on the type of hazards involved. Hence, determining the critical areas and evaluating their problems should be the first step before deciding which remote sensing system would be applicable. A remote sensing system uses the electromagnetic spectrum in the form of the different sensors found on the various orbiting platforms. Electromagnetic radiation occurs as a continuum of wavelengths and frequencies, i.e. from short wavelength, high frequency cosmic waves to long wavelength, low frequency radio waves. Normally, the sensors are multispectral or multiband systems, which means that they record in different bands several images of the same scene. The wavelengths that are of greatest interest in remote sensing are visible and near infrared, thermal infrared and microwave radiation. For the simple exercise of mapping an area, obtaining a satellite image with only the visible (panchromatic and multi-spectral) part of the spectrum enables the "accurate" drawing of the area under study. The level of this accuracy is obviously dependent on the resolution of the image. However, it does not become cost effective to utilize remote sensing techniques for the sole purpose of drawing up a hazard map if the visible or optical image is not purchased together with the near infrared and thermal bands to make the optimum utilization of these techniques.

Using the visible bands will allow the drawing up of the exact boundaries of the landuse under study. The locations and dimensions of human settlements, the size and type of industries and the specific land zoning can depict were for example, the use of agricultural fertilizers can contaminate the groundwater. On the other hand thermal infrared scanning can provide even further information by for example detecting septic tank seepage into the groundwater. The near and thermal infrared bands can also show clearly the aerial distribution of soil moisture, which can be a factor indicative of a near surface water table (Nossin, 1989).

Image processing - Satellite images are recorded in digital units that have to be processed quantitatively to be able to be used. Image processing can be used to correct, enhance and classify all types of continuous imagery. Such processing may include selective enlargement to facilitate the observation of detail within a scene or to isolate a particular study area or feature; contrast modification as a means of optimising the range of image grey tones; the enhancement of tonal edges for the mapping of geological faults, breaks of terrain slope and textural boundaries (Curran, 1985).

Image classification is commonly used for feature or pattern recognition and usually involves converting a continuous tone image into a thematic map. As with most image processing tools, the suitability of image classification for a particular interpretation task will depend on the nature of the application and the resolution of features to be identified in the image. Land cover mapping exercises are frequently based on image classification since all the available bands in an image can help to discriminate between different cover types. The relationship between selected channels can also be used to define different feature categories before or after classification. Remote sensing image interpretation is based on spectral classification, which relies on the observation that different land covers reflect or emit electromagnetic energy in different and often unique ways, while similar land covers have similar spectral signatures (Harrison and Jupp, 1990).

The Landsat Series - Following more than 16 years of continuous acquisitions of multispectral data, the major part of the earth has been imaged by Landsat more than 350 times. The Landsats 1-3 carried the Return Beam Vidicon (RBV) camera and the multispectral Scanner (MSS). The second generation of Landsat satellites, beginning in 1982 with Landsat

4, carries a Thematic Mapper (TM) in addition to the MSS. The most in the series, Landsat 7 is equipped with an enhanced Thematic Mapper. It covers 6 bands aboard of multi-spectral data ranging from the visible to the infrared portion of the spectrum. Landsat has a geometric resolution of 30 metres (i.e. each pixel on the image represents 30x30 metres), which normally provides sufficient level of detail for a vast range of applications, and yet maintains a sufficiently large scene size to give the "big picture" if mapping a large area. The 15 metre panchromatic band is co-registered to the multispectral data, which is useful in its own right but can be used to add crispness and detail to the multispectral data. The thermal band is available at a 60 meter resolution. The repeat cycle (the satellite visits the same spot) for Landsats 1-3 was 18 days, for Landsat 4, 5 & 7 it is 16 days. Landsat 7 was launched on April 15, 1999, and orbits 8 days behind Landsat 5. A single scene may cost from €450.

The continuity of the Landsat images is still very useful, especially when it is necessary to study the evolution of a hazard on the ground and it is a perfect tool to draw up maps of areas for which a 30 metres resolution proves sufficient.

Other Satellites - Apart from the American Landsat series, the French SPOT series (with stereoscopic possibilities), and the radar satellites (able to record images through clouds and during the night), such as the European Space Agency's ERS series and the Canadian Radarsat, have largely dominated the remote sensing field, i.e. until the arrival of a new generation of satellites with much better resolution than before. The better the resolution the more accurate the mapping but also the more expensive. First among this new generation was the IKONOS satellite with a 1-meter resolution and most recently the Quickbird satellite which promises to replace even the use of aerial photos (aeroplanes can fly much lower than the 900 km high orbiting satellites and can have a resolution as high as 0.2m to 0.3m). Indeed the successful launch of QuickBird and its high-resolution sensors has narrowed the gap between satellite images and aerial photos. In the near future it could even replace aerial photos for some applications, depending on resolution and accuracy requirements.

The QuickBird satellite, lifted into orbit on October 18, 2001 has the best resolution for a commercial satellite: 61-72 cm in the panchromatic and 2.44 - 2.88 m in its multispectral sensors, depending upon the off-nadir viewing angle (0°-25°). In addition, it also has along-track and/or across-track stereo capability, which provides a high revisit frequency of from 1 to 3.5 days, depending on the latitude. The sensor has coverage of from 16.5 km to 19 km in the across-track direction, (a Landsat track is 183 km wide). It has a resolution, which is 60 to 90 percent more detailed than any other commercial, high-resolution sensors. The QuickBird's Basic Image product is delivered with 16.5 km by 16.5 km for a single area, and with 16.5 km by 165 km for a strip. It enables the user to map large areas faster with fewer images, and less ground data to manage and process.

QuickBird's high-resolution, high-revisit frequency, large area coverage, and the ability to take images over any area, especially inaccessible areas are certainly the major advantages over the use of other satellites and even aerial photos. A Quickbird image does appear to permit the very accurate drawing of hazard maps. In addition, high-resolution DTMs can be extracted automatically from the stereo data so it is also possible to determine building heights, to predict flood damage, and to monitor and plan better land-use. The drawback being that at 70$ per square km it is very expensive.

Conclusion - Drawing up a hazard map using any of the above remote sensing sensors can prove a very useful tool in the long run if certain criteria are followed. Detailed, expert knowledge of the area under study is imperative and after drawing up the first preliminary map it is essential to visit the area to verify the map by "groundtruthing". Using the best available commercial satellite products on the market does guarantee that the end product, i.e.

the output map will provide more and better information than using imagery with poor resolution. However the best use that one can make of remote sensing products is when it is necessary to observe the area under question for a period of time to either study the evolutionary sequence of phenomena or to make future extrapolations, or in extreme cases of monitoring disasters such as oil spillages or floods. In this instance the use of remote sensing will prove the most useful tool and the most cost effective.

7.3 Data processing using GIS

The importance of developing a strategy for the: identification, quality checking, storage and processing of data required to produce both hazard and vulnerability maps must not be underestimated. In this section, key aspects that need to be considered when developing such a strategy, with particular emphasis on the use of GIS in data handling, data processing and map production, are developed.

Nature and versatility of data - The data required for karst groundwater vulnerability mapping are for the most part those routinely required in most hydrogeological investigations, for example the hydraulic conductivities of the aquifer and overlying rocks. In addition however, data concerning the unique attributes of karst groundwater systems are also required, for example in relation to flow concentration and point recharge.

When considering the parameters to be used, the applicability of the collected data set to the parameter in question must be reviewed. For example, the use of a Recharge-Precipitation factor within the method requires meteorological data that would be needed for any hydrogeological study. Conversely, the K factor that assesses the extent of karstification of the aquifer and used in source protection maps, needs data and a methodology particular to karsts.

Limitation of Data - All data used in the production of a vulnerability map brings limitations to that map. For example, the interpolation of a small number of data across relatively large areas can have significant impact on the accuracy of the vulnerability map. Therefore great care must be taken when interpolating limited data sets with point specific characteristics (including permeability and layer thickness) across disproportionately large areas.

Data Quality - Data errors can be derive from; the measuring process, the interpolation or estimation of data, spatial or temporal modeling, and from mistakes in data entry (Gogu 2000). Often, spatial data are entered into a GIS by digitizing a map. This process can multiply the existing map errors. Uncertainties contained in the source map (geology, soils, landuse, etc.) as well as those amplified by the GIS operations are particularly important sources of error. The uncertainties related to the representation of map data are also significant. However these uncertainties are not very easy to represent in real world situations. For example, polygon mapping units are most likely to be less homogenous than the source (or the eventual output) map may suggest. Indeed, in the real world, the classes shown on the map are not homogeneous and the class boundaries are often gradual. It follows that it is difficult to represent these uncertainties in real world situations.

In addition, there are many other sources of error on source paper maps that are related to the processes of reproduction, deformation, and generalisation. Data derived from interpolation of point observations contain other types of error. The latter are induced by the measurement and interpolation procedures.

In consequence, there is a considerable need for the development of uncertainty analysis (Heuvelink 1998) in Geographical Information System (GIS) with respect to attribute data, positional accuracy, lineage, logical consistency, completeness, and temporal accuracy.

There are many recognised sources of errors in groundwater vulnerability assessments; the National Research Council (1993) classified them as shown in Tab. 16.

Tab. 16: Potential sources of error in groundwater vulnerability maps

Category	Error type
Errors in obtaining data	Accuracy in locating sites Sample collection and handling Laboratory preparation and analysis Interpretation
Errors due to natural spatial & temporal variability	Random sampling error Bias Regionalisation, extrapolation, interpolation Scale effects, changes in variance due to averaging Interpretation
Errors in computerisation (digitising) & storage of data	Data entry Data age Changes in storage format Errors in programs to access data Use of surrogate data and procedures Adjustment in scale Determining boundaries Changes in representation of data Interpretation
Data processing errors	Numerical, truncation, and round-off errors Discretisation errors Problems in solution convergence Interpretation
Modeling and conceptual errors	Process representation and coupling Parameter identification Scale effects Interpretation

Data processing - The reliability and validity of groundwater analyses and in particular that of hazard, vulnerability, and risk assessments, depend strongly on the availability of large volumes of high quality data. Putting all such data into a coherent and logical structure within a computing environment allows the development of powerful tools for use in hydrogeological studies.

For the purpose of hazard mapping, the potential sources of groundwater contamination need to be integrated and analysed in a geographical context in terms of maps. Such maps are classified by Struckmeier and Margat (1995), as special purpose maps within the hydrogeological maps category. As a result of IT developments within the last decade, Geographical Information System (GIS) and Computer Aided Cartography (CAC) have now reached a level of capability that allows for the creation and manipulation of geographic datasets and the production of high quality output maps, which meet the increasingly stringent requirements of groundwater engineers and scientists.

Geographic Information System and Cartography Software - In broad terms, mapping software can be divided in two types: Geographic Information Systems (GIS) and Computer

Aided Cartography (CAC) software. GIS programs generally provide the ability to store, manage, and analyse georeferenced and interconnected data. CAC software on the other hand is mainly used for a high quality visualization of spatial information (Hurni and Christinat 1997). Tab. 17 shows the main differences between GIS and cartography software.

Tab. 17: Comparison between GIS and cartography software (WYSIWYG* - What You See is What You Get)

Geographic Information System (GIS)	Computed Aided Cartography Software (CAC)
Presentation of modeled, real-world objects	Symbolization of objects
Concept of topology is essential for objects modeling	Graphical presentation only
Strict use of layer technique (which ignores e.g. analogous presentation of bridges and subways)	Layer techniques with special cartographic options (e.g. for the presentation of bridges or subways)
The meaning of objects is defined by the attributes in a database	The meaning of objects is defined by their symbolization
Manipulation and analysis functions	2D visualization and configuration options
No generalization of input data	Generalization and cartographic presentation of input data
Not necessarily WYSIWYG* presentation	Cartographic WYSIWYG* presentation (transparency, masks, depth effects etc.)
Integration of raster layers, switching between the different modes may be possible	Raster layers combined with vector layers
Simple printing and plotting options only	Output options conceived for high quality printout
Rather complex use	Rather simple use (mainly Desktop Publishing Programs DTP)

Basic concepts of GIS - a GIS is defined as a system for input, storage, manipulation, and output of geographically referenced data (Goodchild 1996). Most GIS provide a means of representing the real world through integrated layers of constituent spatial information (Corwin 1996). Geographic information can be represented in GIS as objects or fields.

The *object approach* represents the real world through simple objects such as point, lines, and polygons. The objects, representing entities, are characterised by geometry, topology, and non-spatial attribute values (Heuvelink 1998). In hazard mapping, such spatial objects may for example include animal husbandry farms, petrol stations, and industrial pipelines. Attribute values relevant to the assessment of the harmfulness of these objects (which are the hazards in this instance) are likely to include among other the number of animals, the age of the fuel tanks, and the diameter of the pipes respectively.

The *field approach* represents the real world as fields of attribute data without defining objects. Some examples relevant to vulnerability mapping are strata elevation, hydraulic head, and vulnerability zones. This approach provides attribute values in any location.

In GIS, this distinction between *objects* and *fields* is often associated with *vector* data models and *raster* data models. The *vector model* represents spatial phenomena through differences in the distribution of properties of points, lines, and polygons. In this system, each layer is an adapted combination of one or more classes of geometrical features. A *raster model* consists of a rectangular array of cells with values being assigned to each cell. In the raster model, each cell is usually restricted to a single value. Thus, representing the spatial distribution of a number of parameters or variables requires multiple layers.

Environmental specialists require clear representations of the spatial variation of data (Gogu et al. 2001). In GIS, there are two solutions to this problem: (1) field variables (a variable can be given a single, well-defined value at every location), and (2) kernel functions (spatially continuous functions). For digital representation of the spatial variation characterised by fields, six models are distinguished: raster model, grid model (rectangular array of sample points), point model (area irregular distributed sample points), contour model (isolines), polygon model (polygons holding average attribute values), and TIN (triangular irregular network).

Storing and manipulating data through spatial relationships can be done with GIS software packages using a "Geo-relational" model or "Geo-database" model. The first consists of linking a relational database to geometrical features. The modelled entities are organised into categories sharing common characteristics (e.g. points representing wells, piezometers, or gallery wells). A table represents each category. The different attributes occur as columns in the table, and the rows assure the data registration. Other independent tables representing time or spatial dependent data can be attached. Relationships "one to one" or "one to many" can be established between these tables (Levene and Loizou 1999).

The Georelational model uses points, lines, polygons, and related attribute-tables to define various properties. In the Geodatabase model entities are represented as objects with properties, behaviour, and relationships. For example, a "well"-object can be found within a library of objects with the entire attribute scheme attached. The user can simply take it, place it on the map, and fill the data in the tables attached to the object. The Georelational and Geodatabase models are actually very similar. However, the Geodatabase model represents a recent improvement in the implementation of the Georelational model.

Software tools for hazard, vulnerability and risk mapping - basic tools that aid the mapping process are (1) relational databases, (2) geographical information systems (GIS), and (3) computer aided cartography software. It is important to recognise that an organised procedure is needed to store and retrieve the large amounts of data that are usually generated during a risk analysis. A good data management platform should allow quick efficient access to available data (usually in multiple formats) and produce output that is compatible with other applications.

The number of computer programs for spatial data analysis and visualisation has increased during the last few decades. Tab. 18 lists some of the more widely used software packages on the market at this time.

Tab. 18: Overview of the currently used GIS packages and Computer Aided Cartography Software

Geographic Information System (GIS)	Computed Aided Cartography Software (CAC)
GIS Standard Packages	**Desk Top Publishing Software**
Arc/Info	Illustrator
MGE (+ Microstation + RDBMS)	CorelDraw, etc.
Geomedia	
Microstation Graphics	**Computer Aided Design (CAD)**
Smallworld	Microstation
GRASS GIS	AutoCAD, etc.
IDRISI	
ILWIS, etc.	**Mapping Programs**
	RegioGraph
GIS Desktop Software	Map Viewer
OCAD	THEMAK 2
Dry/Nuages	MERCATOR
ArcView 3	EASYMAP, etc.
Atlas GIS	
Map Info	**Programs with additional mapping extensions:**
PC Map	AutoCAD (+ extension)
SiCAD special desktop, etc.	FreeHand (+ extension MaPublisher)
	Illustrator (+ extension MaPublisher)

Basic information requirements - There are three essential requirements to support the production of the maps: a reliable topographic map, good quality data (as required for purpose of hazard, vulnerability and risk analysis respectively) and a suitable representational scheme and map legend.

The topographic base map guides the orientation on the surface and serves as a source of information (e.g. river network, watersheds, and surface properties). It must be up to date and contain the essential topographic information appropriate to its scale.

Cartographic principles - Cartography is a form of communication for describing locations, places, and interpreting two-dimensional arrangements of features. The ability of maps to communicate useful information to a user depends on the map conventions used. The principal aim of cartography is to convey complex thematic information to the map user in an exact, clear, and easily readable manner. This implies that the map must not be overloaded with information and that the colours and symbols used on the map follow a logical system, explained in a legend. To achieve this, it is necessary to follow some basic principles of cartographic communication and map design. Such cartographic rules should be strictly followed, and therefore it is necessary to guide researchers with cartographic directions for use.

The basic elements to produce a reliable map concern the careful selection and use of:

- Symbols
- Classes and class boundaries
- Colours – full colours should be used for the predominant features, whilst hatching is usually reserved for secondary features.
- Scale and projection
- Map annotations (e.g. legend, scale bar, title etc.)

The first three elements are essential for hazard, vulnerability and risk mapping and an inappropriate selection or use can lead easily lead to misinterpretation of the map. Because projection systems differ from one country or area to another, the choice of map projection should be made only after due consultation with the relevant geodetic and cartographic authorities. Each map must have a clear title, scale, date, publisher and copyright to allow correct citation. Furthermore, the legend and the map should be regarded as a single unit, printed on the same paper sheet. Where legends have been prepared in a national language, it is advisable to include the terms in one or several international languages.

Digital cartography - Modern cartographic information systems allow to record, edit and display spatial topographic and thematic information. The task of digital cartography is to make a useful selection out of the processed data and to present it in a useful form on paper or on electronic media (Hurni and Christinat 1997). In order to achieve this there are several functional demands on a digital cartographic production system that can be classified as follows:

- Input of analogue and digital map data by scanning, digitalisation, data import.
- Visualisation of data on screen.
- Internal coordinate system.
- Prerequisite work including global raster image manipulations (rectification – Helmert, affine, projective, rubber-sheeting and other transformations, change of projection, overlay of several raster files, merge of several layers, separation of colours, and others), global vector manipulations (rectification, change of projection, overlay, merging, data elimination, algorithms for data reduction and line simplification and others)
- Editing of raster data.
- Editing of vector data.
- Hybrid processing of vector and raster data.
- Text processing.
- Processing of continuous tone data (image processing).
- Data output.

Conclusions - Because of the large volumes of data required for a consistent and reliable analysis, the use of GIS is strongly recommended for the purpose of groundwater hazard, vulnerability, and risk assessments. This technology can easily sustain the tasks and operations used for the analysis of the spatial and physical relationships between critical environmental parameters. GIS represents a useful tool also for creating various maps and for quick and simple display.

Most of the hazard, vulnerability, and risk mapping procedures are using only the basic GIS tools. Functions such as map overlay, reclassification, and query assist the principal operations of the assessment methods. GIS is particularly suited to overlay and index techniques. However, these types of operations and mapping procedures can already be rigorously performed using a GIS based database. Also, parameter quantification operations can be performed easily using various spatial queries. The automatic generation of contour maps as well as of error evaluations is achieved by performing spatial statistics on the various data (numbers of measurements, mean value, standard deviation, etc), and leads to reliable assumptions on different parameters.

Yet, in order to provide high quality maps, the map outputs obtained from GIS are best treated using specialised, cartographic software. In this respect, the current market offers a large range of professional solutions. The groundwater specialists have to be conscious of the clear distinction between the GIS and cartography, even though these two domains are tightly linked.

7.4 Demonstrating the reliability of our maps

All groundwater vulnerability maps are conservative simplifications of prevailing hydrogeological conditions. Often these simplifications are expressed in terms of a conceptual model that embraces the hydrogeological theories being employed to produce a vulnerability map. In order to test the validity of both the conceptual model on which the mapping technique is based and to demonstrate the usefulness of a vulnerability map at specific locations, a program of verification and validation must be undertaken that ensures the Quality of the final product.

Quality – the totality of features and characteristics of a product or service that bear on its ability to satisfy stated or implied needs (ISO 8402: 1986 Quality Vocabulary)

The importance of verification and validation to the groundwater vulnerability mapping process cannot be overstated. If well designed, they can provide a degree of independent corroboration that raises the “currency” of the vulnerability assessment being employed. The actions taken during verification and validation can be collectively referred to as Quality Assurance.

Quality Assurance (QA) – all those planned and systematic actions necessary to provide adequate confidence that a product or service will satisfy given requirements for quality (ISO 8402: 1986 Quality Vocabulary)

When collecting data for use in the production of a vulnerability map, due consideration must be given to the quality of the data to be employed and the rigors of both the methods and methodologies used to collect that data. The procedures for assessing the quality of collected data are often a function of the way in which the data was acquired or derived. The use of detailed methodologies for each method used to collect data in a consistent and repeatability way is referred to as Quality Control.

Quality Control (QC) – the operational techniques and activities that are used to fulfil requirements for quality (ISO 8402: 1986 Quality Vocabulary)

When preparing data for use in vulnerability mapping, due consideration must be given to ensuring that the quality of the data used is known. Appropriate allowances can then be made within the map-making process that ensure that data of poor or unknown quality do not unduly influence the final map.

Validation – When undertaking vulnerability mapping in a given area, “validation” procedures should be used to ensure that the conceptual understanding of prevailing hydrogeological conditions are valid. Validation methods can include: the analysis of hydrographs and chemographs, isotopic chemistry, use of artificial tracers and the use of analytical and numerical models. The results of any calculation, test or investigations, that are independent of the vulnerability method being used, can be included within the validation process.

Verification – When developing a vulnerability mapping method verification is used to determine that the correct or expected results are realised when implementing the new technique. Verification is most effectively employed when developing physically based methods that utilise numerical “models” to describe the degree of vulnerability across any given area. Verification is not intended to test or prove the validity of the theoretical basis of the vulnerability method being considered, but is used to ensure that the method produces the intended results.

Verification may best be considered as quality assurance of the mechanics of a particular method. This process necessarily includes logical and procedural checks to calculations and in some instances can include comparison with other methods that derive the same or similar outputs. Verification is not the procedure that is used to consider the validity of a groundwater vulnerability map for a specific area. Process that maybe used to verify a particular method can range from simple calculation through to detailed numerical modelling.

7.5 Key Map Validation Tools

7.5.1 Hydraulics and spring hydrographs

The spring hydrograph and/or chemograph provide information on the behaviour of the karst system as a whole; this includes the karst aquifer as well as the catchments of any influent streams. Conversely, borehole hydrographs mainly relate to local effects in the aquifer adjacent to the borehole, which as a result maybe difficult to interpret without complementary data, such as tracing tests and chemographs; this information may only be helpful if it can be compared with data from an associated main spring.

According to Mangin's method (Mangin, 1970; Marsaud, 1996), the analysis of spring hydrograph provides information on:

- the infiltration processes: rapid verses slow infiltration, delayed infiltration (because of sediment or snow cover, or storage in the epikarst),
- the role of the saturated karst: organisation of the conduit network (drainage function), and the relative importance of storage.

Spring chemographs (Plagnes and Bakalowicz, 2001) can be interpreted without the support of the spring hydrograph. They can assist in defining the importance of the epikarst storage and of the "flash flood effect" during flood events, i.e. the appearance of a hydraulic connection between the epikarst and the karst surface and the conduit network of the saturated zone. This is shown in Fig. 40.

Borehole chemographs are much like their hydrographs, generally difficult to interpret due to the impact of prevailing local conditions on the chemistry of the groundwater. Time variable chemical or isotopic changes at a spring or a well depend in part on the processes taking place in the aquifer. However these processes and their impact on chemical or isotopic content are related to flow conditions. In order to establish the hydrogeological functioning of a karst system, the use of artificial and/or natural tracers can be used to decipher the impact of prevailing flow conditions from other processes.

The vulnerability of groundwater within a karst aquifer is heavily influenced by flow conditions. If rapid infiltration and fast flow in conduits is the dominant condition, then sorption, degradation, cation exchange, dispersion and dilution will be of minor importance. In such conditions, a pollution event will travel rapidly through an aquifer; and the pollutant concentration at the outlet will be high. But the system has a short residence time, i.e. the event quickly passes and the aquifer quickly returns to its initial state. In the same way, diffuse pollution rarely presents any permanent, cumulative effect.

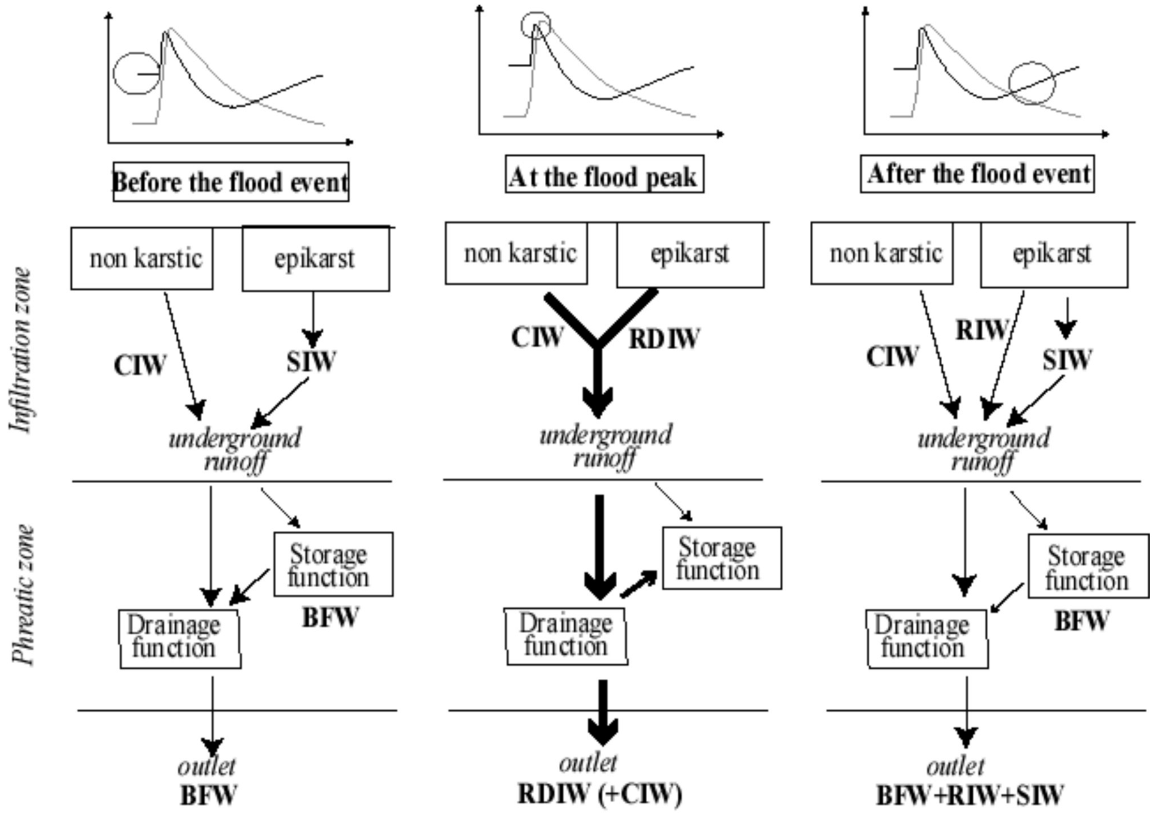

Fig. 40: General conceptual representation of different stages of a flood event, comparing both a flood hydrograph and chemograph (BFW: base flow water, RIW: rapid infiltration water, RDIW: rapid delayed infiltration water, CIW: concentrated infiltration water, SIW: slow infiltration water) (Bakalowicz, 2001).

7.5.2 Natural Tracers

Naturally occurring tracers can sometimes present opportunities for the validation of groundwater vulnerability assessments. By definition their presence within a hydrological system is a result of natural conditions rather than purposely introduced as part of a specific experiment. Natural tracers can be grouped into three principle groups:

- Ions and organic molecules in solution
- Dissolved gases like radon and CO_2
- Environmental isotopes
- Particles, including turbidity and bacteria/microorganisms

Their use in validating a groundwater vulnerability assessment is somewhat different to that of artificial tracers, as their release into the hydrological environment cannot by definition be controlled. They can be used to support the identification of changes in the natural groundwater chemistry, or microorganisms' ecology within a hydrogeological system.

Natural tracers, whilst not providing information form precisely known injection points like artificial tracers, can provide information on an aquifer wide basis. This can include; the whole recharge area and all layers along each flowpath from topsoil through to the discharge point. As a result natural tracers can in some instances not only be used for the validation of a vulnerability assessment in a defined hydrogeological system but also provide for the comparison of vulnerability in two different hydrogeological systems.

Of the many natural tracers, which are placed into four key groups as shown above, only some examples of “Environmental Isotopes” and “Ions and organic molecules in solution” are described below.

Environmental Isotopes - Several isotope systems can be used to validate intrinsic and specific vulnerability assessments. Those most commonly used to distinguish infiltration areas and groundwater travel times are the “water-isotopes” oxygen-18 ($^{18}O/^{16}O$), deuterium ($^{2}H/^{1}H$) and tritium (^{3}H). Less frequently used are; radon (^{222}Rn) and chlorine-36 ($^{36}Cl/Cl$) for dating very short (weeks) or extremely long (>10 000 years) residence times or sources. Strontium-87 ($^{87}Sr/^{86}Sr$) and sulfur-34 ($^{34}S/^{32}S$) are mainly used to identify contact and leaching of carbonate and sulphate rocks of variable stratigraphic age. Nitrogen-15 ($^{15}N/^{14}N$) and oxygen-18 ($^{18}O/^{16}O$) of nitrate (NO_3) are sometimes used as contaminant tracer in specific vulnerability studies as they have “water like” behaviour. Similarly boron-11 ($^{11}B/^{10}B$) higher and lead-207 ($^{207}Pb/^{206}Pb$) can be used as contaminant source indicators in evaluating specific vulnerability assessments. Isotope that have so far very rarely used in carbonate aquifers are uranium ($^{234}U/^{238}U$) and tritium – krypton-85 ratios ($^{3}H/^{85}Kr$) and ratios of tritium to helium (^{3}H /^{3}He) (Cook and Herczeg 2000).

Oxygen-18 and deuterium: These Isotopes of the water composition hydrogen and oxygen are ideal tracers for infiltrating precipitation or surface water. If from monitoring stations, either monthly rain or surface water samples, the isotopic composition is reasonable well known, then infiltration areas can be differentiated. Data on the isotopic composition of precipitation can be obtained from the worldwide IAEA-precipitation-network (International Atomic-Energy Agency) or perhaps more applicably from a denser national network. The isotopic composition of precipitation changes with altitude (“altitude effect”) and season (mainly temperature dependence). In addition, the isotopic composition of precipitation changes slowly over decades, supporting the necessity of long term monitoring. Therefore the infiltration area can be evaluated by differing O-18 composition due to changing altitude, season or sources (river or lake water). Due to the seasonal variation of the precipitation and frequent O-18 samples of spring or well water the mean residence time (<few years) can be considered by comparison of precipitation and groundwater data (Coplen and others 2000,).

Tritium (^{3}H): Tritium within water molecule is the only radioactive isotope ($T_{1/2}$=12.43 years) of hydrogen. Besides natural tritium within the atmosphere, large quantities were introduced into the hydrological cycle by atmospheric thermonuclear testing in the 1950s and 1960s. Tritium measurements allow validation of groundwater residence times of up to 40 years, which can support intrinsic vulnerability assessments. In order to obtain reasonable age estimates, similar to those made using oxygen and deuterium, the input function for the last decade should be reasonably well known (Solomon and Cook 2000). Tritium can be present in; sanitary and industrial waste sites (watch industry) as well as sites used in the nuclear industry. In such case tritium can be present in relatively high quantities and has been used as an artificial tracer (Rank and others 1997).

Ions and organic molecules in solution - A new approach is presently developed to study the behaviour of infiltration water, using its natural content of Dissolved Organic Carbon [DOC] (Emblanch and others, 1998, Batiot and others, 2002). DOC is dissolved by infiltration water in topsoil. As this tracer mineralises during time, its residual amount in karst groundwater (drains, boreholes, outlet) enables evaluation of residence time. This evaluation can provide for an estimation of transit time and thus can support the validation of a vulnerability assessment. Furthermore, assessment of this transit time can provide information on the behaviour of certain organic tracers, which maybe used in assessing specific vulnerability.

7.5.3 Artificial Tracers

A straightforward method to validate a groundwater vulnerability assessment is to release an artificial tracer at the ground surface and to observe its transport from the origin along the pathway to the target. The demands on a validation process by means of artificial tracer tests can be directly derived from the concept of groundwater vulnerability as described in this report. Three basic aspects have to be considered:

- Travel time of a contaminant from the origin to the target;
- Relative quantity of the contaminant that can reach the target;
- Attenuation processes decreasing the contaminant concentration.

In case of an accidental (instantaneous) release, the duration of a contamination event at the target can be used as an additional criterion.

All these properties can be directly obtained from a tracer breakthrough curve: The mean transit time can be taken as the 'travel time' (in some cases it may be more adequate to take the time of the first arrival or the time of the maximum concentration). The recovery rate reflects the 'relative quantity'. The normalised maximum concentration (divided by input mass or input concentration) shows the degree of 'attenuation'. And the time period, in which the tracer concentration exceeds a given limit, can be defined as the 'duration of contamination'.

Intrinsic vulnerability only takes into account the hydrogeological characteristics of an area but is, by definition, independent from the properties of the contaminants. Specific vulnerability additionally depends on the interaction between the system and the different types of contaminants. For validating a vulnerability assessment by means of tracer tests, conservative tracers are thus preferred for intrinsic vulnerability while reactive tracers can be used for specific vulnerability (Goldscheider et al. 2001). However, most of the available tracers are reactive while there are only few conservative tracers, mainly the isotopes of the water molecule. Some salts (anions) and fluorescent tracers behave almost conservative as well and can thus be used to validate an intrinsic vulnerability assessment (Käss 1998).

The transport of conservative contaminants is controlled by advection, dispersion, diffusion and dilution. Long-term storage and surface runoff may decrease the relative quantity of contaminants that can reach the target. The transport of a reactive substance is additionally influenced by a large variety of processes, e.g. cation exchange, biodegradation, oxidation or reduction, precipitation, filtration, sedimentation and volatilisation (Fetter 1999).

The set up of a tracer test for validation purposes must take into account the origin-pathway-target model for groundwater vulnerability assessment. The tracers should be released at the *origin*, usually the land surface, and the breakthrough should be observed at the *target*. For source vulnerability, this is a spring or well. For resource vulnerability, the ideal observation point would be at the basis of the unsaturated zone, directly above the groundwater surface. However this zone is only locally accessible via caves or galleries.

Although tracer tests can be a direct method to validate a vulnerability assessment, there are several problems and limitations (after Goldscheider et al. 2001):

- Vulnerability maps show information for large areas while tracer tests only allow certain points to be checked.
- The results of tracer tests depend on the hydrologic conditions and the injection mode. A tracer that is put on the land surface in a dry summer can be completely absorbed in the soil while it can reach the spring quickly in a rainy winter.

- The results also depend on the properties of the tracer. Conservative tracers are preferred for the validation of intrinsic vulnerability maps but most tracers are not completely conservative (Käss 1998).
- Tracer tests are practicable if the travel time is not too long and the dilution is not too high. They can thus be applied in highly vulnerable zones but not in zones where travel times of years are expected.
- As it is difficult to sample at the groundwater surface, the use of tracer tests for the validation of resource vulnerability maps is problematic.

There are only few examples on the validation of vulnerability maps using artificial tracer tests. Goldscheider et al. (2001) validated an EPIK vulnerability map by means of seven tracer injections. Six of the tracers were spread at the land surface with a watering can; one tracer was injected in a swallow hole. After the injections, an artificial rainfall of 20 mm was produced. The obtained travel time (in this case: time of maximum concentration!) was almost identical for the tracer injected in the swallow hole and the tracers injected on soils of low permeability. This was explained by the presence of macropores. However, there was a strong correlation between the assessed vulnerability, the recovery rate and the normalised maximum concentration. Perrin et al. (2002) used both artificial and natural tracers to assess and validated the intrinsic and specific source and resource vulnerability of different test sites in the Swiss Jura Mountains. They also showed the strong influence of the contamination scenario (type of release of tracers) on the resulting breakthrough at the target.

The use of toxic compounds in tracer tests to validate a particular specific vulnerability assessment is both unthinkable and often illegal. As a result only harmless substances can be used as tracers (Behrens et al. 2001). This presents some difficulty when trying to simulate the fate of toxic heavy metals, chlorinated solvents or pathogen viruses. In this case, we must select non-toxic substances that are known to behave in a comparable way as the respective toxic substances. As an example, some marine bacteriophages show similar transport characteristics in groundwater as particular pathogen viruses but are completely harmless for human beings and aquatic ecosystems. They can thus be used to simulate the behaviour of pathogen viruses (Rossi et al. 1998) and can also be used to validate a virus-specific vulnerability assessment.

7.5.4 Analytical and numerical modelling

Unfortunately, full validation of a vulnerability assessment using analytical and numerical models is an objective that is as yet out of reach. The principle of validating the results of a model (the chosen vulnerability assessment method is always a simplified model of the reality) using another model is a difficult concept. However, in practice a numerical model used in validation can increase the “degree of confidence” in the main assumptions made during the vulnerability assessment. With validation in mind, when utilising numerical models, sensitivity analysis can be used to examine the most uncertain and influential parameters used in the vulnerability method. Additionally, any validation of results requires clear criteria that are not easy to define (e.g. legal aspects or local agreements or regulations).

Numerical models can be chosen on their degree of complexity and data need, much like choosing a particular vulnerability assessment technique. These can range from black-box models (which are unlikely to add confidence in this particular context), to full physically based and spatially distributed models that can be used for accurately computing contaminant concentrations in groundwater. Whatever the degree of complexity of the numerical model used, a calibration using historical data and a validation on a remaining data set (not used during calibration) are needed. To summarise, four kinds of data are needed in order to proceed with accurate and reliable numerical modelling:

- Geometrical and geological data for the spatial discretisation of the modelled domain;
- Parameters or property values for characterising each of the discretised zones with respect to each simulated process. This is likely to provide most difficulty in terms of uncertainty of the parameters used in the modelled processes, due to scale effects and the lack of data that could be used to describe aquifer heterogeneity;
- Stress-factors influencing groundwater quantity (infiltration, pumping and re-injection rates) and groundwater quality (input/output of contaminant fluxes);
- Historical data (distributed both spatially and through time) relating to groundwater quantity (measured piezometric levels, water pressures, spring discharges, hydrographs, tracer tests) and groundwater quality (measured concentrations, chemographs, tracer tests).

Most vulnerability assessment methods were developed to avoid the need for extensive data sets and simplify description of the processes that may act on contaminants as they travel along a flowpath to either the groundwater table or source (spring or pumping well). In practice, only partial validations are feasible using numerical models, as they consider only some of the active processes. However in some instances a model may consider most of the active processes but only in a very limited and well-studied part of the area being assessed.

Trying to describe the main processes involved along the path of a contaminant, one can distinguish how numerical models, describing each of the main compartments of the water cycle, should be coupled in order to form an integrated numerical tool describing contaminant transport in a mechanistic way. As described in Fig. 41, it involves:

- Computation of water budget at the land surface in order to assess actual infiltration. This is not an easy task and many recently developed tools aim to simulate these processes accurately. On that particular topic, interesting and useful literature can be found in (among others): WILSON & LUXMOORE 1988, SMETTEM et al. 1991, LARSEN et al. 1994, MILLY 1994, SIMMERS et al. 1997, BEVEN 2000.
- Use of this modelled infiltration (and associated mass of contaminant) is then used as an input for modelling 1D vertical transport (in some particular cases a 3D approach is needed) of a dissolved contaminant through the different layers (porous, fissured or karstified) of the unsaturated zone. Recent developments in modelling the unsaturated zone take into account the possible influence of epikarst, macropores, and include bio-chemical reactions. However, in practice, huge uncertainties remain and are linked to the values given to each parameter and mostly in the highly karstified parts of the unsaturated zone. On these challenging topics interesting references are (among others): BEVEN & GERMANN 1981, CHEN & WAGENET 1992, CHEN et al. 1993, FORSITH et al. 1995, GWO et al. 1996, GERKE & VAN GENUCHTEN 1993, PERFECT et al. 1996, THERRIEN & SUDICKY 1996, DEMARCO 1998, GRIFFIOEN et al. 1998, BROUYERE 2001.
- Use of the computed fluxes of contaminated water at the base of the unsaturated zone can be used as input to a 3D groundwater saturated flow and transport model of the karstic aquifer. Depending on the scale of the study and the knowledge of the degree of heterogeneity, an equivalent porous medium approach may be adopted. Groundwater modelling in karstic systems is still a challenge. Useful references are the following (among others): KIRALY & MOREL 1976, CULLEN & LAFLEUR 1984, SAUTER 1993, SMITH & SCHWARTZ 1993, HUNTOON 1995, QUINLAN et al. 1996, DASSARGUES et al. 1997, DASSARGUES 1998,

JEANNIN & GRASSO 1997, DASSARGUES & DEROUANE 1998, HALIHAN & WICKS 1998, JEANNIN et al. 1999, WICKS & HOKE 1999.

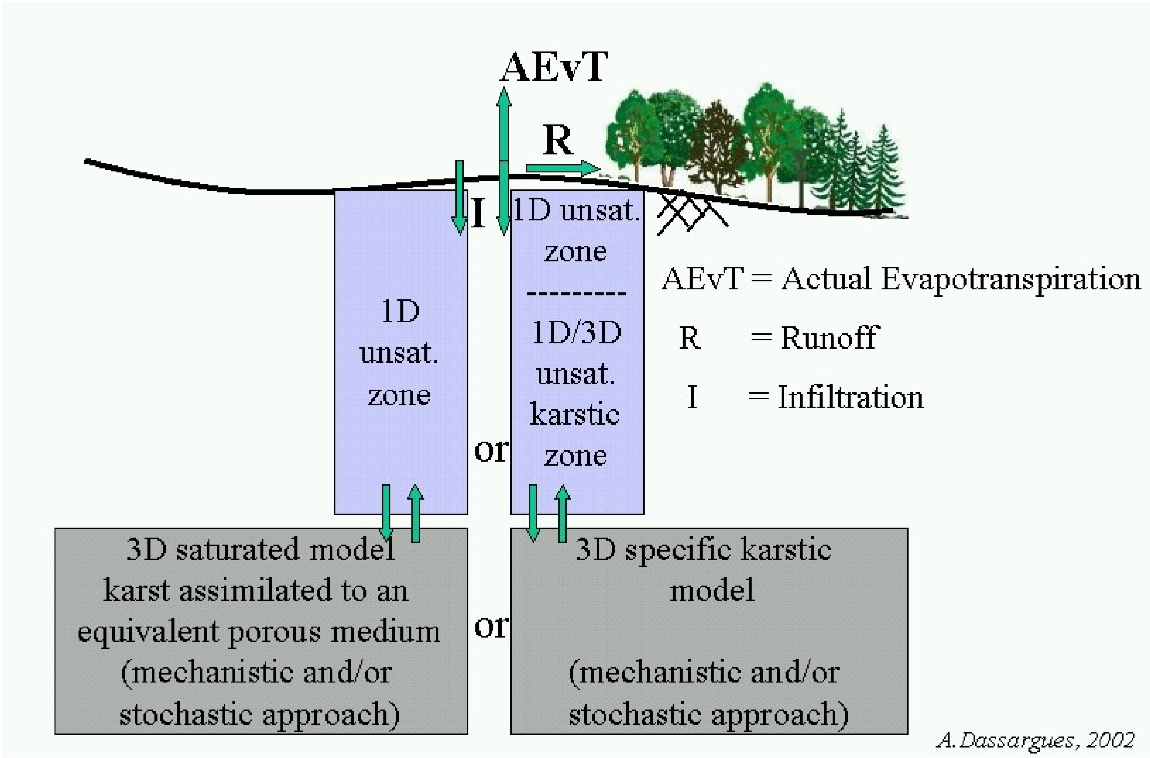

Fig. 41: Conceptualising the main processes affecting transport of dissolved contaminant from the soil surface until the spring or the pumping well: coupled numerical models forming an integrated numerical tool.

In conclusion, numerical models are useful as tools for consistently interpreting the results of field measurements and experiments. Calibration of numerical models using these measurements will ensure the optimum use of this information for validating (at least to some extent) the vulnerability assessment. They can also be considered as useful intermediate tools between field measurements and vulnerability assessments. After calibration, one can perform sensitivity analysis to check how results can vary in different stressed scenarios ('what if' simulations) or to consider the uncertainty of the parameters used. Results of this analysis allow the validation of the assumptions made in the adopted vulnerability assessment technique.

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Part B: Methods and Applications

1 Mapping Methods

1.1 Overview

The three Working Groups of COST Action 620 followed their tasks in a slightly different way. Working Groups 2 and 3 developed not only a conceptual framework, but also commonly agreed, detailed **methods** of specific vulnerability mapping and hazard mapping respectively, described in part A of this report.

Working Group 1 of COST 620 only aimed at developing a commonly agreed **approach** to intrinsic vulnerability mapping and assessment, which is broad enough to encompass all European conditions but sufficiently flexible to be customised for successful use in individual karst areas. The proposed “European Approach” presented in part A of this report considers four factors: overlying layers (O), concentration of flow (C), precipitation regime (P) and karst network development (K). Both resource and source vulnerability maps can be prepared by a combination of these factors. This approach comprises a conceptual framework but does not prescribe detailed guidelines, tables and formulae to quantify vulnerability. Therefore, different individual groups participating at COST 620 have proposed different individual methods of intrinsic vulnerability mapping, which were discussed within Working Group 1, and which are more or less based on the conceptual model.

The first method coming from Working Group 1 of COST 620 was the PI method (Goldscheider et al. 2000). It was developed at the Department of Applied Geology in Karlsruhe (AGK), in close cooperation with the Federal Institute for Geosciences and Natural Resources in Hanover (BGR). The PI method served as a basis for the further development of the conceptual model of the European Approach. Two factors are considered – the protective cover (P) and the infiltration conditions (I). These are largely identical to the factors O and C of the European Approach. However, as the PI method was previously developed, different terms are used for the same factors. The PI method does not consider the karst network development (K factor of European Approach) and is thus only applicable for resource vulnerability mapping. The precipitation regime (P factor of European Approach) is not uses as an independent factor, but is included in the assessment schemes for the P and I factor. The PI method allows for a qualitative assessment of vulnerability and requires a relatively detailed database. Up until now, the method has been applied in 12 test sites, some of which are described in this report.

VULK is an analytical computer programme, which was developed at the Hydrogeology Centre of Neuchâtel (CHYN), under the scope of COST 620 as a tool for intrinsic vulnerability assessment (Jeannin et al. 2001). The acronym stands for VULnerability and Karst. The basic idea is to model the breakthrough curve at a defined target (resource/source) resulting from a hypothetical release of contaminants at a given point (origin) at the land-surface. The calculated curve allows for the determination of the theoretical transfer time, duration and concentration level of a contamination event. It is thus possible to characterise the groundwater vulnerability at this given point, or to validate an existing vulnerability map for this point. VULK aims at a physically based and quantitative understanding of vulnerability, but uses, however, strong simplifications. At present, further development aims at implementing concentration of flow into the model, and coupling it with a GIS.

A "Localised European Approach" (LEA) for intrinsic resource vulnerability mapping in England and Wales was developed during 1998–2002 by Suzanne Dunne (2003). It takes into account the overlying layers (O) and the concentration of flow (C). The method is simpler than the PI method and thus appropriate for areas with a less extensive database. It is a straightforward approach that does not use a numerical index. The resulting vulnerability classes are qualitative and relative. The variations of specific electric conductivity at the spring are used for validation. To date the LEA has been applied in six test sites.

The COP method was developed in 2001 and 2002 by the Hydrogeology Group of the University of Malaga (Vías et al. 2002). The COP method follows the conceptual model and the factors O, C and P of the European Approach and aims at a qualitative and relative assessment of intrinsic vulnerability. The saturated zone of the karst aquifer (K factor) is not considered, and so the method can only be used for resource vulnerability mapping. Thus far, it was applied in two test sites in Southern Spain.

The Time-Input Method (Kralik 2001) is an approach to evaluate groundwater vulnerability especially in mountainous areas. The factors considered are the travel time from the surface to groundwater, and the amount of input as groundwater recharge. The vulnerability is expressed in real time and not classified by dimensionless numbers, which allows for validation.

The methods described in this report are not considered to represent the only possible interpretations of the European Approach. Some of the previously existing methods are largely compatible to the conceptual framework proposed by COST 620, and new, better, methods may be developed on this basis.

1.2 The PI method

1.2.1 Background and Overview

The PI method was developed within the scope of COST 620 at the Department of Applied Geology (AGK), University of Karlsruhe. The work was funded by the German Federal Institute for Geosciences and Natural Resources (BGR) and carried out in co-operation with the BGR and the Geological Survey of Baden-Württemberg (LGRB). The complete results of the project (development and application) were reported by GOLDSCHEIDER et al. (2000a); the method was published by GOLDSCHEIDER et al. (2000b).

The PI method was first applied and compared with other methods in the Engen test site, Swabian Alb, Germany (STURM 1999, KLUTE 2000, GOLDSCHEIDER et al. 2000a). Up until now, the method and modification of it was applied in 12 karst systems in 7 countries. Besides the examples described in this report, the method was tested in the following areas:

- Hochifen-Gottesacker, Austro-German Alps (Kunoth 2000, Strathoff 2000, Goldscheider 2002)
- Winterstaude, Austrian Alps (WERZ 2001, GOLDSCHEIDER 2002)
- Mt. Cornacchia and Mt. della Meta, Latium, Italy (COVIELLO 2001)
- Mühltalquellen, Thuringia, Germany (Sauter et al. 2001)

1.2.2 General Concept of the PI Method

The PI Method is a GIS-based approach to mapping intrinsic groundwater vulnerability with special consideration of karst aquifers. It is based on an origin-pathway-target model: The origin of the assumed hazard is the ground surface; the groundwater table in the uppermost aquifer is the target; the pathway includes the layers between the ground surface and the groundwater surface. Thus, the PI method can be used for resource vulnerability mapping. However, if the resource vulnerability map is intersected with a map showing the flow route in the aquifer towards a spring or well, it can also be used for source vulnerability mapping.

The acronym stands for the two factors protective cover (P) and infiltration conditions (I) (Fig. 42). The P factor describes the protective function of the layers between the ground surface and the groundwater table – soil, subsoil, non karst rock and unsaturated karst rock. Thus, it is equivalent to the O factor (overlying layers) of the European Approach (as the PI method was developed previously, the nomenclature is not identical). The P factor is calculated according to a slightly modified version of the German (GLA) method (HÖLTING et al. 1995) (see chapter background methods) and divided into five classes. Form P = 1 for a very low degree of protection to P = 5 for very thick and protective overlying layers. The spatial distribution of the P factor is shown on the P map.

The I factor describes the infiltration conditions, particularly the degree to which the protective cover is bypassed as a result of lateral surface and subsurface flow in the catchment of swallow holes and sinking streams. Thus, the I factor of the PI method is equivalent to the C factor of the European Approach. The I factor is 1.0 if the infiltration occurs diffusely, e.g. on a flat, highly permeable and free draining surface. In contrast, the protective cover is completely bypassed by a swallow hole, through which surface water may pass directly into the karst aquifer. The I factor is 0.0 in such a case. The catchment of a sinking stream is assigned a value between 0.0 and 1.0, depending on the proportion of lateral flow components. The I map shows the spatial distribution of the I factor.

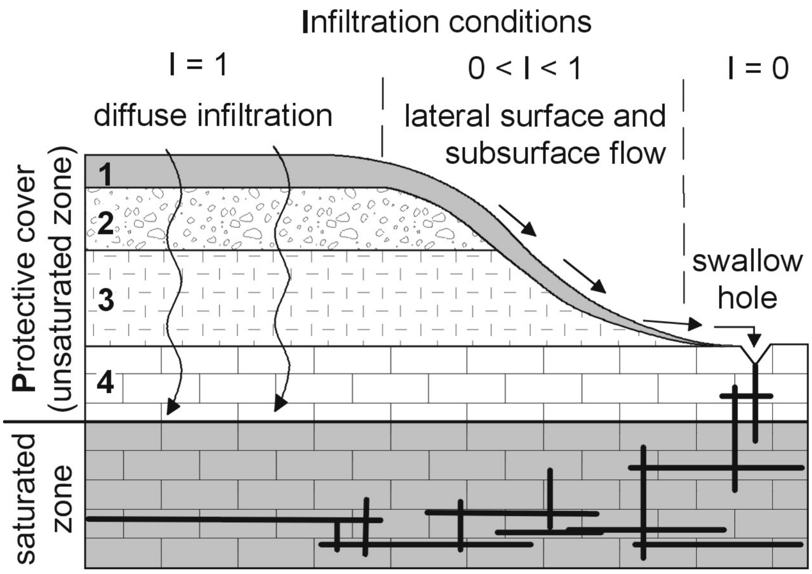

Fig. 42: Illustration of the PI method: The P factor takes into account the effectiveness of the protective cover as a function of the thickness and hydraulic properties of all the strata between the ground surface and the groundwater surface. The protective cover consists of up to four layers: 1. topsoil, 2. subsoil, 3. non karst rock, 4. unsaturated karst rock. The I factor expresses the degree to which the protective cover is bypassed by lateral surface and subsurface flow, especially within the catchments of sinking streams.

The final protection factor π is the product of P and I. It is subdivided into five classes. A protective factor of $\pi \leq 1$ indicates a very low degree of protection and an extreme vulnerability to contamination; $\pi = 5$ indicates a high degree of protection and a very low vulnerability.

The spatial distribution of the π factor is shown on the vulnerability map. Small I and P maps should be printed as insets on this map, so that it can be distinguished how the vulnerability of a particular area is influenced by the two independent factors (Fig. 43).

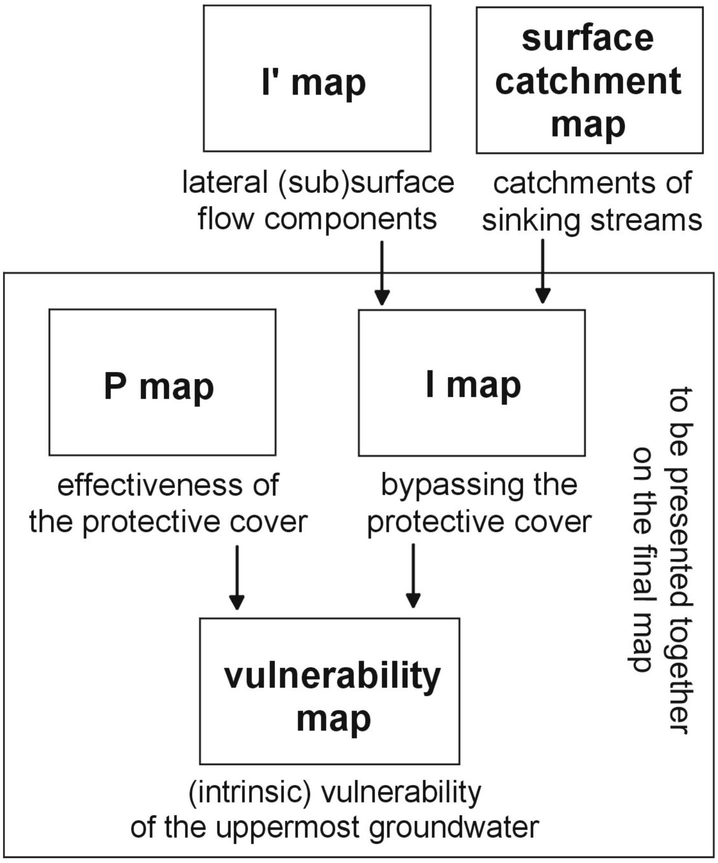

Fig. 43: Simplified flow chart for the PI method: The vulnerability map is obtained by intersecting the P map with the I map. The P map shows the effectiveness of the protective cover as a function of the thickness and permeability of all the strata above the groundwater surface. The I map shows the degree to which the protective cover is bypassed. It is obtained by intersecting the map showing the catchment areas of the sinking streams with the so-called I' map, which shows the distribution of lateral, surface and subsurface flow.

1.2.3 Protective Cover (P factor)

The P factor indicates the effectiveness of the protective cover and is calculated using a modified GLA method (HÖLTING et al. 1995). The calculation scheme is shown in Fig. 44.

The score B for the bedrock is obtained by multiplying the factor L for the lithology and the factor F for the degree of fracturing and karstification. The F factor was modified in order to describe the development of the epikarst and its influence on groundwater vulnerability.

The epikarst is defined as the uppermost zone of karstified rock outcrops, in which permeability due to fissuring and karstification is substantially higher and more uniformly distributed than in the rock below (KLIMCHOUK 1997). Its thickness ranges between a few decimetres and several tens of metres. The possible functions of epikarst are storage and concentration of flow (FORD & WILLIAMS 1989). If the epikarst is developed in a way that leads to extreme concentration of flow, e.g., a bare karrenfield connected with hidden, karstic shafts, the structural factor is assigned a value of zero, expressing that the protective cover of the unsaturated zone below this epikarst is completely bypassed (Fig. 45).

Topsoil - T

eFC [mm] up to 1 m depth	T
> 250	750
> 200-250	500
> 140-200	250
> 90-140	125
> 50-90	50
< 50	0

Recharge - R

Recharge [mm/y]	R
0-100	1.75
>100-200	1.50
>200-300	1.25
>300-400	1.00
>400	0.75

Subsoil - S

Type of subsoil (grain size distribution)	S
clay	500
loamy clay, slightly silty clay	400
slightly sandy clay	350
silty clay, clayey silty loam	320
clayey loam	300
very silty clay, sandy clay	270
very loamy silt	250
slightly clayey loam, clayey silty loam	240
very clayey silt, silty loam	220
very sandy clay, sandy silty loam, slightly sandy loam, loamy silt, clayey silt	200
sandy loam, slightly loamy silt	180
slightly clayey silt, sandy loamy silt, silt, very sandy loam	160

Type of subsoil (grain size distribution)	S
very clayey sand, clayey sand, loamy silty sand	140
sandy silt, very loamy sand	120
loamy sand, very silty sand	90
slightly clayey sand, silty sand, sandy clayey gravel	75
slightly loamy sand, sandy silty gravel	60
slightly silty sand, slightly silty sand with gravel	50
sand	25
sand with gravel, sandy gravel	10
gravel, gravel with breccia	5
non-lithified volcanic material (pyroklastica)	200
peat	400
sapropel	300

Lithology - L

Lithology	L
claystone, slate, marl, siltstone	20
sandstone, quarzite, volcanic rock plutonite, metamorphite	15
porous sandstone, porous volcanic rock (e.g. tuff)	10
conglomerate, breccia, limestone, dolomitic rock, gypsum rock	5

Fracturing - F

Fracturing	F
non-jointed	25.0
slightly jointed	4.0
moderately jointed, slightly karstified or karst features completely sealed	1.0
moderately karstic or karst features mostly sealed	0.5
strongly fractured or strongly karstified and not sealed	0.3
Epikarst strongly developed, not sealed	0.0
not known	1.0

Thickness of each stratum in [m] - M

Bedrock - B
$B = L \cdot F$

Artesian pressure A
1500 points

Total protective function P_{TS}

$$P_{TS} = \left[T + \left(\sum_{i=1}^{m} S_i \cdot M_i + \sum_{j=1}^{n} B_j \cdot M_j \right) \right] \cdot R + A$$

score P_{TS}	effectiveness of protective cover	**P-factor**	example
0-10	very low	**1**	0-2 m gravel
>10-100	low	**2**	1-10 m sand with gravel
>100-1000	medium	**3**	2-20 m slightly silty sand
>1000-10000	high	**4**	2-20 m clay
>10000	very high	**5**	> 20 m clay

P-map

Fig. 44: Determination of the P factor (tables and formula modified after HÖLTING et al. 1995).

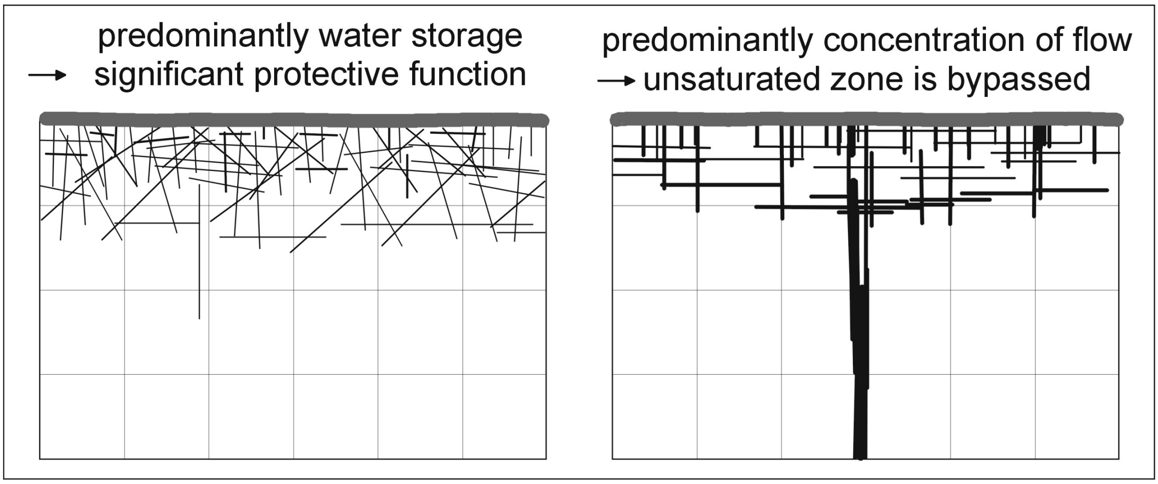

Fig. 45: Epikarst and protective function: a) The unsaturated karst rock (layer 4) may provide a protective function if the epikarst is slightly developed and/or if water storage is the dominant process. b) Concentration of flow in a highly developed epikarst leads to a bypassing of layer 4; this is taken into account by assigning a score of zero to the factor F (for fracturing). Please note: Exokarst features maybe absent in both cases, so that it is difficult to distinguish one from the other.

Surface karst features (exokarst) are only one expression of epikarst, but most of it cannot be seen at the surface. The epikarst zone can be highly developed without any visible karst features. As a consequence, it is assumed that epikarst is present (even if it is not visible) if there are conditions that are favourable for epikarst development, such as pure limestone with widely spaced fractures, or if there are geomorphological indicators of extensive development of epikarst, such as dolines and karrenfields.

It can be misleading to assign a low vulnerability to an area where the aquifer under consideration is overlain by a higher aquifer – in this case, the higher aquifer needs protection. Therefore, the PI method always takes the groundwater table in the uppermost aquifer as the target. As a consequence, a higher aquifer is not considered to be protection for the underlying aquifer, in contrast to the GLA method. Consideration of artesian pressure in the aquifer by an additional score of A = 1500 points was not modified.

The scores for the subsoil and the bedrock are multiplied by the respective thickness in m (factor M). Thin, low permeability strata can be bypassed if they are not laterally extensive, but occur in form of lenses. As a consequence, the lateral continuity of each layer should be taken into account in order to avoid overestimation of the protective function. The score for the total effectiveness of the protective cover P_{TS} is calculated according to a formula similar to the one used in the GLA method (HÖLTING et al. 1995).

The range of possible scores for the total protective function P_{TS} is subdivided into five classes, which are the final P factors in the PI method. Each class covers a score range of one magnitude. The classes are much wider than those in the original GLA method, allowing a better description of the high natural variation of protective cover: $P_{TS} \leq 10$ (e.g., < 2 m of gravel) is considered to provide a very low degree of protection and to be extremely vulnerable (P = 1), while a very high degree of natural protection and a very low vulnerability (P = 5) is assigned to $P_{TS} > 10000$ (e.g., > 20 m of clay). The spatial distribution of the P factor is shown on a P map. For flat areas with a high infiltration capacity, the P factor is multiplied by an I factor of 1. Consequently, the final vulnerability map will be identical to the P map for this area.

A P factor of 5 is assigned to areas outside the considered aquifer from which recharge enters the aquifer by surface and lateral surface or subsurface flow; these areas can be subdivided and classified according to different I values (see next chapter).

1.2.4 Infiltration Conditions (I factor)

1.2.4.1 General concept

The overlying layers can protect the groundwater only if the precipitation infiltrates directly into the ground without significant concentration of flow. However, the disappearance of an intermittent or perennial surface stream into a swallow hole is common in karst areas. In this case, the protective cover is completely bypassed at the swallow hole and bypassed in part by the surface runoff in the catchment area of the sinking stream.

Therefore, the I factor was introduced. It expresses the degree to which the protective cover is bypassed as a result of lateral, surface and subsurface concentration of flow, especially within the catchment area of a sinking stream. If the infiltration occurs directly on a flat surface without significant concentration of flow, the I factor is 1.0, indicating that the protective cover is not bypassed and is 100 % effective. On the other hand, the protective cover is completely bypassed by a swallow hole through which surface water directly enters the karst aquifer. In such a case, the I factor is 0.0. The catchment area of a sinking stream is assigned a value between 0.0 and 1.0 according to the extent of surface and subsurface flow.

It has to be emphasised that the I factor is not precisely defined in terms of hydrology. It is a half-quantitative tool to express the vulnerability of groundwater resulting from bypassing of the protective cover by lateral surface and subsurface flow. The I factor is used for further GIS operations to generate the vulnerability map.

1.2.4.2 Hydrological Basis

The vulnerability of an area to groundwater contamination is dependent on the pathway of a possible contaminant from the ground surface to the groundwater table. As contaminants are usually transported in water, it is necessary to describe the possible flow paths of the water. We can distinguish between three relevant processes: infiltration with subsequent percolation, surface flow, and subsurface flow. Which of these processes predominates depends on both the properties of the site and the characteristics of the rainfall event, as well as the previous precipitation history and the degree of saturation of the soil.

Diffuse infiltration of rain water from the surface into the soil and the subsequent downward percolation through the soil is the dominant hydrological process if the rainfall intensity is less than the capacity of the soil to absorb the water and if the hydraulic conductivity of the total soil profile is high enough to allow downward movement of the water. Gentle slopes, dense vegetation – especially forest cover – and coarse-textured soils with thick organic horizons and stable peds favour infiltration (DYCK & PESCHKE 1995).

Surface flow occurs when not all of the rainwater is able to penetrate the soil surface. There are two main types: Hortonian runoff and saturated surface flow.

Hortonian runoff occurs when the intensity of a rainfall event exceeds the infiltration capacity of the topsoil and the surplus rainwater flows away on the surface. The necessary condition for Hortonian runoff is that the intensity of the rain is significantly higher than the hydraulic conductivity of the topsoil. The amount (depth) of surplus water, which is sufficient to produce surface runoff, is dependent on the slope of ground surface (PESCHKE et al. 1999).

Saturated surface flow occurs when a rainfall event is sufficiently long and intense to saturate the soil and exhaust its throughflow capacity or if the soil was saturated due to previous precipitation and the additional precipitation cannot infiltrate but flows away on the surface. This process is favoured when lower permeability layers are present below thin, relatively highly permeable topsoil. The necessary condition for this type of flow is that the total amount of precipitation is more than the effective porosity; similar to Hortonian runoff, the amount of surplus water that is sufficiently high to produce surface runoff depends on the ground surface gradient (PESCHKE et al. 1999).

Subsurface flow occurs when the hydraulic conductivity of the topsoil is high enough for the infiltration of rainwater while lower permeability layers in or below the soil do not allow the further downward percolation to continue. In this case, the layers above the low permeability zone become temporarily saturated, allowing movement parallel to the slope. The velocity of the subsurface flow is strongly dependent on the slope gradient, the hydraulic conductivity of the topsoil, and on preferential flow paths. We can distinguish between two relevant types:

Subsurface storm water flow in diffuse pathways is a fast flow process, which occurs in very highly permeable soils. The flow velocity depends on the hydraulic conductivity and the slope gradient (ZUIDEMA 1985).

Subsurface storm water flow in preferential pathways is another fast flow process. Soil pipes, desiccation fissures, worm holes and mouse holes are usually dry but become filled with water during intensive rain events, enabling very fast flow (LEHNHARDT 1984).

1.2.4.3 Determination of the I Factor

The I factor expresses the degree to which the protective cover is bypassed by lateral surface and subsurface flow. The spatial distribution of the I factor is shown on the I map. Such flow is considered to be especially dangerous within the catchment area of a sinking stream because contaminants can directly enter the karst groundwater. Therefore, the I factor (the I map) is obtained using the following two components:

The I’ factor expresses the estimated direct infiltration relative to surface and lateral subsurface flow. The controlling factors are soil properties, slope and vegetation. The spatial distribution of the I’ factor is shown on the I’ map.

The ‘surface catchment map’ shows the surface catchment areas of sinking streams disappearing into a swallow hole and buffer zones of 10 m and 100 m on both sides of the sinking streams.

The amount of surface and subsurface flow is dependent on rainfall intensity and site properties. Characteristics of single events, like precipitation rate, cannot be included in the concept of vulnerability – otherwise we would have to draw a different vulnerability map for each rain event. Therefore, the proportion of surface and subsurface flow is estimated only on the basis of the site properties and assuming average storm rainfall, which might occur several times per year.

On the basis of the hydrological concepts described in the previous section, KLUTE (2000) worked out a system to deduce the dominant flow process from the hydraulic conductivity and depth of lower permeability layers within or below the soil (Fig. 46). The critical values for hydraulic conductivity and thickness were calculated using data and theoretical approaches from the hydrological literature, mainly from ZUIDEMA (1985), DYCK & PESCHKE (1995) and PESCHKE et al. (1999):

- Infiltration is the dominant process when the hydraulic conductivity of the topsoil is greater than 10^{-5} m/s and the thickness is more than 100 cm.

- Fast subsurface storm-water flow is the dominant process when the thickness is between 30 and 100 cm and the conductivity is greater than 10^{-5} m/s; if it exceeds 10^{-4} m/s, very fast subsurface flow of more than 50 m/d is to be expected; macropores favour this process.
- Saturated overland flow is the dominant process if we find low permeable layers at depths of less than 30 cm and if the conductivity of the topsoil is greater than 10^{-5} m/s.
- Hortonian flow occurs rarely (rainfall intensity of 30 mm/h on steep slopes and 50 mm/h on gentle slopes) if the conductivity of the topsoil is between 10^{-5} and 10^{-6} m/s.
- Hortonian flow occurs frequently (rainfall intensity of 3 mm/h on steep slopes and 30 mm/h on gentle slopes) if the conductivity of the topsoil is less than 10^{-5} m/s.

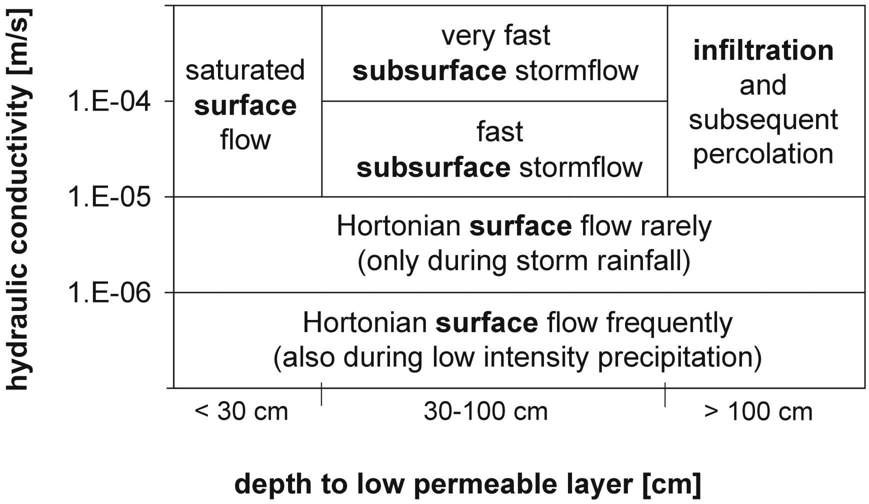

Fig. 46: Determination of the predominant flow process as a function of the saturated hydraulic conductivity and the depth to low permeability layers. If it is not possible to distinguish all the six processes, it is often sufficient to differentiate between infiltration, subsurface flow and surface flow. This can be done on the basis of direct field observations and geological data.

This system makes it possible to delineate areas with different flow processes predominate (KLUTE 2000). However, there are often not enough detailed data to distinguish between the six different processes described above. In this case it is sufficient to differentiate between the three processes infiltration, subsurface flow and surface flow. This can often be done on the basis of geological data, information on the soil type and/or direct field observations. For example: Infiltration has to be expected on highly permeable rendzina soil on karst rocks; subsurface flow predominates on coarse rock debris covering low permeability formations; surface flow takes place on outcrops of marl and claystone formations.

The proportion of each of these flow processes depends on the factors vegetation (land use) and slope of the ground surface. In general, forest cover favours infiltration, whereas agricultural areas are more likely to produce surface runoff. The flow velocity of subsurface flow can be estimated using the Darcy equation (except for preferential flow) and is directly proportional to the slope gradient. Hortonian runoff and saturated flow can occur even on very gentle slopes if the precipitation exceeds infiltration or if the topsoil is saturated, but steep slopes favour surface flow and increase its flow velocity.

A system to assess the proportion of lateral surface and subsurface flow was developed, based on the dominant flow process and the factors vegetation and slope. The slope was done using the divisions of the German soil mapping guidelines (AG-Boden 1996). The proportion of lateral flow is expressed by the so-called I' factor. Its spatial distribution is shown on the I' map. However, for vulnerability mapping in karst areas, it is indispensable to distinguish whether this flow occurs inside or outside the catchment area of a sinking stream as well as to take into account the distance of the evaluated site to the stream. With respect to groundwater vulnerability, the most dangerous situation is lateral flow close to a swallow hole or sinking stream, while the least dangerous situation is flow that leaves the system under consideration without sinking or seeping underground. Therefore, the final I map is obtained by intersection of the I' map with a map showing the catchment areas of sinking streams. Five zones are delineated on this „surface catchment map" in order of decreasing risk (Fig. 47):

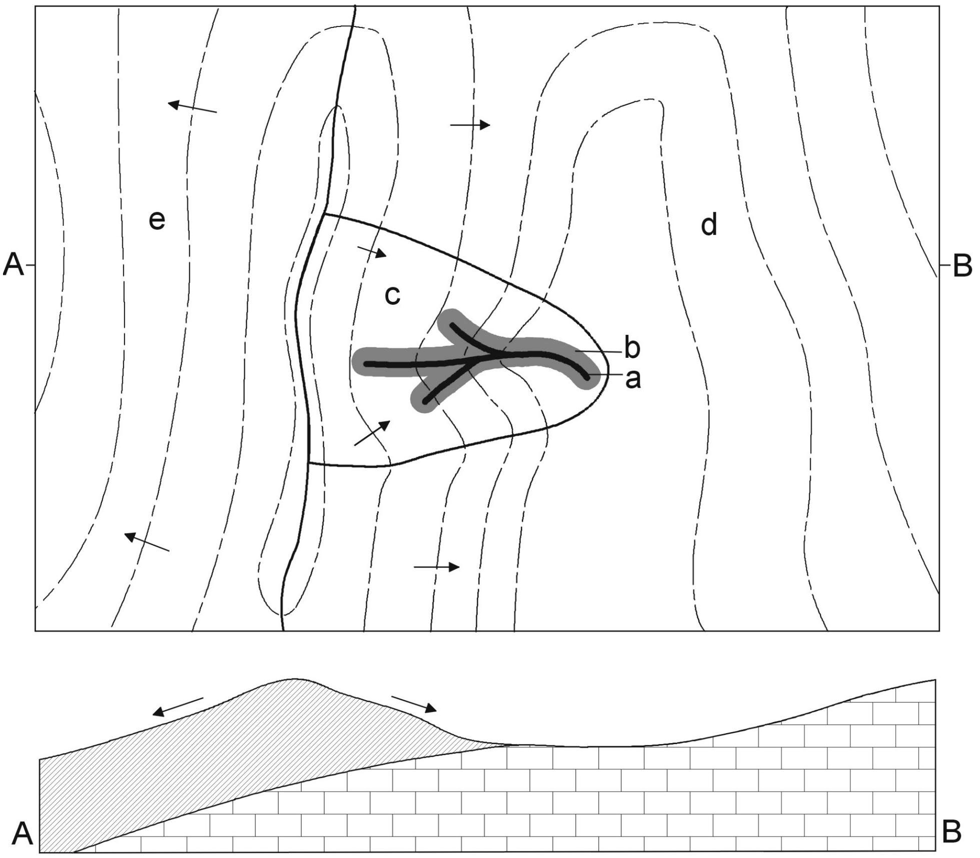

Fig. 47: Topographic sketch and geological profile illustrating the five different zones of the „surface catchment map" in the order of decreasing risk: a) sinking stream with 10 m buffer, b) 100 m buffer, c) catchment of the sinking stream, d) rest of the area discharging into the karst, e) area discharging out of the karst. The arrows indicate lateral surface and subsurface flow components (GOLDSCHEIDER 2002).

a) Swallow holes, the sinking streams and 10 m buffer zones on both sides of these streams.

b) 100 m buffer zones on both sides of the swallow holes and sinking streams.

c) The rest of the surface catchment areas of the sinking streams.

d) Areas outside the catchment of sinking streams but inside the topographic catchment of the (karst) system under consideration; surface and subsurface flow cannot enter a swallow hole but can infiltrate somewhere else, e.g. at the base of a slope or in a closed depression.

e) Areas that discharge by surface or subsurface flow out of the (karst) system under consideration. In that zone, surface and subsurface flow can never reach the groundwater.

The I’ map and the map of the surface catchment area are intersected according to the scheme presented in Fig. 48. The I map shows the degree to which the protective cover is bypassed by lateral surface and subsurface flow.

1st Step: Determination of the dominant flow process

Saturated hydraulic conductivity [m/s]		Depth to low permeability layer < 30 cm	30-100 cm	> 100 cm
	$> 10^{-4}$	Type D	Type C	Type A
	$> 10^{-5}$-10^{-4}		Type B	
	$> 10^{-6}$-10^{-5}	Type E		
	$< 10^{-6}$	Type F		

2nd Step: Determination of the I'-factor

Forest				
dominant flow process		Slope < 3.5 %	3.5 - 27 %	> 27 %
infiltration	Type A	**1.0**	**1.0**	**1.0**
subsurface flow	Type B	1.0	0.8	0.6
	Type C	**1.0**	**0.6**	**0.6**
surface flow	Type D	0.8	0.6	0.4
	Type E	1.0	0.6	0.4
	Type F	**0.8**	**0.4**	**0.2**

Field/Meadow/Pature				
dominant flow process		Slope < 3.5 %	3.5 - 27 %	> 27 %
infiltration	Type A	**1.0**	**1.0**	**0.8**
subsurface flow	Type B	1.0	0.6	0.4
	Type C	**1.0**	**0.4**	**0.2**
surface flow	Type D	0.6	0.4	0.2
	Type E	0.8	0.4	0.2
	Type F	**0.6**	**0.2**	**0.0**

3d Step: Determination of the I-factor

Surface Catchment Map		I' factor 0.0	0.2	0.4	0.6	0.8	1.0
a	swallow hole, sinking stream and 10 m buffer	0.0	0.0	0.0	0.0	0.0	0.0
b	100 m buffer on both sides of sinking stream	0.0	0.2	0.4	0.6	0.8	1.0
c	catchment of sinking stream	0.2	0.4	0.6	0.8	1.0	1.0
d	area discharging inside karst area	0.4	0.6	0.8	1.0	1.0	1.0
e	area discharging out of the karst area	1.0	1.0	1.0	1.0	1.0	1.0

$\downarrow$

I-map

Fig. 48: Calculation and assessment of the I factor. If it is not possible to distinguish six different dominant flow processes, it is sufficient to distinguish between infiltration (white), subsurface flow (light grey) and surface flow (dark grey). In this case, the bold numbers can be used to determine the I’ factor in the 2nd step.

1.2.5 Construction of the Vulnerability Map

The vulnerability map shows the intrinsic vulnerability and the natural protection of the uppermost aquifer. The map shows the spatial distribution of the protection factor π, which is obtained by multiplying the P and I factors:

$$\pi = P \cdot I$$

The π factor ranges between 0.0 and 5.0, with high values representing a high degree of natural protection and low vulnerability. Small maps of the protective cover and the infiltration conditions are also printed as insets on the vulnerability map so that it can be determined whether the vulnerability of a particular area is due to a thin protective cover or to surface and subsurface concentration of flow. The areas on each of the three maps are assigned to one of five classes, symbolised by five colours: from red for high risk to blue for low risk. Consequently, one legend can be used for all three maps (Tab. 19).

Tab. 19: Legend for the vulnerability map, the P and the I map

	vulnerability map vulnerability of groundwater		P-map protective function of overlying layers		I-map degree of bypassing	
	description	π-factor	description	P-factor	description	I-factor
red	extreme	0-1	very low	1	very high	0.0-0.2
orange	high	>1-2	low	2	high	0.4
yellow	moderate	>2-3	moderate	3	moderate	0.6
green	low	>3-4	high	4	low	0.8
blue	very low	>4-5	very high	5	very low	1.0

As the information on the vulnerability map is always for the uppermost aquifer, a thick line indicates graphically the presence of aquifers above the main aquifer under consideration.

Dolines that are too small to be classified using the P and I factors are given special treatment: An extreme vulnerability is assigned both to active ponor dolines and to dry dolines that are not filled by sediments. A high vulnerability is assigned to partially filled dolines. In any case, the existence of dolines serves as an indicator for extensively developed epikarst and for a low degree of protection provided by the unsaturated karst rock. They should be shown on the vulnerability map with the customary symbols.

1.2.6 References

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1.3 The VULK analytical transport model and mapping method

1.3.1 Introduction

VULK is an analytical computer programme, which was developed at the Hydrogeology Centre of Neuchâtel (CHYN) under the scope of COST 620 as a tool for intrinsic vulnerability assessment (Jeannin et al. 2001). The acronym VULK stands for **vul**nerability and **k**arst. The conceptual framework underlying VULK comprises a simple method for transfer time mapping, both for resource and source vulnerability. The computer programme allows for calculating contaminant transport at selected points. Further development aims at coupling VULK with a GIS, so that it will be possible to create attenuation maps showing the maximum concentration of a potential contamination event. It is also foreseen to implementing concentration of flow and specific transport processes into the model.

Jeannin et al. (2001) described the mathematical background of VULK in detail. This section aims at presenting the general idea of this computer programme, as well as its actual and potential future application for intrinsic and specific source and resource vulnerability assessment and mapping, or validation of vulnerability maps that were made using other methods (e.g. PI, COP, LEA, Time-Input; see the respective chapters in this report).

1.3.2 Basic idea of VULK

COST 620 states that vulnerability assessment has to answer three basic questions (see Brouyère, this report). If a pollution occurs somewhere in a catchment,

- How long does it take to reach the target?

- At which concentration will the target be polluted?
- For how long will the target be polluted?

The first two questions are generally more important that the last one. The target may be the groundwater surface (resource vulnerability) or a spring/well (source vulnerability). For intrinsic vulnerability, a conservative contaminant is taken as the reference. However, the same three questions also apply for reactive contaminants and so the concept can also be extended to specific vulnerability assessment.

The three basic questions directly lead to the definition of the criteria to be considered:

- *Transfer time* between the release of contaminants and the arrival at the target
- *Concentration* level at the target
- *Duration* of contamination at the target

The basic idea of VULK is consequently to model the breakthrough curve at a defined target resulting from an instantaneous release of conservative contaminants at a given point (origin) within the system, which is usually located on the land-surface. The calculated breakthrough curve allows determining the transfer time, duration and concentration level of a potential contamination event at the given point. It is thus possible to characterise the groundwater vulnerability at the given point or to validate an existing vulnerability map for this point. In many cases, a vulnerability assessment will only comprise transfer time and concentration, while the duration is a less important criterion.

1.3.3 Model concept

The VULK tool has been developed to simulate mass transport resulting from an instantaneous input of conservative contaminant at a given point (DIRAC-type input function). However, it is also possible to use other input functions, such as continuous contaminant release. The model is based on a 1-D single- or dual-porosity analytical advective-dispersive transport solution for non-reactive steady-flow transient transport. The simulated signal thus corresponds to the travel time distribution of a 1-D section.

All compartments on the pathway between the point of contaminant release (origin) and the discharge point (target) are considered as separate sub-systems (Fig. 49). Up to five sub-systems can be taken into account: The soil, subsoil, non karst rock and unsaturated karst rock are relevant for resource vulnerability assessment, and the saturated zone of the karst aquifer has to be considered additionally for source vulnerability assessment.

Thus, VULK uses the factors O (overlying layers) and K (karst network development) of the European Approach. The concentration of flow (C factor) in the catchment of sinking streams is not considered until present.

The sub-systems are coupled by means of successive convolutions, i.e. the output of one system is the input of the next one. Dilution processes can also be taken into account. Depending on the case to be solved, the user can select the number of sub-systems and chose between a single- and a dual-porosity approach.

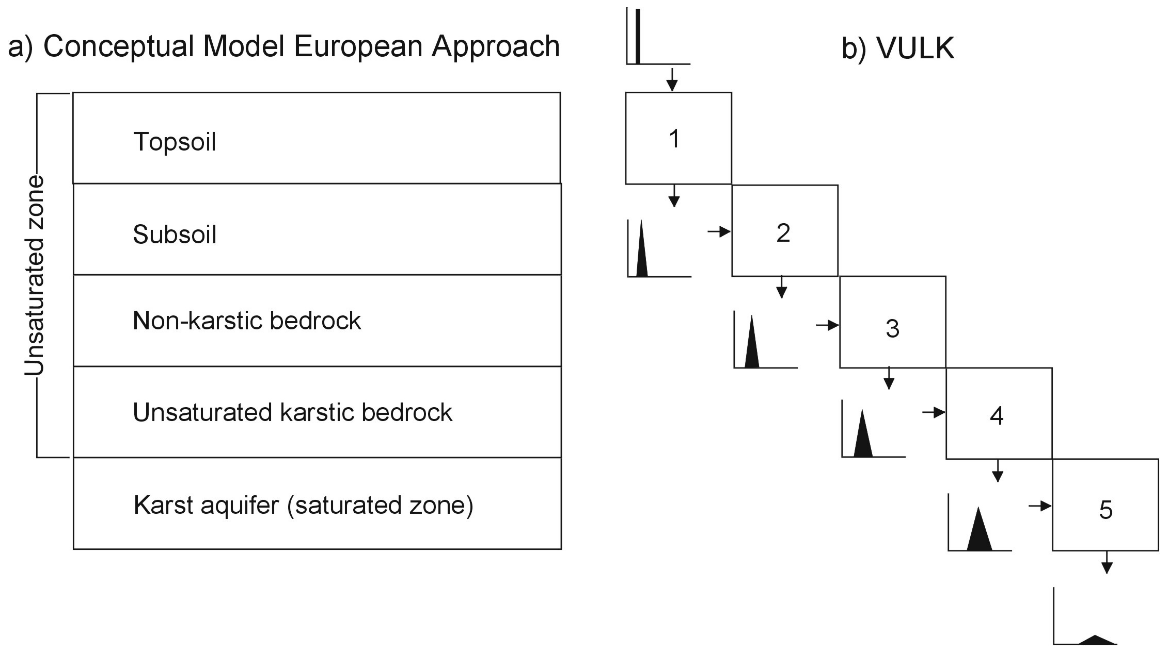

Fig. 49: Conceptual model of the European Approach and principle of the VULK model (Jeannin et al. 2001).

1.3.4 Input data

As a first step, the user has to decide, whether a single or dual porosity approach is to be used, and how many sub-systems are to be modelled (between 1 and 5).

For the single-porosity model, the parameters in each sub-system are: the *flow velocity*, the *flow distance* through the sub-system, the *coefficient of longitudinal dispersivity* of the media, and a *dilution factor*. For the dual-porosity model, the *porosities* of both media and the *exchange coefficient* have to be introduced additionally.

The *flow velocity* can either be measured directly by means of field experiments (tracer tests) or calculated on the basis of the hydraulic properties of the respective sub-system. In fact the hydraulic conductivity (saturated) of the medium is estimated and a hydraulic gradient of 1 is assumed in order to transform the hydraulic conductivity into a Darcy flux. This flux has to be divided by the medium porosity in order to provide an estimate of the flow velocity.

In the overlying layers (sub-systems 1–4), the layer thickness is taken as the *flow distance*; in the saturated part of the aquifer (sub-system 5), the horizontal distance to the spring or well is used (however, the *true flow path length* of a water molecule through the underground is impossible to determine). The layer thickness can be obtained using field methods (geophysical sounding, boreholes), geological maps and sections; the horizontal distance to the spring can simply be taken from a topographic map.

The *dilution* is simply the ratio between the output and the input discharge. For instance it is 0.1 if the input discharge is 10 % of the output one. The multiplications of the dilution factors of the respective sub-systems have to be equal to the ratio between the discharge rate at the pollution input point and the one at the outlet of the system (often a karstic spring).

The *coefficient of dispersivity* is difficult to assess. It is strongly depending on the flow distance and its influence on the breakthrough is not as large as flow velocity or dilution. Then, as a first approximation it is suggested to put it as 5 % of the flow distance.

For the dual-porosity model the *porosity* of the main flow system has to be distinguished from the one of the surrounding matrix where flow is supposed to be only of local significance, i.e.

completely controlled by flow conditions in the adjacent conduit. Porosity of the main system is similar to the one of the single porosity case. Porosity of the matrix has to be evaluated based on our knowledge of the medium (e.g. rock type). The mass transfer between the two porosities is simulated according to the first order exchange kinetic law, with an *exchange coefficient* acting as calibration parameter.

1.3.5 Output data

The output data of VULK are theoretical travel time distributions (unit pulse input) and breakthrough curves (relative concentration input), which allow determining the *transfer time*, the *concentration* and the *duration* of a contamination event. Jeannin et al. (2001) defined mathematical criteria how these values can be determined from the breakthrough curves in a comparable way. The *transfer time* is defined as the period between the times at the centre of gravity of the input function and the centre of gravity of the observed breakthrough curve. The observed maximum *concentration* of the breakthrough curve is normalised by the input concentration. The *duration* can be defined as the breakthrough curve standard deviation.

At present, it is only possible to calculate these criteria (transfer time, concentration and eventually duration) for selected points. Future development aims at coupling the transport model VULK with a GIS, which will allow for the creation of vulnerability maps based on the combination of these three criteria. However, a simple method of transfer time mapping has already been established and is described in the following section.

1.3.6 VULK as a mapping method

The VULK computer programme itself allows calculating the transport of a contaminant that is released at a given point on the land-surface, and, consequently, characterising the groundwater vulnerability at this point. However, the conceptual model underling VULK (Fig. 49) can also directly be used as a method of transfer time mapping, both for the resource and the source.

The methodology of transfer time mapping is shown in Fig. 50. For a resource map, only the four sub-systems of the unsaturated zone are considered (topsoil, subsoil, non karst rock, unsaturated karst). For a source map, the lateral transport in the saturated zone is additionally taken into account. For each sub-system, two layers of information are required: the mean flow velocity [m/s] and the thickness or distance respectively [m]. Dividing distance by velocity for each layer gives the transfer time through this layer, and adding up all the transfer times gives the transfer time of the whole system.

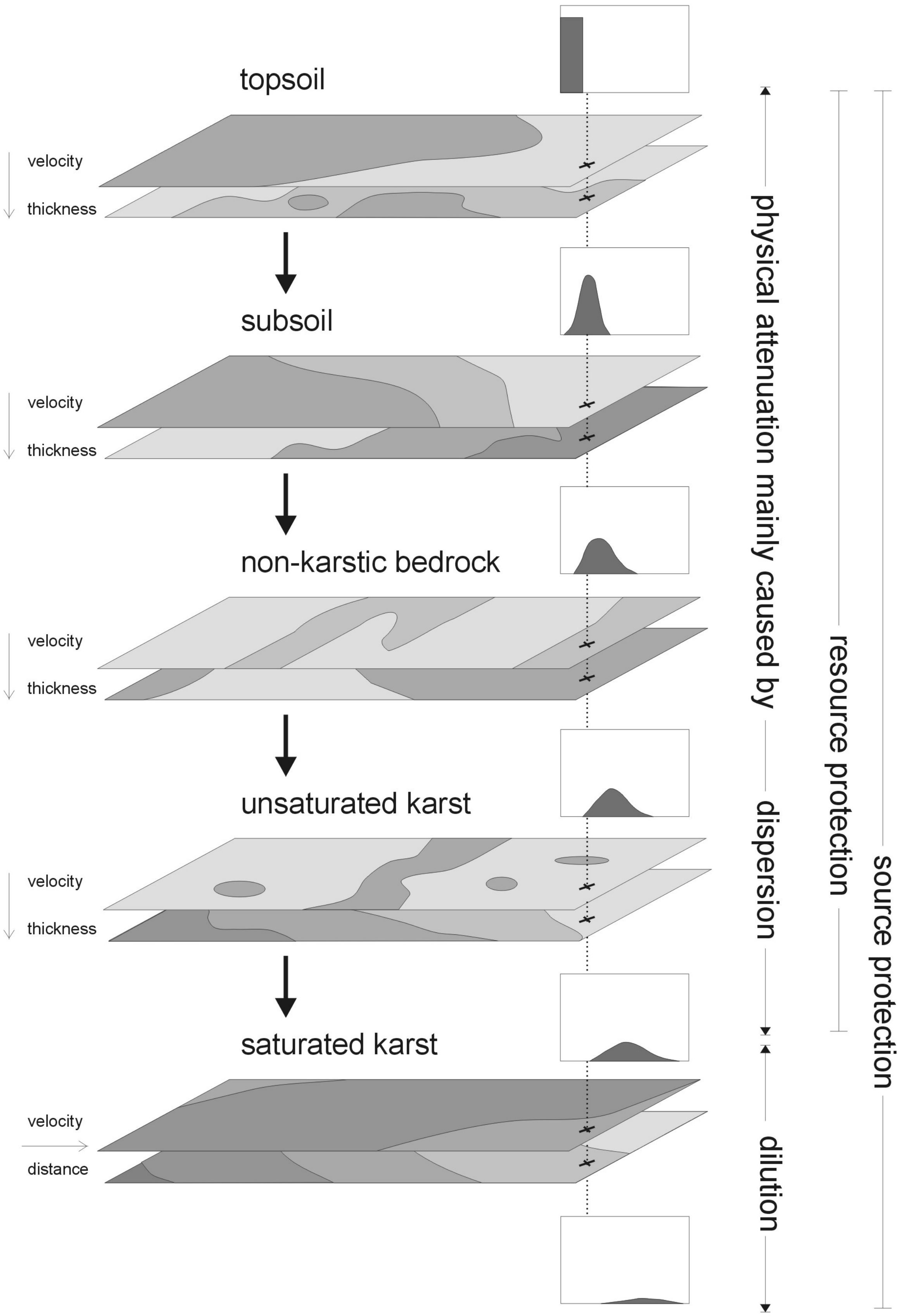

Fig. 50: Transit time mapping and calculation of theoretical contaminant breakthrough curves for selected points (graphic: Michael Sinreich, unpublished).

The mean flow velocity strongly depends on the respective hydrologic conditions, of course. It is suggested to use flow velocities that are likely to occur during average storm rainfall conditions. It has to be stated that the proposed mapping method is based on simplifications and uses data that are difficult to assess in a reliable way for large areas. However, the advantage of this approach is that the map can be validated by means of tracer tests.

In addition to the transfer time maps, it is possible to calculate the contaminant attenuation for selected points in the system using the VULK computer programme (Fig. 50). Plotting the transfer time and the attenuation (output divided by input concentration) in a diagram, allows characterising the groundwater vulnerability at these selected points.

1.3.7 Other possible applications and future developments

The VULK computer programme can also be used for the validation of an existing vulnerability map, which was prepared using another method of vulnerability assessment, e.g. PI, COP, LEA or the Time-Input Method described in this report. The results of the VULK simulation are to be compared with the evaluation of groundwater vulnerability on map. A good correlation between the simulation and the map indicates a high reliability of the vulnerability map – and of the mapping method applied. However, VULK uses strongly simplified assumptions and can thus not be used as the only validation method. The breakthrough curves calculated with VULK should be compared with breakthrough curves obtained from 'real' tracer tests with conservative tracers.

At present, VULK is further developed at the Hydrogeology Centre of Neuchâtel (CHYN) within the framework of two PhD theses, in order to allow for wider applications.

The PhD thesis of Alain Pochon aims at extending and modifying VULK so that it can be used for intrinsic vulnerability mapping in an effective way in all types of hydrogeological environments. Therefore, the VULK programme and mapping method is applied in various test sites and validated by means of tracer tests. VULK is to be coupled with a GIS, so that not only transfer time, but also contaminant attenuation can be determined on area. Another development aims at implementing the concentration of flow (the C factor of the European Approach) into the VULK model. This is indispensable for the application in karst systems that are not only recharged by diffuse vertical infiltration through the soil, but also by sinking streams and/or by concentrated infiltration of lateral flow components (surface or subsurface flow) at the base of slopes.

The PhD thesis of Michael Sinreich aims at taking into account reactive contaminant transport processes, particularly retardation and degradation, so that VULK can also be used for specific vulnerability assessment and mapping. It is also foreseen to use VULK within the framework of groundwater risk assessment. Risk assessment means to evaluate the potential consequences of a contamination event. Any type of contamination scenario can be used as input function for VULK, for example instantaneous or continuous. As the sub-systems are treated separately, it is possible to simulate a contaminant release at any point within the system, for example at the land surface, below the soil zone or within the groundwater body. And it is possible to simulate the consequences of the given contamination event at any point within the system, for example at the groundwater surface or at a spring or well. So VULK appears to be a very flexible tool for groundwater risk assessment.

1.3.8 Reference

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1.4 A Localised European Approach (LEA)

1.4.1 Overview

As the 'European Approach' does not provide detailed assessment schemes but, rather, a framework for intrinsic groundwater vulnerability mapping, it is necessary to propose methods, which are adapted to the local conditions in terms of hydrogeological setting and data available. Such a 'Localised European Approach' (LEA) was developed by Suzanne Dunne, while she was working with Dr. David Drew at Trinity College Dublin during 1998-2002 (Dunne 2003). The method may be used as an alternative to the PI method. It is a straightforward approach that does not use a numerical index. The research was funded by the Environment Agency England and Wales. The field areas were all based in the UK (four in England and two in South Wales) and the research was conducted with the hydrological settings and data availability with the UK in mind. An example is presented in the chapter "applications".

The general concept of this method coincides with the European Approach to intrinsic resource vulnerability mapping, i.e. the main factors that are taken into account are the overlying layers (O factor) and the concentration of flow (C factor).

1.4.2 Overlying Layers (O factor)

The general concept of this parameter is that the longer the pollutant stays in the protective cover layer the greater the chance for attenuation to occur. In the PI method the same concept is used (Goldscheider et al. 2000):

"Most natural processes that decrease contaminant concentration are directly or indirectly related to travel time."

Hence travel times through the protective cover were calculated using the Darcy's flow equation, for a variety of textures and a range of thicknesses. These calculations however led to very long travel times in thin deposits of low permeability and extremely short travel times in coarse grained deposits. In theory the calculations may be accurate for undisturbed pristine deposits of clay. However in reality there are factors such as wormholes, cracks and rootlets may form macropores. Macropores may enable recharge to effectively bypass the protective cover. Hence it was decided that a minimum thickness of 1 m should be used in order to take account of these preferential flow paths. As only a portion of the porosity is used to transport recharge, it was decided that the specific yield should be used instead of the porosity; this resulted in more realistic travel times.

A summary matrix is presented in Tab. 20, which relates the protective cover classes to vulnerability classes.

Tab. 20: Protective cover related to vulnerability and transit time classes.

Protective Cover	Vulnerability	Estimated Transit Time
Bare karstic rock, very thin or	Extreme	Instantaneous - hours
Soil/deposit <3m thick	Very high	Hours - < 1 Day
Silt 1-3m thick (below 3m deposit), fine	High	Days - <1 Month
Clay >1m, silt >4m dependent on	Moderate	>1 Month
Clay >2m, silt >40m dependent on	Low	>1 Year
Clay >12m thick dependent on specific	Very low	10 Years

1.4.3 Flow Concentration (C factor)

Within the catchment areas of sinking streams and dolines in a karstic area runoff may form much of the concentrated recharge. Hence source areas of runoff need to be identified for the purpose of vulnerability mapping. Also it is worth noting that the role of the protective cover is somewhat reversed inside such catchments. For example while the groundwater below a thick low permeability deposit will be well protected, it is possible that the deposit will generate runoff that will enter the aquifer at a swallow hole.

Sinking streams represent extremely vulnerable zones in a karst setting. Runoff from the catchment of a sinking stream may entrain pollutants and flush them directly into the aquifer. There is a permanent and direct connection between areas along the path of the stream and the swallow hole. The swallow hole acts as an open window into the karst system.

Dolines may have steep or gentle sides. Irrespective of the form of the doline, it is an enclosed depression that concentrates recharge from its catchment into a central point. Dolines are also likely to connect recharge with the karst network (Ford and Williams, 1989).

The maximum area that can contribute runoff to a stream is the topographic catchment. However the area that frequently contributes to the stream is usually much smaller than this (Betson, 1964). Betson found that runoff usually originates from a small, but relatively consistent part of the watershed, this is known as the contributing area.

The factors that influence the generation of runoff consist of the topography, vegetation, soil, recharge and the spatial relationship of the parameters. The area immediately around the stream tends to be in good hydraulic connection with the stream (Finlayson, 1977, Dunne et al. 1975). Vegetation and soil profiles may be used as a surrogate for estimating contributing areas. It was concluded that concerning the slope, a 10 degree slope appears to be significant, above which runoff is controlled by the topography. Concerning the soils, the SPR (standard percentage runoff) could be used as a measure of the likelihood that the soil will shed potential recharge.

Within the HOST (Hydrology of Soil Types) classification system, there exists a series of "physical response models" that attempt to describe the pathway taken by rainfall, i.e. whether it's a dominantly vertical one or lateral (Boormand, Hollis, Lilly, 1995). The classification system derives a Standard Percentage Runoff value for each of the 29 HOST classes. Standard Percentage runoff is that percentage of rainfall that causes the short-term increase in flow seen at the outlet.

As contributing areas expand and contract within and between rainfall events an average typical/extreme event is hypothesised. As stated in the PI description if one were to really take into account the variability of rainfall events a different vulnerability map would have to be constructed for each event – which is a worthless and impossible task.

The vulnerability of surface karst features is of prime importance in a limestone setting. Hence zones of extreme vulnerability should be constructed around dolines and sinking streams. A doline and its recharge area or 50 m buffer zone around the outside of the doline could be classed as extreme vulnerability. Around sinking streams a 10 m buffer zone around the stream should be delineated. The zone should be widened where slopes of more than 10 degrees and of HOST class of more than 50 % SPR drain towards the stream.

1.4.4 Combination of the Two Parameters

Unlike many other vulnerability methods existed. The proposed method does not use a numerical index. Where surface karst features are encountered the vulnerability rating simply over-rides the vulnerability rating that the protective cover layer gives it. These areas (with karst features) are classified as zones of extreme vulnerability; in all other areas, vulnerability only depends on the overlying layer.

1.4.5 Validation by Means of Conductivity Data

By measuring the conductivity of a spring the conductivity of the whole hydrological system is being monitored. The effect of the aquifer on water chemistry is seen by a translation of an input of low conductivity water into an output of higher conductivity water. Hence conductivity may be graphed alongside rainfall to see the influence of rainfall events ad the reaction of the karst system to these events.

There are some basic assumptions underlying the interpretation of conductivity data from karst springs:

- Rapid and frequent changes in conductivity may be indicative of sinking streams recharging the aquifer.
- A less rapid and less flashy response of the conductivity may be interpreted as representing a covered karst
- The variance is a measure of the variation of the samples from the mean. A flashier conductivity regime would have a higher variance than a more consistent conductivity.

1.4.6 References

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LandSim release one, manual for release two – Environment Agency R&D Publication 120.

1.5 The COP method

1.5.1 Background

The European Approach (Daly et al. 2002 and this report) is a general conceptual framework for groundwater vulnerability mapping. It highlights the necessity of developing methods, which allow for the flexible application in different European regions and with different availability of data, time and money.

Vulnerability maps, conceived as an environmental management tool, should be both practical and useful. Many discussions took place on this subject within the Hydrogeology Group of the University of Malaga (GHUMA), a participant in the "Intrinsic Vulnerability" Working Group of Action COST 620. These investigations enabled the selection of the variables, parameters and factors to be used as well as the sources and level of detail in the preliminary data. The quantification system and category for each variable were also established, together with the combination and weighting of variables, required to obtain the factors, and the cartographic scale for results agreed. This work was supported by the Spanish Ministry of Re-

search and Science (PB98-1397 and REN2002-01797/HID Projects) and the Research Groups of the Junta de Andalucía (RNM-139 and RNM-308).

The O, C and P factors of the European Approach have been quantified and categorised, taking into account the aforementioned work and their combination and weighting determined. The definition of vulnerability ranges has been established in order to develop the COP method, which can be used for intrinsic vulnerability mapping (VÍAS et al., 2002). The proposed method has been tested on two carbonate aquifer pilot sites in Southern Spain (VÍAS et al., 2002; ANDREO et al., 2002; see also part B in this report). The Líbar and Torremolinos pilot sites have different climatological, hydrogeological and geological (lithological, tectonic and geomorphological) characteristics. Both aquifers had been previously investigated using other vulnerability mapping methods (VÍAS, 2000; VÍAS et al., 2001; LONGO et al., 2001; BRECHENMACHER, 2002). Thus, the application of the COP method to these aquifers enabled the checking of the procedures and the vulnerability assessment. It was also possible to compare the vulnerability map with those obtained by using other methods.

The proposed method presented in this report is just one possible way in which the European Approach (DALY et al. 2002, and this report) may be applied, and is a first step developed within COST Action 620. In subsequent steps, the method will be improved after its application to other karst aquifers in Southern Spain. The vulnerability maps will be validated by tracer tests and by hydrogeological tools (hydrodynamics, hydrochemistry and isotopes).

1.5.2 General characteristics of the proposed method

The COP method is based on three factors (flow Concentration, Overlying layers and Precipitation), as is the European Approach for assessing the intrinsic vulnerability of groundwater resources (DALY et al. 2002, and this report). To assess the vulnerability of the source an additional factor (K = Karst network) of the European Approach must be considered.

COP is an acronym derived from the initials of the vulnerability factors. Two conditions are necessary to assess intrinsic vulnerability by the COP method: firstly, that the contaminant depends on the characteristics of the water to move through the aquifer, and secondly, that the contaminant infiltrates from the surface by means of rainfall.

The O factor refers to the protection of the unsaturated zone of the aquifer against a contaminant event. It indicates the capability of the unsaturated zone, by means of various processes, to filter out or attenuate contamination and thus reduce its adverse effects.

The C and P factors are used as modifiers that correct the degree of protection provided by the overlying layers (O factor). The C factor takes into account the surface conditions that control water flowing towards zones of rapid infiltration. These zones have less capacity to attenuate contamination because of the shorter transit time of contaminants to reach the saturated part of the aquifer. The C factor varies between 0 and 1. Thus, the protection capacity of groundwater, as described by Factor O, may be zero when Factor C = 0 and may be the same as that given by Factor O when Factor C = 1.

The O and C factors refer to the medium in which transport takes place. The P factor considers the characteristics of the agent (water) that transports the contaminants through the unsaturated zone. The influence of precipitation on vulnerability is not as great as that of the flow concentration, and so the value of the P factor ranges from 0.4 to 1. Thus, the aquifer protection capacity (Factor O) will fall by half when Factor P = 0.4, and will remain unchanged when it is equal to 1. The main significance of the P factor is that it is a parameter that was developed to differentiate zones with widely varying rates of rainfall, such as occurs in the continent of Europe.

Both factors P and O can be used to evaluate the vulnerability to contamination of any type of aquifer, while the C factor is specific to karst aquifers.

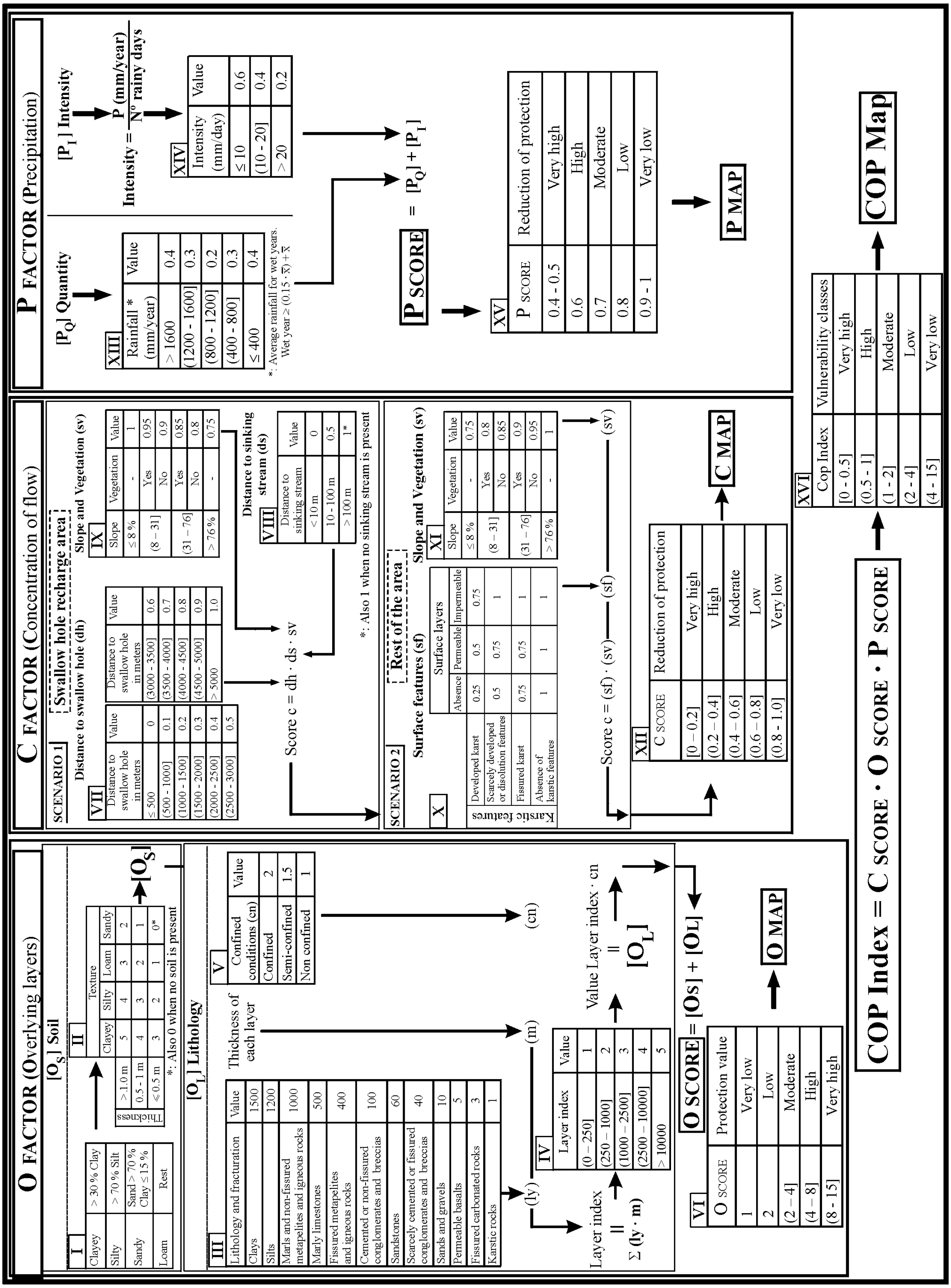

Fig. 51: Diagram of the COP method, showing the differentiation of the C, O and P factors.

Five intervals have been established to evaluate these three factors, so as to distinguish relative degrees of groundwater protection. The final vulnerability index is obtained by multiplying the three factors and from the resulting index five classes of vulnerability are obtained, ranging from Very Low to Very High.

The COP method is summarised in Fig. 51. It shows the different values assigned to the variables evaluated for each factor, the procedure used to calculate the C, O and P factors, and the final vulnerability index.

1.5.3 O factor (Overlying layers)

Following the European Approach (DALY et al. 2002, and this report), the O factor takes into account the protective function of the unsaturated zone and the properties of its layers to attenuate potential contamination. Daly et al. (2002) propose to subdivide the unsaturated zone into the following four layers:

- Topsoil: the biologically active zone; the soil in pedological terms.
- Subsoil: non-consolidated material, such as sand, gravel and clay.
- Non-karst rocks such as sandstone and shale.
- Karstified rocks in the unsaturated zone.

In the proposed COP method, however, only two layers are used instead of four in order to evaluate the O factor, each related to layers in the unsaturated zone with a different hydrogeological behaviour:

- *Soil* [O_S]
- *Lithology* of the unsaturated zone [O_L]

The **soil subfactor** [O_S] deals with the biologically active part of the unsaturated zone. One of the main features that characterises this zone of an aquifer is the self-cleaning process. Two parameters or variables enable an evaluation of the sub-factor soil [O_S]: texture and thickness.

Texture depends mainly on grain size distribution, and so four classes were adopted (Fig. 51-table I) following the classification proposed in Ferreras and Fidalgo (1991).

Soil thickness is a highly variable parameter, especially in Mediterranean areas. In the south of Spain, soils are mapped at a small (low-detail) scale, with little indication as to their thickness. In order to facilitate the method and for ease of recognition in the field, three classes are distinguished: 0-0.5 m, 0.5-1 m, and more than 1 m.

The soil sub-factor [O_S] is evaluated using the textural class and the thickness of the soil profile (Fig. 51-table II).

The **lithology subfactor** [O_L] reflects the attenuation capacity of each layer of the unsaturated zone. For its quantification, three parameters were adopted:

- Lithology and fracturation (ly)
- Thickness of each layer (m)
- Degree to which an aquifer is confined(cn)

Values for “ly” are given in Fig. 51-table III. The assessment criteria are the type of rock (which determines its hydrogeological characteristics, mainly effective porosity and hydraulic conductivity) and the degree of fracturation. The measured thickness in the field (in metres) is the value adopted to quantify “m”. Successive summing of the product of the thickness and the lithology characteristics enables the calculation of the protection index (the Layers index in equation 1); this value rises with increasing numbers of layers in the unsaturated zone and with increasing thickness. This concept of adding layers is based on the AVI method (VAN STEMPVOORT et al., 1993) and on the PI method (GOLDSCHEIDER et al., 2000, and this report).

$$\text{Layers index} = [\sum (\text{ly} \cdot \text{m})] \quad [1]$$

The Layers index ranges between a minimum value of 0 and a maximum value that varies depending on the number and thickness of layers in the aquifer. The Layers index has been divided into 5 intervals (Fig. 1-table IV), and a protection value has been assigned to limit the maximum value of this parameter. In the AVI and PI methods, a 10-order geometric progression is used to establish the protective index ("vulnerability classes" equals orders of magnitude). However, this type of classification reduces the variations in the protection of the aquifer, especially when the present proposal of vulnerability assessment is used. Therefore, Fig. 51-table IV contains narrower intervals, and thus the capacity to differentiate protection zones is more detailed for medium situations, while PI more accurately shows extreme vulnerability.

The "cn" parameter is considered in the same way as in the GOD method (FOSTER, 1987) and the PI method (GOLDSCHEIDER et al., 2000). Both of these methods establish variations in the degree of protection based on the degree to which an aquifer is confined. In the COP method, the "cn" parameter is a weighting coefficient for the classes of the layers index [2]. The values assigned to the "cn" parameter (Fig. 51-table V) show a confined aquifer as more highly protected, whereas an unconfined aquifer is not affected by this parameter (cn = 1).

The final value of the O_L subfactor is obtained by the following expression [2]:

$$O_L = \text{Layers index} \cdot \text{cn} \quad [2]$$

The subfactor O_L ranges from a value of 1, equivalent to a minimum degree of protection, to a maximum value of 10. The protection factor O is obtained by combining the subfactors *soil* [O_S] and *lithology* [O_L]:

$$\text{O factor} = [\text{OS}] + [\text{OL}] \quad [3]$$

The value of the O factor ranges between 1 and 15. Thus, 1 means the lowest degree of protection and 15 represents the highest. Because the attenuation capacity constantly increases with the sum of the protective layers, the values of the protective factor have been grouped into five classes, following an almost geometric 2-order progression (Fig. 51-table VI). The Very Low and Low values correspond to areas where carbonate materials outcrop and where the soil is poorly developed or absent. Values described as Moderate and High refer to areas where the degree of protection is high, both because of the presence of soil and because of low-permeability lithologies. Very High values characterise the sectors of the aquifer that are confined (the O factor is higher than 10 for a confined aquifer).

1.5.4 C factor (flow Concentration)

According to DALY et al. (2002), the C factor is taken as a corrector coefficient of the O factor (Overlying layers). The C factor represents the degree of concentration of the water flow towards karstic features that are directly connected with the saturated zone. It thus also represents how the protection capacity is reduced in such areas.

The present proposal to assess the C factor is based on the PI method (GOLDSCHEIDER et al., 2000) and the EPIK method (DOERFLIGER, 1996). Two scenarios can be differentiated: 1) the catchment area of a stream sinking through a swallow hole and 2) the rest of the area. In the first case, all the water circulating through the catchment area is considered to flow towards the swallow hole. The values of the C factor depend on the distance to the swallow hole and to the sinking stream, as well as on the topographic characteristics (slope and vegetation). In the second case, runoff or infiltration can exist, depending on the slope, the vegetation and, especially, on the characteristics of the surface (karst features and permeability). In both scenarios, slope and vegetation are considered, but in different ways. Thus, in scenario 1, the shallower the slope and the greater the development of vegetation, the smaller is the modification of the value by the O factor and, therefore, the greater the protection. In scenario 2, however, the shallower the slope and the greater the development of vegetation, the less is the protection, as infiltration exceeds runoff.

Two scenarios can be distinguished.

Scenario 1 describes the situation within the catchment area of a swallow hole. The recharge area of a swallow hole is characterised as being covered by a layer of low permeability. Thus, all the water that circulates on the surface flows towards the swallow hole. Under such conditions, evaluation of the C factor requires several variables to be taken into account:

- Distance to swallow hole (d_h)
- Distance to sinking stream (d_s)
- Slope (s)
- Vegetation (v)

It is assumed that $d_h = 0$ in zones located less than 500 m from the swallow hole and that the protection capacity of the aquifer increases as a linear function of the distance. The value of "d_h" follows a 0.1-difference arithmetic progression and the distance follows a 500-difference arithmetic progression (Fig. 51-table VII). The "d_h" parameter establishes the distance from a swallow hole when a Hortonian flow predominates. If this flow type does not occur, the value of "d_h" is 1.

The distance to the sinking stream (d_s) is taken into account, as shown in Fig. 1 - table VIII. Zones less than 10 m from the stream are classified as areas of zero protection; in those up to 100 m away, protection is reduced by half, and at greater distances the protection is unchanged (GOLDSCHEIDER et al. 2000).

The vegetation parameter (v) takes into account the presence or absence of permanent vegetation and its density, which could affect the infiltration-runoff regime. In contrast to vegetation, the slope parameter (s) is positively correlated with the generation of runoff and negatively so with infiltration. In this proposal, four ranges of slope (as a percentage) have been selected and the combination of these with the vegetation provides the value for the slope-vegetation parameter (sv) (Fig. 51-table IX). The values of (sv) show that the greater the slope and the less the vegetation, the lower the protection, and so sv < 1. Water flows more easily to the swallow hole and transit time to the water table is less.

The value of the C factor under these recharge conditions is obtained by multiplying the values of the parameter for slope and vegetation (sv) by those for the distances from the swallow hole (d_h) and from the sinking stream (d_s), as in the following expression:

$$C = dh \cdot ds \cdot sv \qquad [4]$$

Scenario 2 describes the situation in the rest of the area. In areas where the aquifer is not recharged via a swallow hole, the C factor is evaluated by the combination of three variables:

- Surface features (sf)
- Slope (s)

- Vegetation (v)

The *Surface features* parameter (sf) indicates the geomorphological features of carbonate rocks and the presence or absence of a layer (permeable or impermeable) above these materials that determines the importance of runoff and/or infiltration processes. The values for the "sf" parameter are shown in Fig. 51-table X.

In this situation, evaluation of the vegetation and of the slope is carried out in the opposite way to that performed inside the catchment area of a swallow hole. Thus, when the slope is steeper and vegetation is absent, runoff is favoured with respect to infiltration, and so the protection offered by the O factor is not reduced. The values assigned to the "sv" parameter outside the catchment area of a swallow hole are given in Fig. 51-table XI.

To obtain the value of C in scenario 2, the slope-vegetation (sv) parameter is weighted by the surface features (sf) parameter according to the following equation:

$$C = sf \cdot sv \qquad [5]$$

In scenario 1, the C factor ranges between 0 and 1 whilst in scenario 2, the C factor ranges between 0.1875 and 1. This is because the possibility of a potential contaminant reaching the saturated zone of a karst aquifer is higher in the first case. A value of C=0 means that its natural protection in terms of Overlaying layers (O factor) is zero because of the presence of a swallow hole. A value of C=1 characterises an aquifer in which the protection, calculated from the O factor, is not modified.

After obtaining the C values, five classes of protection were established (Fig. 51-table XII). The Very High class corresponds to zones where protection is reduced because of the presence of one or more swallow holes. The High class refers to zones near a swallow hole or sinking stream or to zones that present highly developed karst forms where runoff is nil. The Moderate and Low classes describe zones where karst forms are influenced by surface runoff. The Very Low class is used for sectors of the aquifer where only runoff processes are present or where the distance from a swallow hole or sinking stream is very considerable.

1.5.5 P Factor (Precipitation)

According to DALY et al. (2002) this factor reflects the total quantity of precipitation. It also, includes the frequency, duration and intensity of extreme rainfall events, which can have a major influence on the quantity and rate of infiltration. Thus, the P factor comprises all aspects of precipitation that determine the way contaminants follow the pathway from the surface to the groundwater. The O and C factors provide the ground characteristics affecting the transport of contaminants, whereas the P factor refers to the availability of water to transport the contaminant.

The P factor modifies the protection capacity of the aquifer depending on the quantity and intensity of the rainfall. Thus, a greater capacity of the transport agent (water) to carry contaminants towards the aquifer implies a higher vulnerability.

The P factor is evaluated by two subfactors:

- *Quantity* of Precipitation [P_Q]
- *Intensity* of Precipitation [P_I]

The **Quantity of rain sub-factor** [P_Q] describes the effect of rainfall quantity and the resulting annual net recharge on groundwater vulnerability. There are two aspects, which have to be considered – transit time and dilution. Increasing precipitation height results in decreasing transit time and increasing dilution. However, decreasing transit time means increasing vulnerability, while increasing dilution means decreasing vulnerability.

Mean annual precipitation values are ranged into five intervals (Fig. 51-table XIII), each with a value that depends on the transit time of the potential contaminant and the dilution capacity. These data are mean values of a historical series of wet years, the latter defined as precipitation values that are 15% above average. Such years, therefore, are unfavourable from the point of view of aquifer protection.

Based on experience, theoretical reflections and taking into account the SINTACS (CIVITA and DE MAIO, 1997) and the PI (GOLDSCHEIDER et al., 2000) methods, we consequently assume that the protection given by O factor is poorly modified if precipitation values are below 400 mm/year. An increasing in precipitation, up to 1200 mm, decreases protection, because the transport process (transit time) is more important than the dilution process. When precipitation exceeds 1200 mm/year the dilution of potential contaminant is more important than transit time and, consequently the protection given by O factor is scarcely modified.

The **Intensity subfactor** [P_I] concerns the temporal distribution of precipitation in a certain period of time. To estimate this subfactor, two variables were considered: 1) mean annual precipitation for the wet years and 2) average number of rainy days (in a wet year), as in the following equation [6]:

$$\textit{Mean annual Intensity (mm/day)} = \frac{\textit{Mean annual precipitation (mm)}}{\textit{Mean number of rainy days}} \qquad [6]$$

This subfactor enables a comparison between zones within the continent of Europe, where rainfall and intensity conditions are highly variable. For example, in Mediterranean areas, precipitation is more intense than in central and northern Europe.

The value assigned to the [P_I] subfactor follows the criterion that higher intensity provokes a higher recharge and thus the protection of the resource is reduced. The intensity of rainfall in karst media facilitates the development of a high and, mainly, fast infiltration volume, through fissures or karst conduits. More intense rainfall yields more runoff to those conduits that favour concentrated infiltration. If rainfall intensity is low, more diffuse and slower infiltration occurs, and the probability of other processes, such as evapotranspiration, is higher. This eventually produces a lower recharge volume.

All the above information is contained in the proposed value for the *intensity* subfactor [P_I], in which an increase in the P_I value is associated with a decrease in protection (Fig. 51-table XIV). The *Intensity* intervals with the values in equation [6] are proposed for national or regional use (the Iberian peninsula), since precipitation data for Andalusia revealed no important variations on a local scale.

The final value of the P factor was obtained by adding the values of the two subfactors [P_Q] and [P_I], as in the following equation [7].

$$\text{P score} = [PQ] + [PI] \qquad [7]$$

In Mediterranean karst aquifers, there must be precipitation with a certain degree of intensity for recharge to occur. Thus, the variation in P as a function of [P_Q] follows an arithmetic progression with two units of difference. The values of the P factor, therefore, reflect the greater importance of [P_I] than of [P_Q].

Five classes were established for the values of the P factor (Fig. 51-table XV). These values range between 0.4, the value for minimum protection, and 1, the value for the highest protection. Values closer to 1 indicate that precipitation has little influence on the protection calculated from the O factor. Values closer to 0.4 indicate that precipitation, as a function of quantity and intensity, diminishes the protection of the aquifer and increases its vulnerability. The extreme class intervals have a greater range due to the lower probability of such values occurring. Their representation, therefore, is limited to exceptional situations.

1.5.6 COP vulnerability index

The factors of the COP method have been combined to evaluate intrinsic resource vulnerability, as proposed in the following formula:

$$\text{COP Index} = C \cdot O \cdot P \qquad [8]$$

The final values of the O, C and P factors are multiplied, because each one is considered a factor modifying the protective capacity of karst aquifers (O factor).

The values for the intrinsic vulnerability index range between 0 and 15. Following the proposal by VRBA and ZAPOROZEC (1994), the values for this index are grouped into five vulnerability intervals (Fig. 51-table XVI).

The COP index values are really a modification of the O factor values, consisting of a reclassification of groups and their associated vulnerability. The extreme classes of the O factor have been split and their values reclassified. The interval limits for the Very High and High classes are assigned mainly depending on the influence of the C factor on carbonate rocks, and to a lesser degree on that of the P factor. Those of the Very Low class correspond to zones in which the C and P factors have little influence on protection. The Moderate and Low classes refer to zones where potential protection is low to average, in which the C and P factors do not have a decisive influence on vulnerability (which they do in the High and Very High classes).

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1.6 The Time-Input method

1.6.1 Introduction

The Time-Input method provides a new method to assess groundwater vulnerability especially in mountainous areas based on the European Approach (Daly and others 2002). Its main factors are (1) the travel-time (*TIME*) from the surface to groundwater (about 60%) enhanced by (2) the amount of precipitation input as groundwater recharge (*INPUT*; about 40%). This weighting is somewhat empirical giving the travel-time a slightly higher importance than the groundwater recharge. In contrast to other assessment schemes vulnerability is expressed in real time and input values in real quantities instead of dimensionless numbers. Even these time values are not the exact mean travel-time to groundwater, but its relative numbers have the advantage, that the credibility of results is easier to check and the evaluation process is more transparent (Kralik and Keimel 2003, Fig. 52).

An application of the *TIME –INPUT* method in a forested mountainous dolomite area, the Zöbelboden test site, is presented in this report.

1.6.2 Assessment Scheme

The method is based on three main preconditions:

- For purely intrinsic vulnerability assessments, potential contaminants behave similarly to an ideal tracer and move more or less like the infiltrating water. Specific vulnerability assessments are based on intrinsic vulnerability data, but need further corrections for retardation and decay of specific contaminants. Specific vulnerability assessment is not discussed here.
- The main target of a vulnerability assessment is the surface of the uppermost groundwater body (resource protection). This enables a consistent investigation of the total recharge area. In contrast, for the protection of particular wells or springs (source protection), the distance to the source and the lateral movement in the saturated zone are considered.
- The “mean bad conditions” of a hydrological year are assessed preferentially. In this case, the mean conditions of periods with fairly rapid travel-times and a high input of water are investigated. Extreme events are not considered and would need a special assessment with little chance for actual evaluation in nature.

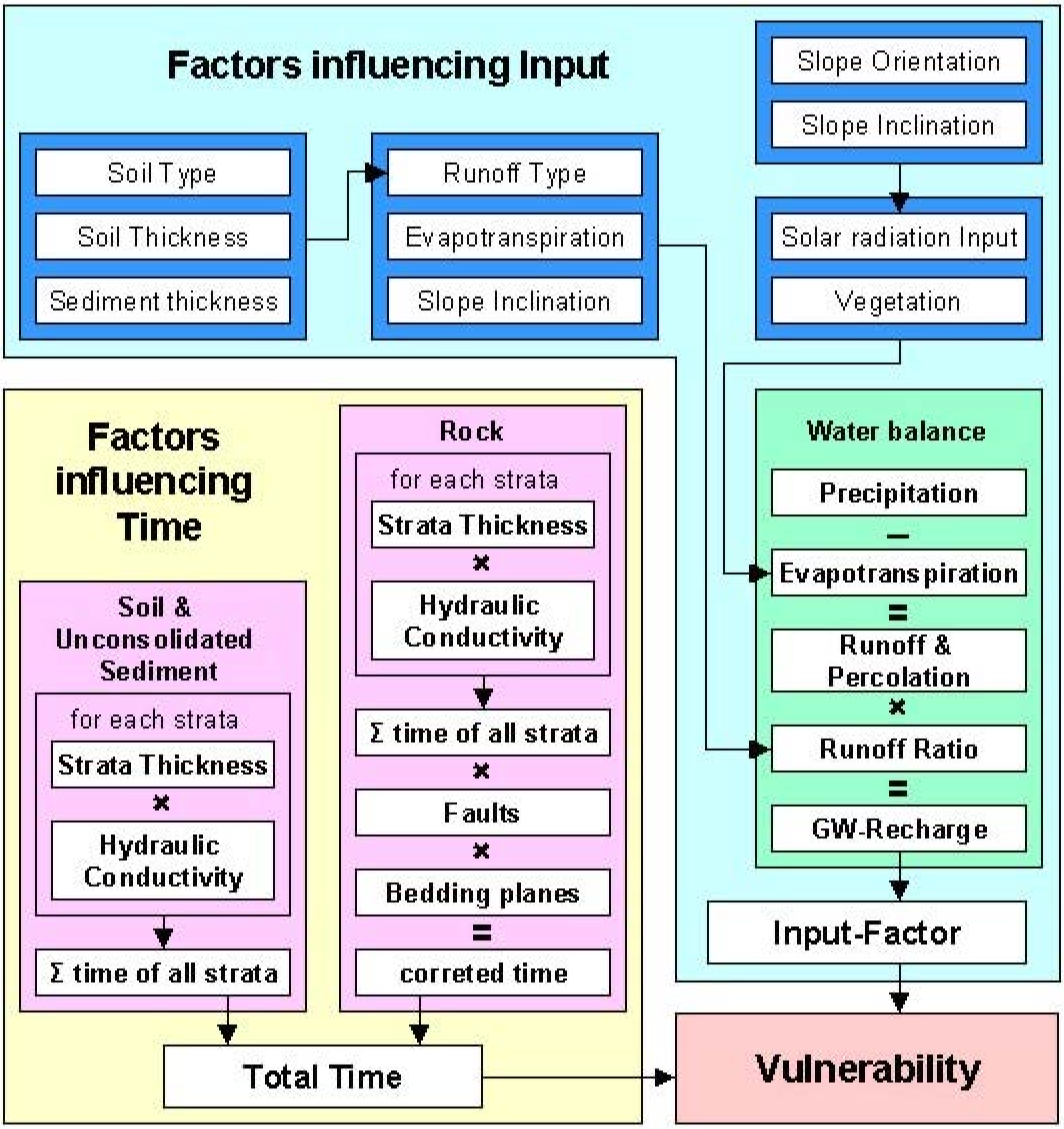

Fig. 52: Flow chart visualising the combination of vulnerability assessment data to the main assessment factors Time and Input.

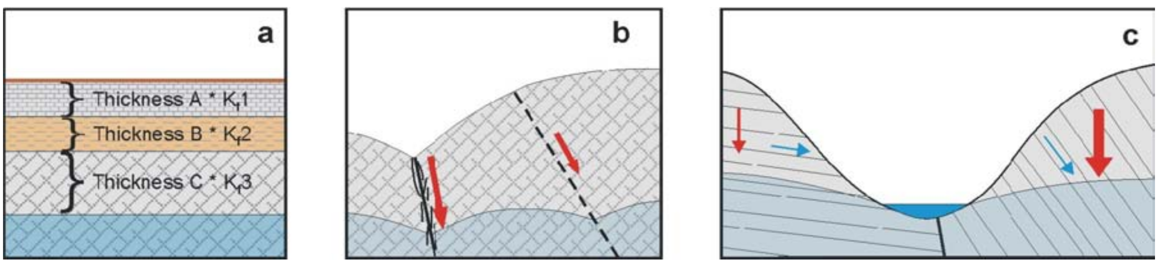

Fig. 53: Factors influencing Time: a) thee sum of the hydraulic conductivity multiplied by the thickness of each strata results in the basic travel-time; b) faults are often the most important factor influencing travel-time; different correction factors should be used for different types and sizes of faults; c) in layered rock, the travel-time is often influenced by bedding planes; its significance depends on degree and type of inclination (towards runoff or towards groundwater).

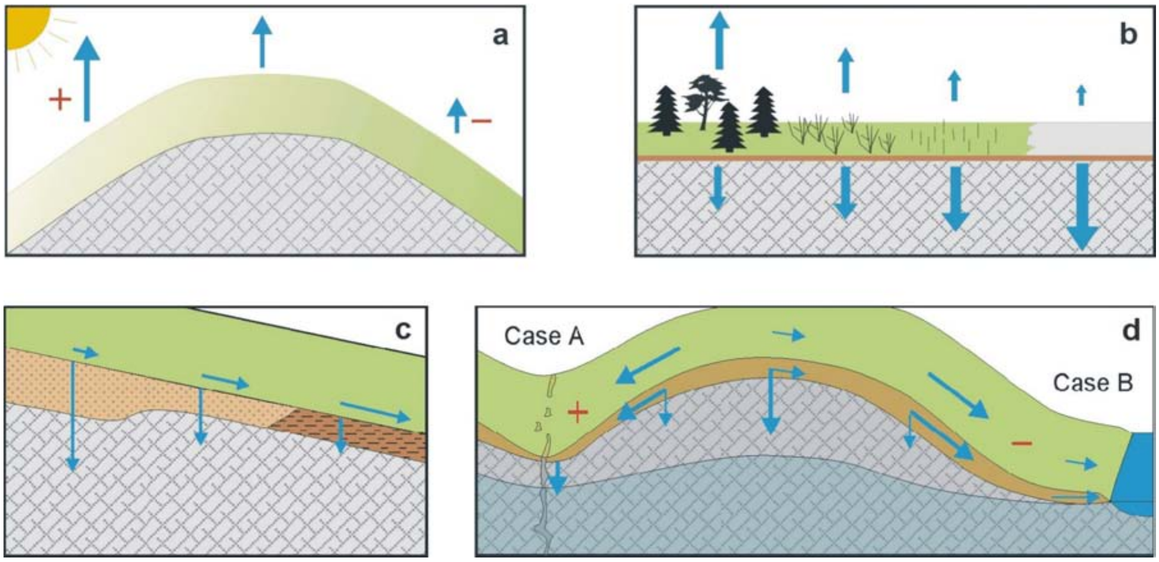

Fig. 54: Factors influencing Input: a) influence of solar radiation-input (determined by slope inclination and slope aspect) on evapotranspiration; b) influence of vegetation type on evapotranspiration; c) soil thickness and soil type and influencing the ratio between runoff and infiltration; d) dependence of slope inclination and catchment area on runoff ratio (surface runoff and interflow vs. infiltration): case A - sinking stream - accumulation of runoff to groundwater recharge; case B - surface water - no accumulation of runoff to groundwater recharge.

The fundamental data for groundwater vulnerability assessment are collected from existing geological, hydrogeological and soil maps, remote sensing data, aerial photographs, field measurements and field observations. All these data are usually stored in a computerised database.

The two main factors travel-time and input are combined by the simple equation:

$$\text{Vulnerability} = \text{TIME [s] x INPUT [f(mm)]} \tag{1}$$

Vulnerability is mainly expressed as travel-*TIME*-classes (measured in seconds [*s*]) modified by the *INPUT*-correction factor (*f*) based on groundwater recharge measured in millimetres per year [*mm*] (Fig. 52).

The basic data required to obtain the first main factor travel-*TIME* (Fig. 53) are the thickness of each layer of the overlying unconsolidated deposits and the different bedrock strata. The unconsolidated deposits, often regarded as protective cover, include soil and subsoil, while the bedrock comprises non-carbonate rocks and the unsaturated zone in one or more carbonate formations (Daly and others 2002). The thickness of soils can be obtained from direct measurements, soil maps and interpolation from measurements of characteristic rounded hilltops, slopes and troughs. The thickness of the rocks and the overlying sediments is evaluated most accurately by boreholes and geophysical measurements, but usually must be estimated from geological maps. Information on faults and karstification has to be obtained by structural and karst morphology mapping. This can be significantly augmented by use of remote sensing and aerial photographs. For each stratum the mean hydraulic conductivity has to be estimated with sufficient resolution. In rock strata and particularly in bedded formations, the hydraulic conductivity will be much enhanced by faults, the inclination of bedding planes towards the groundwater and karstification features like swallow holes and karrenfields.

The main factor *INPUT* (groundwater recharge) is classified as a correction factor similar to factor W in the German vulnerability assessment scheme (Hölting and others 1995). Low recharge quantities have high correction factors thus increasing time, whereas high recharge quantities reduce time and therefore increase vulnerability. This main *INPUT*-factor depends on the amount of precipitation, the solar radiation input, the slope inclination and aspect, the vegetation, the type and thickness of the soil as well as the catchment area (Fig. 54). Correction factors need to be modified according to the climatic zone. The vulnerability is shown in modified time classes. The main factor travel-*TIME* is corrected by the main *INPUT*-factor (groundwater recharge). This has the advantage that the physical parameters *TIME* and *INPUT* can be evaluated separately.

1.6.3 Acquisition of assessment data

Geographical map and digital elevation model: The geographical maps commonly used are in the scale of 1:2.000 to 50.000 and are overlain by a grid of 10–50 m base length. This allows a reasonable resolution and is close to the limit of locating a position in an area, which is partly rugged mountainous terrain. The investigated area thus has about 5,000–20,000 cell units; a number, that can be easily handled with basic computer programs. The slope inclination and aspect of each cell unit can usually calculated by digital elevation models.

Geological map: Usually the geological or hydrogeological maps provide the geological background information. These also provide the basis to estimate the thickness of the layers and to delineate areas with the dip of bedding planes towards and away from the groundwater.

Aerial photographs and satellite images: Digital coloured ortho-photographs, older black and white aerial photographs as well as satellite images are important tools to collect basic information, like distribution of vegetation, soil type, the location and strike of tectonic faults.

Field measurements: In addition to the aforementioned interpretation of remote sensing, at least several days of fieldwork are necessary to obtain additional data and to calibrate and verify remote sensing interpretations. The main soil types have to be identified and measurements of their thickness and the inclination and direction aspect of slopes have to be checked. Estimates of the thickness of sediment (regolith, talus, scree etc.) from erosion trenches, artificial trenches (road cuts) or drill cores have to be gathered from critical points on top of geological mapping information. Finally, additional structural observations and measurements have to be made. The soil type and thickness can be obtained from the mean value of 8-10 penetration tests with a 3 cm diameter soil auger in each basic cell. Usually, aerial soil information will be obtained by assigning typical morphologies such as, hilltops, plateaux, depressions, trenches, steep and gentle slopes particular soil assemblages. This morphological information are obtained from geographical maps and aerial photographs.

Morphological features such as slope inclination and aspect as well as structural parameters may be checked and occasionally corrected by simple geological compass-measurements.

1.6.4 QA/QC of each step

The quality control of each step of data acquisition has to be carried out by standard procedures and field documentation (Csuros 1994). The source of the data: existing data and maps, data obtained from aerial photographs or model interpolation, as well as field and laboratory measurements require documentation in the vulnerability database. Similarly, all correction factors and vulnerability calculations must be transparent and reproducible by hydrogeologists and officials using the vulnerability assessment.

1.6.5 Calculating the Intrinsic Vulnerability

1.6.5.1 Travel-TIME

The basic data to calculate the main factor travel-*TIME* (Fig. 52) are the thickness of each stratum of the unsaturated zone. The mean travel-time by vertical infiltration in more or less homogeneous substrata can be calculated by dividing the thickness of the layers by their hydraulic conductivity.

The soil thickness has to be obtained by direct measurements from characteristic rounded hilltops, slopes and troughs and interpolation from aerial photographs. The K-values for the soil and the sediment are based on estimates, grain-size measurements or core measurements. The thickness of the overlying sediments and the bedrock has to be estimated from geological map, geophysical investigations from boreholes and trenches.

Information on faults and karstification should be obtained by structural and karst morphology mapping. This can be considerably augmented by interpretation of structural elements from aerial photographs. For each stratum the mean hydraulic conductivity has to be estimated with sufficient resolution. In bedrock the hydraulic conductivity will be significantly enhanced by faults, inclination of bedding planes towards the groundwater and karst features, like swallow holes and karrenfields. In these locations the mean hydraulic conductivity has to be adapted by an acceleration factor (Tab. 21).

Tab. 21: Correction factors for tectonics (left) and bedding inclination (right)

Classes	**Structures**	**Correction-factors**
1	Major fault- and Deformationzones	20
2	Small fault zones	10

Classes	**Inclination of bedding planes**	**Correction-factors**
1	0-5°	1
2	5-45° off the groundwater	0,5
3	5-45° towards groundwater	3
4	46-90°	4

1.6.5.2 INPUT (groundwater recharge):

The quantitative input to the groundwater is expressed as groundwater recharge in mm/year. It is calculated using a simple water balance equation (Chow and others 1988):

$$\text{Groundwater recharge} = \text{Precipitation} - \text{Surface flow} - \text{Interflow} - \text{Evapotranspiration} \quad (2)$$

Precipitation: In investigation areas with one single rain gauge, usually the mean precipitation value of recent years is assigned to each cell unit. In cases of areas with significant differences (e.g. lee-side or altitude gradient) correction factors can be applied.

Runoff ratio*:* A differentiation has to be made between infiltration to the groundwater and the surface runoff or the surface near interflow, which leaves the investigation area via streams and rivers.

Soil type, soil thickness and slope inclination are the most important factors influencing the runoff ratio. The classification scheme using the hydraulic conductivity of soil and the depth to the uppermost impervious layer as proposed by Goldscheider et al. (2000) offers a pragmatic solution (see Fig. 46 in chapter 1.2 on the PI method in this report). To integrate this complex relationship a table of runoff correction-factors linked to infiltration-type (A-F: Goldscheider and others 2000), slope inclination (<5°, 5-30, >30°) and the three vegetation types (see evapotranspiration) has to be prepared (Tab. 22).

Tab. 22: Correction factors for obtaining the amount of infiltration caused by the dominant flow process, the vegetation and the slope inclination (modified Goldscheider and others 2000).

dominant flow process	<5° slope inclination		5-30° slope inclination		>30° slope inclination	
	forest	other vegetation	forest	other vegetation	forest	other vegetation
Type A	1.0	1.0	1.0	1.0	1.0	0.8
Type B	1.0	1.0	0.8	0.6	0.6	0.4
Type C	1.0	1.0	0.6	0.4	0.4	0.2
Type D	0.8	0.6	0.6	0.4	0.4	0.2
Type E	1.0	0.8	0.6	0.4	0.4	0.2
Type F	0.8	0.6	0.4	0.2	0.2	0

Evapotranspiration depends on slope aspect as well as inclination due to the time and angle of solar radiation. Dyck and Peschke (1995) demonstrated that in mountainous areas evapotranspiration rate could vary up to 100 % depending on a sunny or shaded aspect.

Vegetation types (forest, scrub and grassland, bare rock) showed to have a significant impact on the evapotranspiration rate (Katzensteiner 1999), which are reflected by the use of adequate correction factors (Tab. 23). The percentage of mean evapotranspiration decreases e.g. in low Alpine areas from 35% and 23% to 7%. Therefore, vegetation can be simplified into three classes as forest, scrub and grassland, as well as bare rock.

Tab. 23: Assessment of actual evapotranspiration (ET) according to vegetation in % of the annual precipitation (modified from Katzensteiner 1999).

No.	Vegetation	ET (%)**
1	Forests (3.1)*	35
2	Scrub and grassland (3.2)*	23
3	Bare rock (3.3.2)*	7

* Land-use classes according Corin Landcover (EC 1989)
** (Katzensteiner 1999)

Tab. 24: Correction factors of sun radiation input (α) derived from catchment slope inclination and orientation (modified according Dyck and Peschke 1995).

Slope inclination	Slope orientation				
	N	NE & NW	E & W	SE & SW	S
0-5°	0.97	0.98	1.00	1.01	1.02
6-10°	0.92	0.94	1.01	1.04	1.08
11-20°	0.85	0.89	1.03	1.12	1.19
21-30°	0.77	0.83	1.08	1.25	1.35
31-45°	0.7	0.8	1.17	1.43	1.58
>45°	0.65	0.78	1.25	1.55	1.75

The two main factors travel-time and input are combined by the abovementioned equation (1)

The application of this intrinsic vulnerability assessment scheme in combination with a hazard assessment (see test site Zöbelboden in this report) indicates that fault zones and the lowest parts of slopes closest to the groundwater are the most sensitive areas for groundwater contamination in mountainous areas. Most springs emerge in the latter area as well.

1.6.6 Evaluation of main factors

The main advantage of this vulnerability assessment scheme with real physical values of time and input is that the evaluation of its main factors can be undertaken using different techniques. Such techniques include investigations of the discharge and dynamics (hydrographs) of springs or wells, analysis of isotope or natural tracers, water balance calculations, tracer experiments and model calculations. The simple basic concept of the Time-Input Method makes it very flexible for use in areas with different hydrogeological settings, data sources and scales.

Discharge and chemograph analysis of springs: discharge, temperature, electrical conductivity, pH and water chemistry can be measured periodically at springs and surface waters (small sub-catchments). This allows the identification of sub-catchments with excess or a deficit of the nominal discharge. Likewise, those sub-catchments may be identified with highly variable water composition and rapid travel-times of at least part of the water input (Pinault and others 2001; Kralik 2001).

O-18, Deuterium and Tritium model evaluation: Oxygen-18, Deuterium and Tritium model calculations of different spring or well samples are able to evaluate mean residence times of sub-catchments at mean bad hydrological conditions.

Natural tracers and tracer experiments: Artificial tracer (e.g. fluorescence dyes) or natural tracers as e.g. major ions, TOC, nitrate, microbiology etc. are able to confirm or disprove shorter or longer mean residence times calculated by the TIME-INPUT-method.

Model calculation: Normally just basic balance models are used. However, more sophisticated models could help to validate and improve the vulnerability assessment.

1.6.7 Discussion

The TIME-INPUT-method presented here is intended for use as a quick and practical procedure at different scales. It can be used for detailed studies in small areas like environmental impact assessments or larger areas or groundwater bodies as required in the European Water Framework Directive (Directive 2000/60/EC). The limiting factor is the availability of basic data. A minimum amount of basic information is necessary to warrant confidence in the vulnerability assessment. The method has been tested in a relatively complicated mountainous dolomite karst area (test site Zöbelboden in this report), but can also be applied with minor modifications to porous aquifers. Several critical aspects have to be considered to obtain good quality data in a relatively short time:

Factors influencing travel-time (retention time of the infiltrating water): The main uncertainties are related to the estimation of the thickness and the assigned hydraulic conductivities of the unsaturated zones due to a common lack of geophysical investigations and sufficient boreholes. However, due mapping of the altitudes of spring discharges, the thickness of the unsaturated zone can be estimated, in most cases, with acceptable levels of error.

In many strongly tectonised formations fault zones are responsible for rapid travel-time to groundwater. Only detailed hydrogeological field observations and structural analysis supported by aerial photographs make it possible to analyse these important fault zones.

The classification of the travel-time of infiltration from the land surface to the groundwater surface into several classes certainly indicates tendencies rather than accurate estimates. It could also be grouped into three vulnerability classes (Tab. 25): High (travel-times less than few days), medium (1-4 weeks) and low vulnerability (> months) during mean bad conditions. Theses mean bad conditions could occur during and after a series of major rainfalls, or during snow melt.

Tab. 25: Attribution of total bulk infiltration (travel-times) to time classes

Time-Classes	Time-Intervals	Bulk infiltration times in seconds
1 extreme	<24 hours	<86400
2 high	1 - 7 days	86400 - 604800
3 medium	1-4.5 weeks	604800 - 2592000
4 low	> 1 month	> 2592000

Factors influencing input (groundwater recharge): The 50 x 50 m or lower resolution of some digital elevation models can smooth out morphological structures. Therefore, in areas of steep rock slopes and steep trenches based on aerial photographs, isolines of topographical maps and control measurements should be used to correct the slope inclinations. Water accumulating morphological structures like trenches and small depressions are important for vulnerability assessments, and are obtained with the aid of aerial photographs and field observations. In karst areas, karst-morphological mapping is essential.

Seasonal variation of the water saturation of soils or desiccation cracks in clay-rich soils cannot be considered. Only the mean infiltration conditions based on one hydrological year can be assessed in a seasonally independent vulnerability map.

Dyck and Peschke (1995) showed that in mountainous areas with the same vegetation type, the evapotranspiration rate can vary up to 100% depending on a sunny or shaded aspect.

Particular attention needs to be given to the common situation where runoff or interflow (runoff close to the surface) contributes nearly quantitatively to the groundwater through swallow holes, other karst features or faults bypassing overlying protective layers or leaves the recharge area mainly by surface flow through rivers and tributaries. In areas with extensive karst features a bypassing factor as in the European Method (Daly and others 2002) can easily added to the Time-Input scheme.

The assumption that high input is more vulnerable than low input is based on the premise that higher recharge will more likely wash down larger quantities of contaminants to the groundwater (Tab. 26). In other cases the correction factors must also be modified but in the opposite direction. For example, when high input significantly dilutes a quantitatively limited contaminant below the regulatory limit or toxicity values. These correction factors have to be adapted to the climatic and hydrologic conditions of an investigation area.

Tab. 26: Correction factors for the Input (groundwater recharge by the amount of infiltrating water) for a humid Alpine climate.

GW-recharge by infiltrating waters	Correction Factor Q
0-200 mm	1.50
200-400 mm	1.25
400-600 mm	1.00
600-800 mm	0.75
800-1000 mm	0.50
>1000 mm	0.25

As a second step after the groundwater assessment, a minimum evaluation of the two physical key factors of (1) travel-*TIME* as well as (2) *INPUT* (recharge) has to be performed. They can be very basic or include very time consuming investigations. Like in an iterative process they will modify and improve the correction factors. Hydrology and hydrogeology offer various methods to verify these physical parameters independently.

To extend these intrinsic vulnerability investigations to *specific vulnerability,* often clay minerals, organic matter and carbonate content have to be estimated in the field or measured in the laboratory for each stratum. In combination with hazard assessments the TIME-INPUT vulnerability method can be an integral part of a risk assessment procedure.

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2 Applications

2.1 Overview

The methods of intrinsic and specific vulnerability mapping, hazard and risk mapping proposed by COST Action 620 were applied in various national test sites all over Europe. Fig. 55 shows the location of all test sites within the European karst landscapes, and Tab. 27 gives an overview of the methods that were applied there.

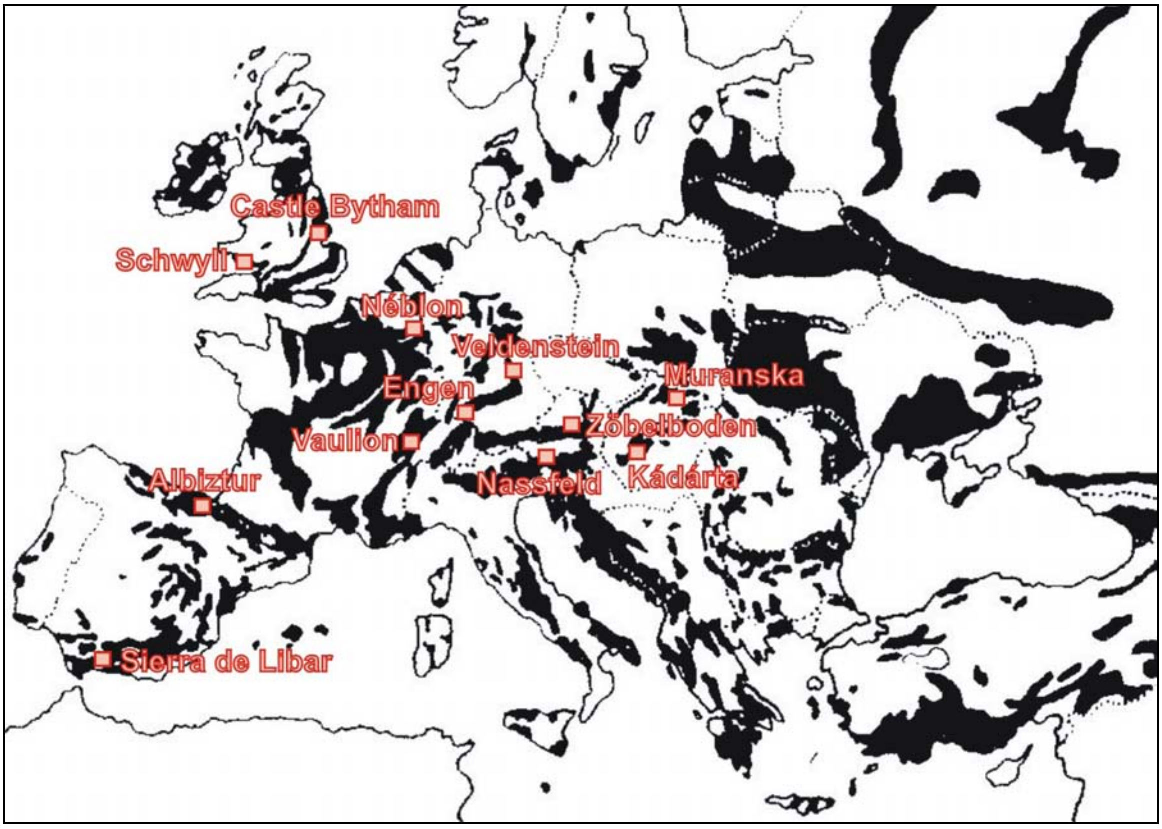

Fig. 55: Karst systems in Europe with location of the COST 620 test sites described in this chapter.

Intrinsic vulnerability mapping was done in all of the test sites. One or more of the different methods developed within the framework of COST 620 (PI, VULK, LEA, COP, Time-Input) were applied. In some of the test sites, the new methods were compared with previously existing methods. Specific vulnerability maps for different types of contaminants (microorganisms, pesticides, nitrate, aromatic hydrocarbons) were prepared for two test sites. Hazard mapping was done in four test sites. In two of these sites, the vulnerability map was combined with the hazard map in order to prepare a risk map.

In the following sections, the most comprehensive examples are presented first. The Sierra de Líbar in Southern Spain is the only test site where all elements proposed by COST 620 were applied – intrinsic and specific vulnerability mapping, hazard and risk mapping. The Engen test site in Germany also represents an instructive example comprising intrinsic vulnerability, hazard and risk mapping. The application of the VULK model and mapping method in Vaulion, Switzerland, resulted in the preparation of physically based intrinsic and specific resource and source vulnerability maps. The two field applications in Austria comprise intrinsic vulnerability mapping combined with hazard mapping.

Then come the examples, where only one of the proposed methods of intrinsic vulnerability mapping (PI, VULK, LEA, COP, Time-Input) was applied. These examples are particularly interesting for people who want to learn about the use and application of the individual methods. The example from the Franconian Alb in Germany shows the application of an interesting GIS supported multiple regression approach to assessing overlying layer thickness.

The last but not least examples are those, which represent the application of preliminary versions of the European Approach. These examples were important for generating discussion within the COST 620 group and show the evolution of our concepts.

Tab. 27: Overview of intrinsic and specific vulnerability mapping, hazard and risk mapping in the different national test sites. PI, VULK, LEA, COP and Time-Input are the methods of intrinsic vulnerability mapping developed within the framework of COST 620. They represent interpretations of the European Approach and are described in this report. The term pEA stands for "preliminary European Approach", a concept that was developed by COST 620 and tested in various sites as a basis for further developments.

test site	country	method of intrinsic vulnerability mapping							specific vulnerability	hazards	risk	validation
		PI	COP	LEA	VULK	Time-Input	pEA	others				
Libar	E	X	X						X	X	X	
Engen	D	X						X		X	X	
Vaulion	CH				X				X			X
Nassfeld	A	X						X		X		
Zöbelboden	A					X				X		
Veldenstein	D	X										
Neblon	B	X										
Albiztur	E	X										
Lincolnshire	GB			X								
Schwyll	GB			X								
Muranska	SK						X					
Kadarta	H						X					

2.2 Sierra de Líbar, Southern Spain

– Comparative application of the PI and COP methods of intrinsic vulnerability mapping, specific vulnerability mapping, hazard and risk mapping –

2.2.1 Background

The Sierra de Líbar is a karst aquifer in Southern Spain that has been used as a test site to apply the methods and concepts developed by the three Working Groups of COST Action 620. The work is the result of cooperation between research groups from three European Universities: Málaga (Spain), Karlsruhe (Germany) and Neuchâtel (Switzerland). Researchers from these countries involved in the three Working Groups of COST Action 620 have collaborated in order to present a complete example of the application of the concepts proposed by this Action. The work of the Hydrogeology Group of the University of Málaga was supported by projects from the Spanish Ministry of Research and Science (PB98-1397 and REN2002-01797/HID) and the Research Groups of the Junta de Andalucía (RNM-139 and RNM-308). The researchers from Neuchâtel and Karlsruhe collaborated within the framework of COST 620 Short Term Scientific Missions and other funds.

In the Sierra de Líbar, intrinsic groundwater vulnerability mapping has been carried out using various methods (LONGO et al. 2001, BRECHENMACHER 2002 and VÍAS et al., 2002). This section only presents the application of the PI and COP method, as these were developed within the framework of COST 620. The PI method was applied by researchers from Karlsruhe (Brechenmacher, Goldscheider and Neukum) in cooperation with the Málaga team (Andreo, Vías,

Vías, Perles, Carrasco, Vadillo and Jiménez). This latter group developed and applied the COP method. Thus, it is possible to compare the results obtained by the two methods.

Specific vulnerability maps were prepared for two potential contaminants that are present in the test site: faecal coliform bacteria and aromatic hydrocarbons. This work was carried out by researchers from Málaga (Vadillo and Vías) and Neuchâtel (Sinreich).

The hazard and risk mapping in the Sierra de Líbar was carried out by colleagues from Karlsruhe (Neukum and Hötzl) and, for the hazard map, Málaga (Jiménez).

2.2.2 Test site characteristics

The Sierra de Líbar covers a surface area of 103 km^2 and is located between the provinces of Málaga and Cádiz, Andalusia, Spain (Fig. 55). The region is characterised by high seasonal rainfall, with a mean precipitation of over 1500 mm per year. Geologically, it is formed of Jurassic dolomites and limestones, and Cretaceous marls and marly limestones. The geological structure comprises a large anticline in a N40E direction, predominantly made of Jurassic limestones; Cretaceous rocks remained in the synclines. This fold structure has been overthrusted by clayey Tertiary Flysch and was subsequently cut by transverse faults. The lithology and geological structure predetermine a landscape characterised by steep slopes and plateau-shaped mountain ridges. There are a large variety of well-developed karst landforms, including karrenfields, vertical shafts and poljes with swallow holes, and caves (DELANNOY 1987).

The aquifer is formed of Jurassic dolomites and limestones with a thickness of over 400 m. Most of the discharge occurs through springs located on the SE border. The data suggest that these springs respond rapidly to precipitation, with sharp changes in flow, and with a karstic behaviour (BENAVENTE & MANGIN 1984, SÁNCHEZ GONZÁLEZ et al. 1998, CARRASCO et al. 2001; JIMÉNEZ et al. 2002). In the Sierra de Líbar, soils overlying limestones are absent, with the exception of poljes developed in synclinal structures where Cretaceous marls and marly-limestones exist.

2.2.3 Intrinsic vulnerability

2.2.3.1 PI method

The PI method (GOLDSCHEIDER et al. 2000) evaluates intrinsic groundwater resource vulnerability as the product of two factors: effectiveness of the Protective cover (P) and Infiltration conditions (I). The evaluation of these parameters as proposed by BRECHENMACHER (2002) for the Sierra de Líbar is shown in Tab. 28.

Tab. 28. Values for PI method variables in Sierra de Líbar site (Brechenmacher 2002).

Factor	Variable			Values
P	Soil		▪ Discontinuous on carbonates and settlements ▪ On marls, marly limestones and talus ▪ On Colluvium, tertiary sediments and mediumd deep soils in poljes ▪ Deep in poljes, dolines and on Keuper	0 50 125 250
	Subsoil		▪ Neonumidic clays, Algeciras unit and doline fillings ▪ Flysch of the internal Subbetic ▪ Talus and colluvium ▪ Sediment fan ▪ Blocks with fine material ▪ Rock fall	400 270 75 60 10 5
	Bedrock	Lithology	▪ Cretaceous marly-limestones ▪ Limestone and dolomites Jurassic and Keuper carniolas	20 5
		Fracturing	▪ Cretaceous marly-limestones ▪ Jurassic limestone and dolomites	4 0
	Recharge		▪ > 400 mm/year	0.75
	Artesian pressure		▪ Unconfined	0
I	Predominant flow process		▪ Infiltration and subsequent percolation ▪ Fast subsurface stormflow ▪ Very fast subsurface stormflow ▪ Overland flow frequently saturated ▪ Karst infiltration ▪ Complex flow processes (settlements)	Depend on slope and vegetation
	Surface catchment area		▪ Swallow hole, sink stream 10 m buffer ▪ 100 m buffer on each side of sink stream ▪ Catchment of sink stream ▪ Area discharging inside karst area ▪ Area discharging outside karst area	Depend on flow process

The evaluation scheme for the P factor of the PI method uses a sub-factor F describing the degree of fracturing and epikarst development. A value of zero is assigned to this factor in case of strongly developed epikarst, which is not sealed by soil and/or clayey sediments. As the Sierra de Líbar is characterised by extreme epikarst development (karren fields, abundant deep vertical shafts, large caves), a value of zero was assigned to the factor F for large areas and the vulnerability was consequently classified to be "Very High", independent of the thickness of the unsaturated karstic bedrock (BRECHENMACHER 2002). A "Very High" vulnerability was also assigned to the inner areas of the poljes where the slope exceeds 3.5 %, as these areas are drained by surface runoff, which sinks via swallow holes into the karst aquifer.

"High" vulnerability can be observed in the inner areas of the poljes of Líbar and Pozuelo, where the slope is less than 3.5%, and where no water fluxes are generated towards the swallow holes. Vulnerability is also "High" in areas where Quaternary materials outcrop, over Jurassic limestones, due to the extreme permeability of these materials (Fig. 56)

The areas of lesser vulnerability mainly correspond to the eastern sector of the system, where the materials (Cretaceous marly limestones) are of very low permeability. The minimum vulnerability values are found in the NE and SW sectors of the aquifer, where the thickness of these materials exceeds 160 m (BRECHENMACHER 2002).

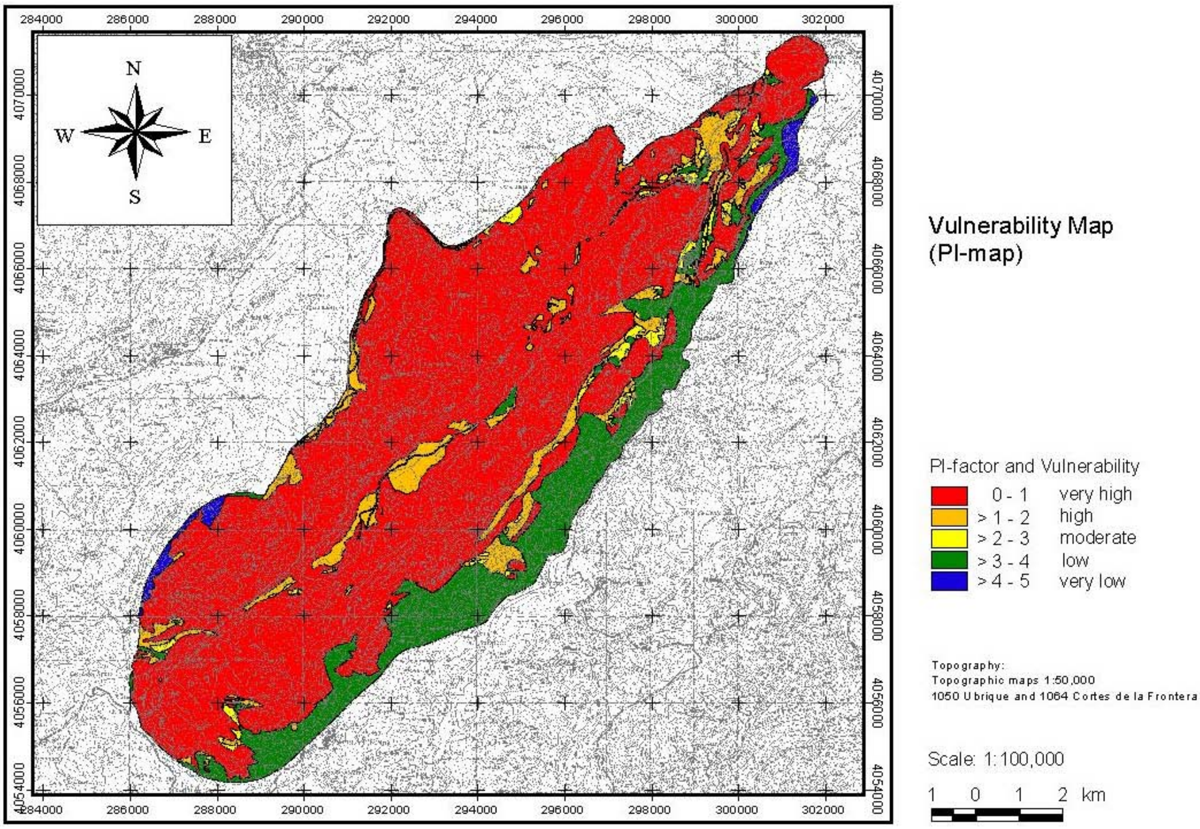

Fig. 56. PI vulnerability map of the Sierra de Líbar (Brechenmacher 2002).

2.2.3.2 COP method

The COP method (VÍAS et al. 2002) also evaluates the vulnerability of the aquifer to contamination, but in this case by means of the product of three factors: Overlying layers (O), which refers to the protective capacity of the unsaturated zone, flow Concentration (C), referring to the capacity of surface water flows to bypass the unsaturated zone, and Precipitation (P), which takes into account the characteristics of the contaminant-transporting agent. The application of the COP method to Sierra de Líbar is summarised in Tab. 29, which shows the values of the different parameters observed, and the scores assigned to each one.

Tab. 29. Values for COP factors and variables in Sierra de Líbar site

Factor	Subfactor	Variable		Values
C	Scenario A: C inside the catchment area of swallow hole	Distance to swallowhole	▪ Between 0 and 3500 meters	0 – 0.6
		Slope and vegetation	▪ 0 – 8 % ▪ 8 – 31 % and without vegetation ▪ 31 – 76 % and without vegetation ▪ > 76 %	1 0.9 0.8 0.75
	Scenario B: C rest of the aquifer	Surface features	▪ Karst well developed and uncovered ▪ Karst well developed covered by a permeable bed ▪ Karst scarcely developed and uncovered ▪ Absence of karst features	0.25 0.5 0.5 1
		Slope and vegetation	▪ 0 – 8 % ▪ 8 – 31 % and without vegetation ▪ 31 – 76 % and without vegetation ▪ > 76 %	0.75 0.85 0.95 1
O	Soils [O_S]	Texture and thickness	▪ Clayey and > 1 m ▪ Clayey and 0.5 - 1 m ▪ Clayey and 0 - 0.5 m ▪ Silty and 0.5 - 1 m	5 4 3 3
	Lithology [O_L]	Lithology and fracturation	▪ Marl Flysch ▪ Cretaceous marly-limestones ▪ Quaternary ▪ Jurassic limestone and dolomites	400 200 10 1
		Confined conditions	▪ Unconfined	1
P	Quantity [P_Q]	Average precipitation for wet years	▪ > 1600 mm/ year ▪ > 1200 and ≤ 1600 mm/year ▪ > 800 and ≤ 1200 mm/ year	0.4 0.3 0.2
	Intensity [P_I]	Precipitation and nº days	▪ > 20 mm/day ▪ > 10 and ≤ 20 mm/ day	0.2 0.4

The COP final map (Fig. 57) shows “High” vulnerability in most of the system, which is related to the high degree of karstification of the carbonate outcrop, which favours rapid infiltration from the surface to the saturated zone.

Within the carbonate outcrop, the differences in vulnerability (between “High” and “Very High” categories) arise from the presence of very specific conditions of factors O or C. Thus, the O factor determines the “Very High” degree of vulnerability in areas where the thickness of the unsaturated zone does not exceed 250 m, as in the case, for example, on the NW boundary of the aquifer. The C factor, on the other hand, is crucial in determining the “Very High” vulnerability in areas where the surface cover favours infiltration processes rather than runoff, for example where karst forms are not covered by an impermeable layer or where karst is not highly developed but the shallowness of the slope and the presence of vegetation favour infiltration. The poljes of the central part of the aquifer, those of Líbar, Pozuelo and Zurraque, are classified as being of “Very High” vulnerability due to the presence of swallow holes that bypass the protective capacity of the unsaturated zone of the aquifer minimizing the transit time of the water through the unsaturated zone.

The areas classified as “High” vulnerability in the outcrop are Jurassic limestones and dolomites, where the thickness of the unsaturated zone exceeds 250 m and the slope is greater than 8%.

In certain areas, the vulnerability is “Moderate” where carbonate outcrops are covered by a layer of soil or by Quaternary materials, i.e. areas with Cretaceous marly-limestones where the C and P factors (because of the shallow slope and the high intensity, respectively) reduce the potential attenuation capability assigned to the unsaturated zone by the O factor.

The “Low” and “Very Low” vulnerability areas correspond to the outcrops of Cretaceous marly-limestones, beyond the poljes, and of marly Flysch, where the low permeability of the surface cover and the gradient of the slope favour runoff towards the external part of the aquifer.

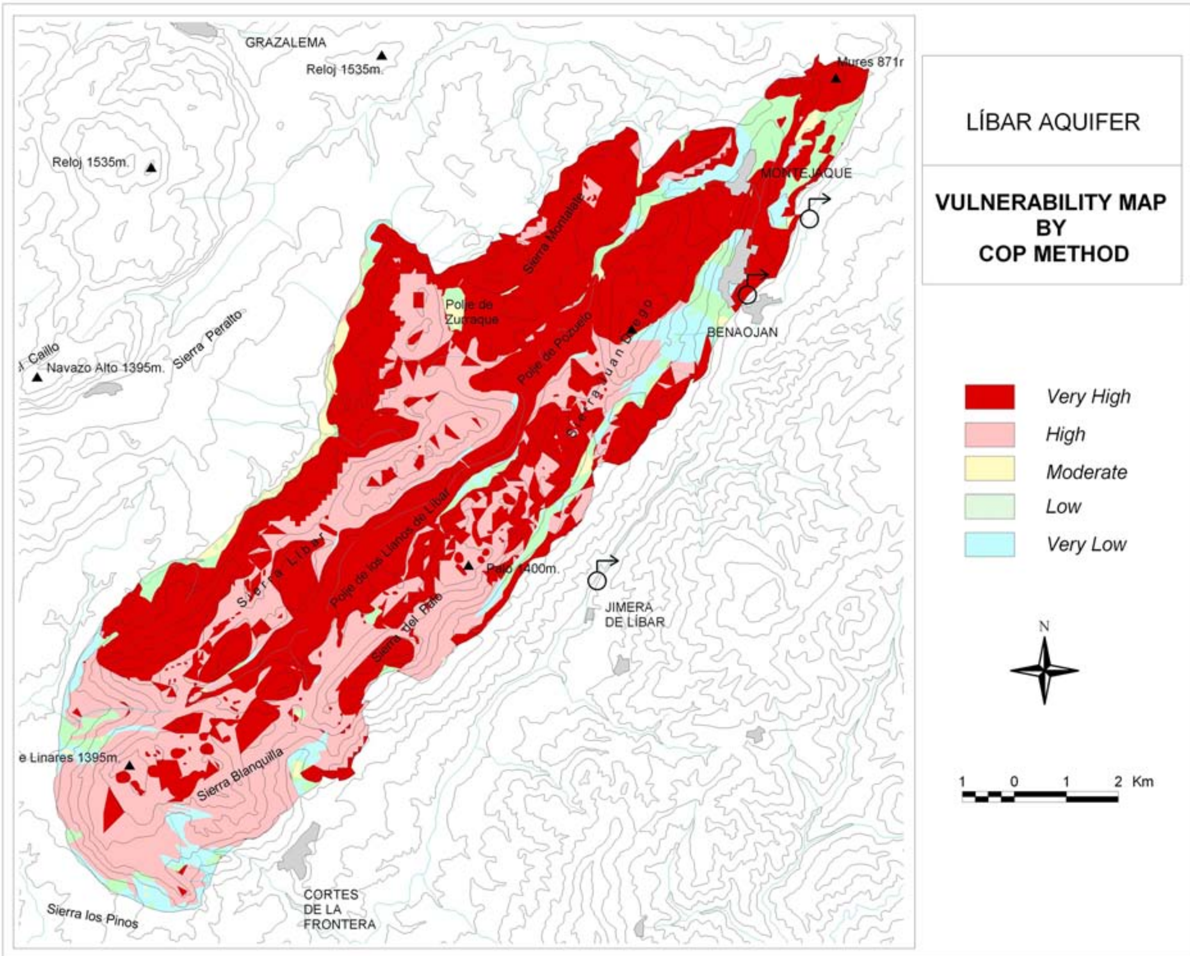

Fig. 57. Vulnerability map by COP method in Sierra de Líbar

2.2.3.3 Comparison between PI and COP vulnerability maps

Both the PI and the COP vulnerability map for the Sierra de Líbar show a “High” to “Very High” vulnerability in the catchments of swallow holes and on outcrops of karstified rocks, while the vulnerability is lower on outcrops of Cretaceous marly limestone and clayey Flysch formations. These evaluations are consistent. However, the two maps show significant differences in detail.

On the PI map, large areas are classified as having “Very High” vulnerability, because of the extremely developed epikarst. The philosophy underlying this classification is that the presence of bare karren fields connected with vertical shafts indicates a very low protective function of the unsaturated karstic bedrock, largely independent from its thickness.

The COP method, on the other hand, does discriminate areas with different degrees of vulnerability within the carbonate rock outcrops. It establishes differences by means of the O factor, which takes into account the thickness of the unsaturated zone (i.e. whether it is less or greater than 250 m), and also by means of the C factor, distinguishing areas with different degrees of surface karst development and with different slope gradients.

With respect to the protective factor described by the two methods (P of the PI method and O of the COP method), approximately the same degree of protection in the outcrops of Cretaceous materials was observed, whilst the degree of protection in the Jurassic materials was greater according to the COP method. Only sectors in which the thickness of the unsaturated zone was less than 250 m were assigned the same degree of protection by the two methods.

In contrast, the differences between the C factor (in the COP method) and the I factor (in the PI method) were more marked in the measurement of the "bypass effect" in the unsaturated zone by the sinking streams. According to the COP method, the vulnerability of the Jurassic carbonate rocks is "High", due to the high degree of karstification present, which produces an infiltration of rainfall through karstic conduits. According to the PI method though there is no surface runoff in these areas, and so the probability of the concentration of flows towards the swallow holes is nil; consequently, there is no bypass of surface flows towards the saturated zone. The presence of swallow holes is recognised in both methods as resulting in zones of high bypass activity, although the differences in the catchment areas (poljes in the central part of the aquifer) correspond to variations in the slope in the case of the PI method, but also to variations in the distance from the swallow hole in the case of the COP method.

The influence of precipitation on vulnerability is a factor that enables us to discriminate between aquifers in different locations, but within the boundaries of a single aquifer, the differences in vulnerability due to precipitation are very slight. In the COP method, by means of the P factor, zones are differentiated depending on the quantity of rainfall and on the intensity of precipitation. In the Líbar aquifer, the highly intense precipitation induces a rapid recharge of the aquifer, which increases its vulnerability. On the other hand, the high volumes of precipitation reduce the negative effect of the intensity, as the dilution effect predominates over that of the transport of contaminants. Thus, the incidence of the P factor only has a moderate effect on the aquifer, and so the protection is reduced by only 40%. The PI method assigns a recharge value of 0.75 to the whole study area, that is, the attenuation capacity of the protective layers is reduced by 25% due to the high recharge volume.

2.2.4 Specific vulnerability

Two groups of contaminants were selected for specific vulnerability mapping in Sierra de Líbar, according to the methodology developed within the framework of COST 620:

- Microbial contaminants (Faecal coliforms), mainly deriving from (1) slaughterhouses, (2) urban sewage waters and (3) animal excrement. They may be discharged in the study area as diffuse contamination in the course of application of manure or as point contamination next to villages and cattle or sheep farms.
- Hydrocarbon contaminants, mainly belonging to the aromatic hydrocarbons (Benzene, Toluene, Ethylbenzene, Xylene). These BTEX can be produced in the petrol station situated within the catchment, as well as in some oil-fuel tanks for houses and petrol tanks for agricultural use. Also the mountain roads on which the petrol is transported constitute a BTEX hazard.

Both of these contaminants, and their respective specific vulnerability maps, illustrate different contaminant behaviour; it being the contaminant properties themselves that are responsible for the differences in the specific vulnerability maps as there is no difference in the intrinsic vulnerability map used in each case.

The S factor, represented by the specific attenuation index, accounts for both layer and contaminant properties including the bypass of protective layers due to preferential flowpaths, and has to be merged with the O factor of the COP method. So, an estimation of the diffuse or slow flow (= total flow – preferential flow) for each layer was firstly done taking into account the geological and geomorphological features of the outcrops and the hydrodynamic and hydrochemical characteristics of groundwater (CARRASCO et al. 2001, JIMÉNEZ et al. 2002). In the Sierra de Líbar aquifer the diffuse flow expressed as a percentage of the total flow was estimated as about 80% in the topsoil, 30% in the fissured non-karstic bedrock and about 10% in the highly karstified Jurassic carbonates.

Following the proposed assessment procedure (SINREICH & ZWAHLEN 2002 and chapter 4.6. in this volume), the layers and contaminant properties, together with the layer thicknesses, provide a specific attenuation index for each point, which is restricted for each layer due to the rate of diffuse flow. The specific attenuation classes ranged from “Very Low” in case of uncovered karst up to “High” where the karst is overlain by topsoil and/or non-karstic bedrock. The specific attenuation index (S factor) is added to the O factor score of the COP method. The modified O factor is then multiplied with the C factor and P factor analogously to the intrinsic vulnerability assessment.

2.2.4.1 Microbiological contamination scenario

Faecal coliforms (e.g. *Escherichia coli*) suffer retardation and degradation in the subsurface by (1) sorption, (2) filtering and (3) die off. They possess a (1) high sorption coefficient (high K_D) that can lead to a significant attenuation within the clayey topsoil layer and also within the marls of the non-karst rock (high clay content). Their particle size (medium) allows (2) filtering processes in the narrow matrix aperture of all layers. Additionally, those bacteria may (3) die off in the subsurface in a timeframe of days to weeks (medium half-life).

However, where the medium favours bypass flow by preferential (fast) flow in macropores and cracks of the topsoil and in fissures and conduits of the bedrocks, none of the three retardation and degradation processes (sorption, filtering and die off) are effective as a major mechanism of lowering of contamination.

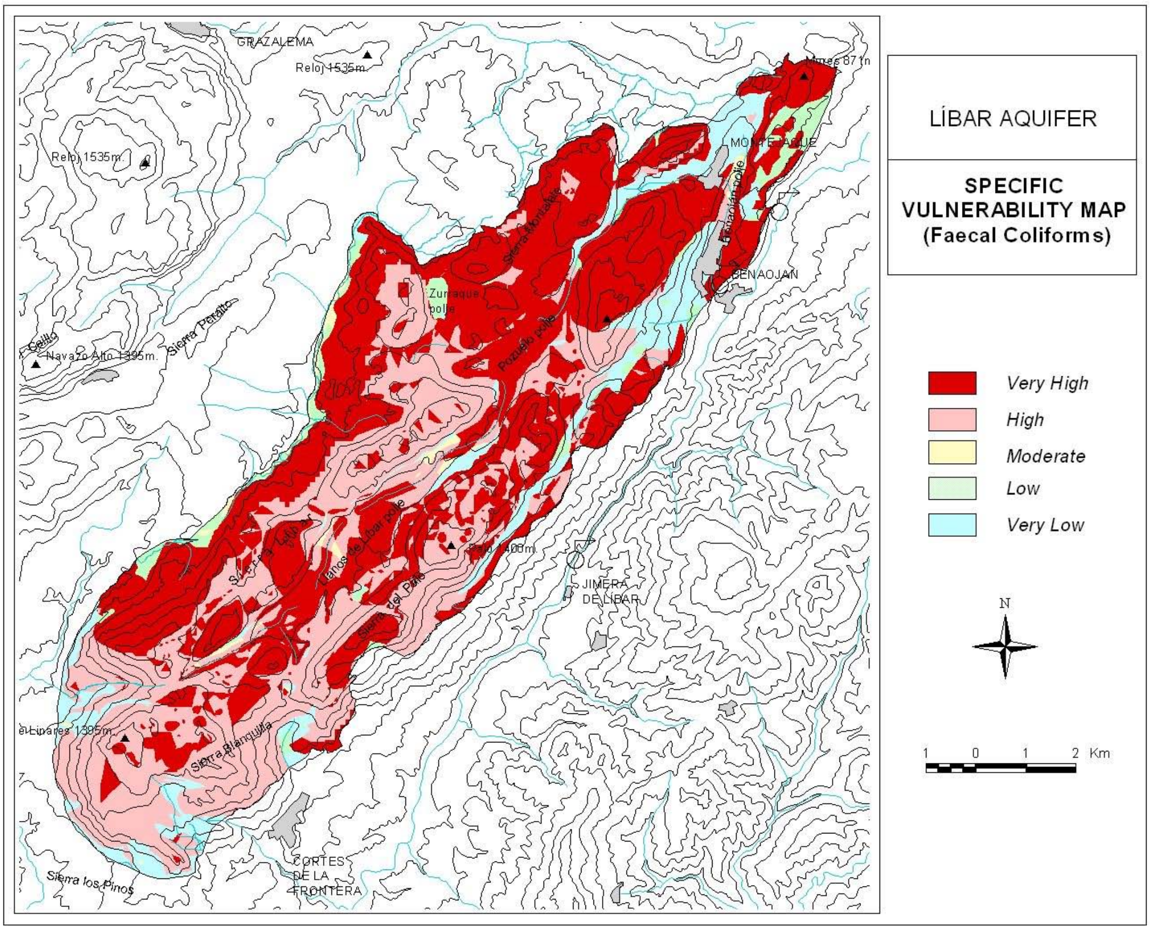

Fig. 58. Microbiological (Faecal coliforms) specific vulnerability map for Sierra de Líbar

In areas of "Very High" or "High" vulnerability as calculated by the COP method, the protection is not increased on the specific vulnerability map (Fig. 58). These areas coincide with the carbonate outcrops of limestone and dolomite of Jurassic age, in which there is a total absence of soils or non karst bedrock, and the rate of preferential (fast) flow is more than 90%. Thus, in these areas, bacteria only suffer die off degradation. However, as the transit time to the resource is small the degradation due to die off is negligible and so neither the aquifer nor the contaminant itself allows more protection than that determined by the intrinsic vulnerability.

On the contrary, for areas having "Moderate" or "Low" intrinsic vulnerability as calculated by the COP method, the vulnerability decreases to "Low" or "Very Low", respectively, with the almost total disappearance of the "Moderate" grade of protection. These areas coincide with soil covered marly limestones and flysch. The clay existing in these lithologies, jointly with the decrease of the preferential flow, facilitates more interaction time with the rock, thus utilising the sorption and filtration capacities of the medium, as well as allowing more time for die-off to occur.

2.2.4.2 Aromatic hydrocarbon contamination scenario

Organic compounds are potentially affected by sorption, biodegradation and volatilisation. As BTEX generally are not very sorbant (low Octanol/Water partition coefficient -K_{OW}-), this process is not taken into regard. However, BTEX can be microbially degraded (moderate half life degradation rate -DT50-) if organic matter supports the existence of microbes. But since this is the case for none of the layers in this catchment (low organic matter content even in the topsoil), biodegradation is also a process regarded as not significant for BTEX attenuation.

Thus volatilisation is the only process undergone by BTEX. As a group they have to be classified as medium-volatile (medium vapour pressure), even if benzene on its own is a very volatile substance. The location in the south of Spain benefits the process of volatilisation (high temperature).

BTEX forms the water-soluble components of gasoline, and their components are included among the group of compounds termed volatile organic compounds (VOC), thus the volatilisation due to the turbulent flow within the conduits of a karst environment is a significant degradation process.

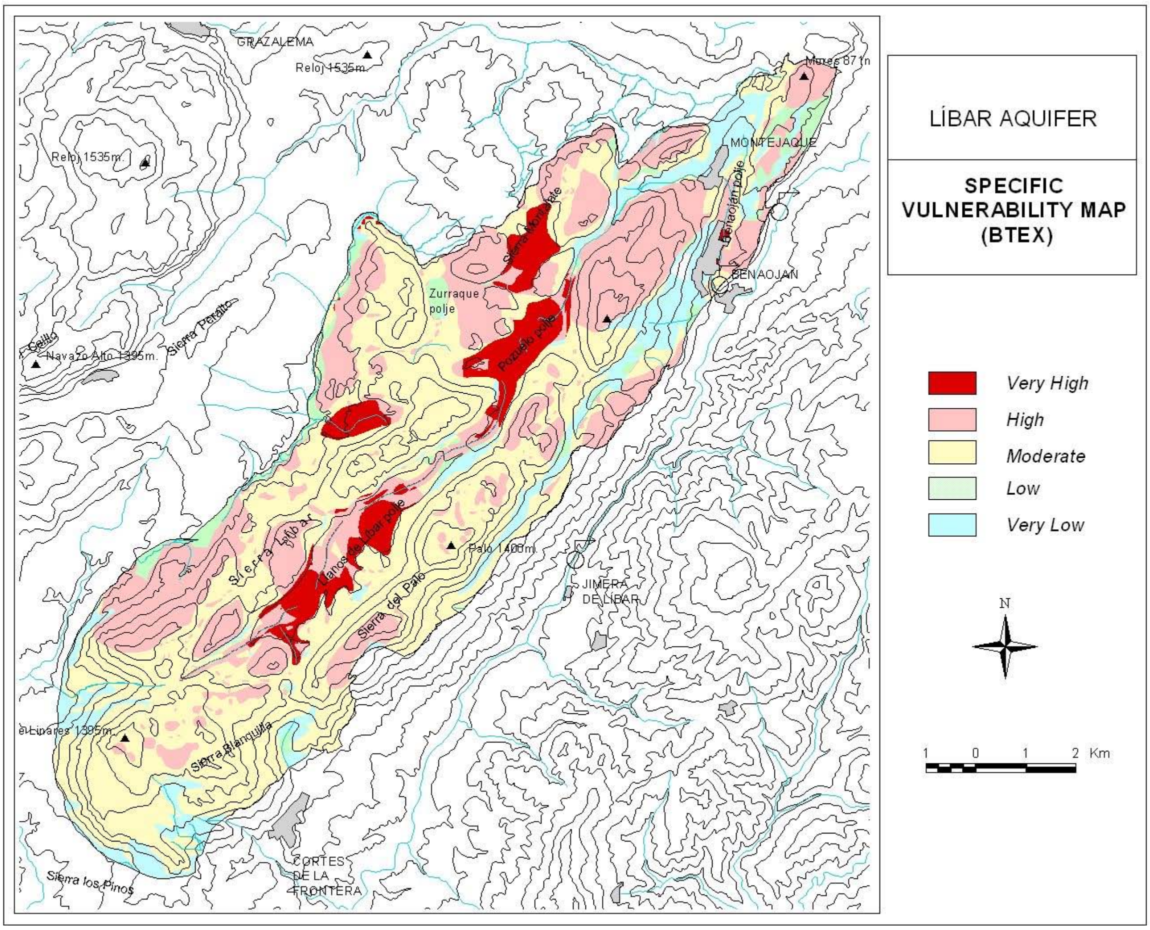

Fig. 59: Hydrocarbons (BTEX) specific vulnerability map for Sierra de Líbar.

Volatilisation is a very important process in karst environments because the applied method judges it as active in the turbulent preferential flow. Therefore, BTEX reach a high specific attenuation where the unsaturated zone is thick enough, the topsoil having lost its prominent importance for specific attenuation in that situation.

The specific vulnerability map for BTEX (Fig. 59) is significantly different to the COP intrinsic vulnerability map. In general, there is an important decrease of vulnerability in all the areas. In most cases the attenuation is of at least one grade. The attenuation of vulnerability over the bare carbonate rock is not a fault of the method but the result of the fact that the most important process of attenuation of these contaminants is the volatilisation that evolves as turbulent flow (> 90%) in big fissures or conduits in the limestones and dolomites. Moreover, greater thickness of the layers favours the contact time between BTEX and the air of the non-saturated zone of the aquifer, with more volatilisation as a net result.

Only in the poljes (Pozuelo and Llanos de Líbar) is the “Very High” vulnerability maintained, due to the smaller depth to the resource in these parts of the aquifer.

The majority of the areas with “Very High” or “High” vulnerability have their vulnerability decreased by one grade, although certain areas situated near the NW and SE border of the aquifer decrease by even two orders.

Areas of “Low” intrinsic vulnerability have their vulnerability decreased to “Very Low”, because the volatilisation of BTEX in the carbonate layers is added to an already appreciable intrinsic protection.

2.2.5 Hazard mapping

2.2.5.1 Description of hazards

The hazard mapping in the Sierra de Líbar largely followed the procedure as proposed by COST Action 620 (see chapter 5 “hazard mapping” in part A of this report). The main part of the test site is uninhabited. Three little villages are situated in the northern part of the test site, which also hosts most of the industrial activities and small farms. The two largest poljes in the central part of the Sierra de Líbar are used as feedlots for cattle, sheep, horses and pigs. The hazards were mapped on a topographic map at scale 1: 10.000.

The first step consisted of the surveying the infrastructure of the test site e.g. villages, roads and railway lines. Solitary houses and buildings were also mapped.

In a second step all the mapped hazards were visited in the field to assess their properties with respect to the quantity of relevant substances and any reduction factor. Further hazards were mapped simultaneously during the fieldwork. Often the required data were unknown and, thus, the ranking factor (Qn) and the reduction factor (Rf) were estimated on the basis of the relative size and the technical conditions of the hazard (see below).

The following hazards were identified in the test site (the numbers in brackets refer to the hazard inventory provided in chapter 5, part A): Urbanization with leaking sewer pipes and sewer systems (1.1.1); Houses and villages without sewer systems (1.1.3); Waste water discharge into surface watercourses (1.1.10); Garbage dump (1.2.1); Gasoline station (1.3.6); Road, unsecured (1.4.1); Railway line (1.4.5); Railway tunnel, unsecured (1.4.6); Railway station (1.4.7); Graveyard (1.6.2); Food industry (2.4.10); Animal barn (3.1.1); and Feedlot (3.1.2).

Due to the relatively small amount of data involved no special database management system was used. However, for data handling and graphical processing a geographical information system was used. Within this software, a vector data model with points, lines and polygons was chosen to receive comparable data to the vulnerability map which also consists of vector data. A database consisting of specific layers (covers) was established for each type of hazard taking into account the spatial properties of the hazards.

For the calculation of the Hazard Index (HI), all required coefficients (H, Qn, Rf) were entered in the form of attributes (columns). The Hazard Index was evaluated with a calculating tool available in the GIS and stored as a separate column. The final database thus includes layers (hazard types) with attribute information stored in tables. The columns of these tables contain spatial information and values for H, Qn, Rf, HI and the Hazard Index Classes. Each row of the table represents one hazard with all the representative data.

The weighting factor (H) describes the harmfulness of the hazard to groundwater. In the case of geographically overlapping hazards, the hazard with the highest value was chosen to represent the harmfulness of the hazard at this location.

The ranking value of the hazards (Qn) ranges between 0.8 and 1.2. For example, the ranking value for a railway tunnel is 0.8 if less than 50 trains pass per day, and 1.2 if more than 100 trains and freight trains pass per day.

The reduction factor (Rf) considers the probability for a contamination event to occur. If no information is available for such an assessment an Rf value of one is used. No information was available relating to the probability of a contamination event for any of the mapped hazards. Information concerning the technical status and the level of maintenance with a view to reduce the Hazard Index was available only for the graveyards. Due to the local manner of burial the reduction factor could be used. The deceased are not buried but laid out in coffins, which are stored in cubicles. For this reason the reduction factor for the graveyards was defined as $R_f = 0.5$.

The Hazard Index classification ranges between 16 and 54. The hazard index in the test site reflects only a small portion of the possible range from 0 to 120. However, the rural characteristic of the test site seems to be well described.

2.2.5.2 Graphical interpretation

The graphical interpretation of the Hazard Index Classes was obtained using standardised Markersets, Linesets and Shadesets. By means of a simple program the map was produced within the GIS.

For an area of such as that included in the test site a 1:25.000 scale map is useful for a general overview. For areas with a higher density of hazards more accurate maps are necessary. For such areas (e.g. towns and villages) a map at scale 1:10.000 is recommended.

The work in the present test site showed, that the “mapping-scale” of 1:10.000 is useful to identify the different hazards in the first step as well as during the fieldwork.

Two maps were produced representing the collected data. The unclassified map shows all hazards in red to give an impression of the hazard types relevant to the test site (Fig. 60). The classified map shows the hazards according to their hazard index class representing the hazard level (Fig. 61).

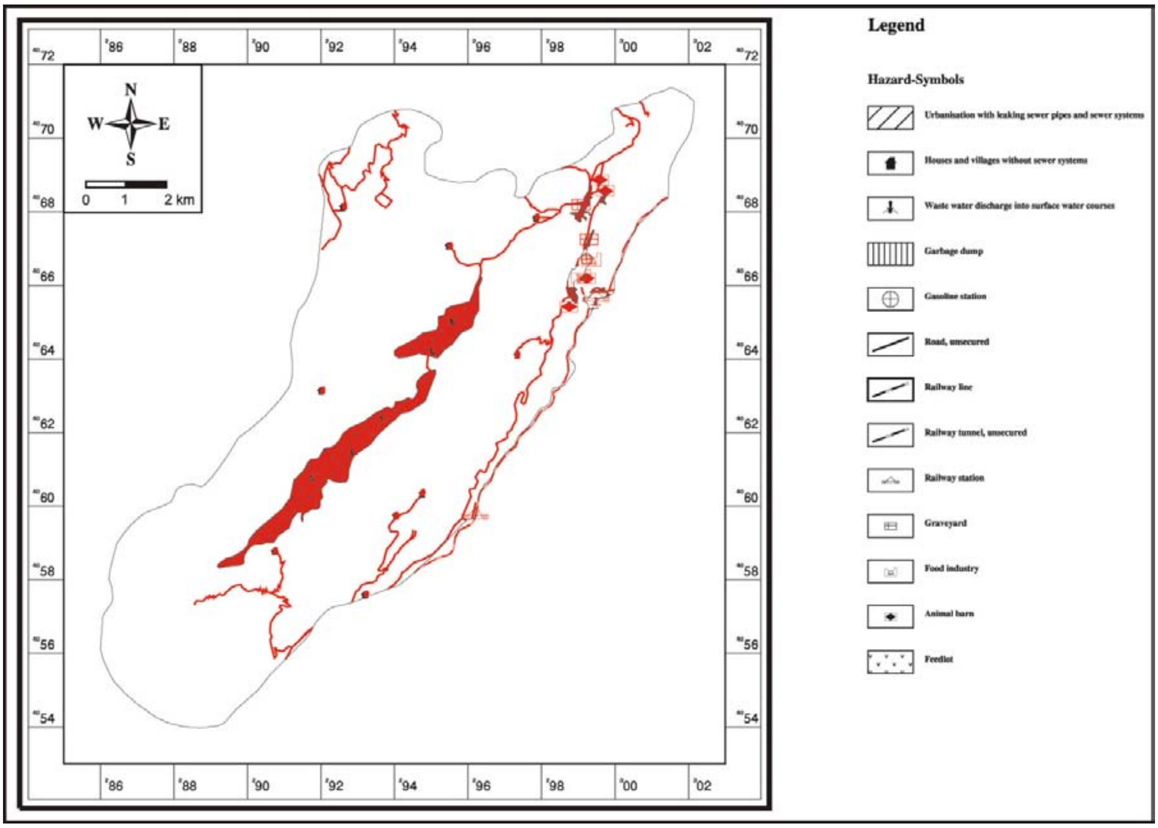

Fig. 60. Unclassified hazard map of the Sierra de Líbar, showing the hazards relevant to the test site

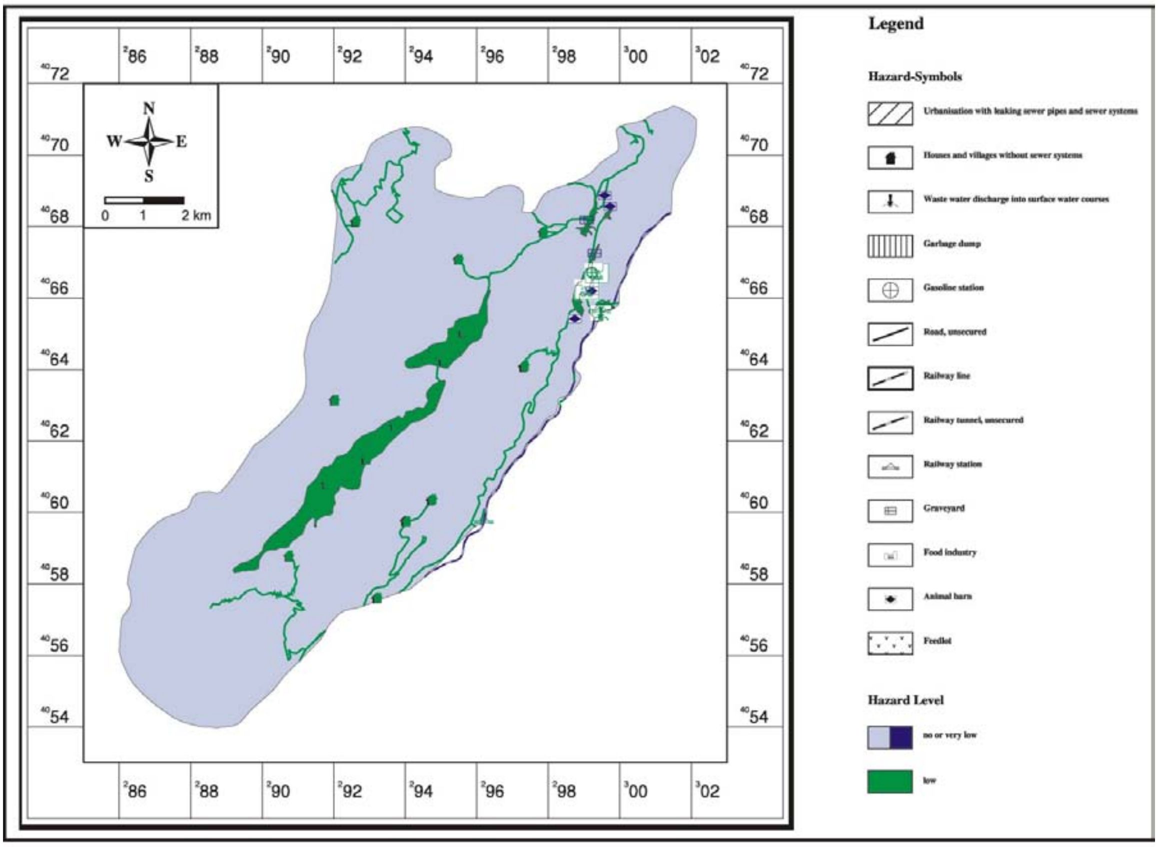

Fig. 61. Classified Hazard map of the Sierra de Líbar

2.2.5.3 Usefulness of hazard map

The test site in the Betic Codillera is made up of strong karstified limestones and contains only a small number of hazards. It represents a typical rural district. Infrastructure as well as industrial activities are poorly represented. Activities with high weighting values such as power plants and industrial storage do not exist. Wide areas are used for agricultural practice, mainly as feedlots.

Consequently the hazards found in the test site are classified by “No” or a “Very Low” hazard level. The gasoline station and the houses without sewer systems are categorised as having a “Low” hazard level. The railway line as well as the graveyard represent “No” or a “Very Low” hazard level.

Nevertheless, due to livestock husbandry the groundwater has suffered point contamination in the past. Even though there are few industrial activities, the hazard map in combination with the vulnerability map is very useful for carrying out a risk assessment to avoid further contamination of the groundwater.

2.2.6 Risk mapping

Several definitions of risk to groundwater have been introduced in the technical literature. The US EPA uses a risk assessment approach in determining the required clean up level for ground water and soil at contaminated sites (FETTER 1999). MORRIS & FOSTER (2000) defined groundwater pollution risk “as the probability that groundwater in the aquifer will become contaminated to an unacceptable level by activities on the immediately overlaying land-surface”. This approach uses the interaction between the infiltrating contaminant load and the vulnerability of the aquifer at the location concerned. The risk assessment followed in the test site Sierra de Libar is nearly identical to those proposed by MORRIS & FOSTER (2000) and is the most simple way to assess the risk to groundwater within the framework of the COST Action 620. Considering the intrinsic vulnerability and the hazard assessment scheme with their definitions, the risk definition in this assessment scheme can be described as the risk to groundwater pollution from each hazard when its contamination load is released. The risk maps therefore show the risk to groundwater pollution of each mapped hazard in the sense of resource protection. Thus, the groundwater and the characteristics of the saturated zone of the aquifer are not included in the risk assessment.

2.2.6.1 Risk assessment

According to MORRIS & FOSTER (2000) and FOSTER & HIRATA (1988) the proposed risk assessment is calculated by overlaying the intrinsic vulnerability map and the hazard map. The risk map of the Sierra de Libar was produced using the vulnerability map constructed with the PI-Method (BRECHENMACHER 2002) and the hazard map as previously described. The risk value is determined by the simple equation:

$$R = 1/HI \cdot \pi$$

where R is the Risk Value, HI is the Hazard Index and π is the PI-Factor.

As the values of the Hazard Index and the PI-factor are inversely related, the reciprocal Hazard Index was used to ensure non-ambiguous risk values.

The width of the classes was determined from a consideration of the classes of the vulnerability map as well as those of the hazard map (Figs. 1 and 6). The limits of the risk classes are the product of the limits of the vulnerability classes and the Hazard Index (Tab. 30), an operation that is very easy to implement with other vulnerability methods.

Tab. 30. Classification of the risk map

π - factor	Hazard Index	1/HI	π • (1/HI)	Risk Class	Risk Level	Colour
4 - 5	0 - 24	> 0.042	> 0.167	1	no or very low	blue
3 - 4	24 - 48	0.042 - 0.021	0.167 - 0.063	2	low	green
2 - 3	48 - 72	0.021 - 0.014	0.063 - 0.028	3	moderate	yellow
1 - 2	72 - 96	0.014 - 0.010	0.028 - 0.010	4	high	orange
0 - 1	96 - 120	< 0.010	< 0.010	5	very high	red

This approach of risk assessment implies that hazards even with a low or very low hazard level could produce a "Very High" risk level if the vulnerability is very high. In comparison a hazard with a very high hazard level would produce only a "Moderate" or "High" risk level if the vulnerability is less than very high. This is interesting for land-use planning.

To keep the original spatial information of the point hazards, the linear hazards (in this case: roads and railway lines) were buffered with a 10 m radius, which is realistic if we consider the affected surface in the case of contaminant release. Furthermore this step facilitates the further processing of the data and leads in some cases to a simplification of the risk map. Another advantage in keeping the spatial information of the point hazards is that the point symbols from the hazard map can be adopted and the graphical interpretation of the risk map is easier.

2.2.6.2 Results

The hazards occurring in this area are in general of the least dangerous type, with Hazard Indices of 20 to 50, and occur as villages in the northeast and as feedlots in the poljes in the central region. Conversely, the vulnerability of most of the area is classified as "high" or "extreme" due to the strong karstification of the outcropping limestones. The calculated risk level, according the above formula, in an area with more or less unique vulnerability classification, depends strongly on the occurrence of the hazards. As a result the risk distribution (Fig. 62) shows a differentiated spatial classification as compared to the PI vulnerability map (Fig. 56). Therefore in spite of the high vulnerability of the marginal mountains and their slopes towards the central polje there is a relatively low risk assessment ("low" to "very low") due to the absence of hazards. But in the direct adjacent polje floor as well as in the marginal mountains where the houses without sewer system are, these "low" hazards provide a "very high" risk level for the groundwater due to the high PI factor for the karst. Also the rather low hazard of the feedlots in the central poljes leads to "Moderate" risk levels due to the relatively high vulnerability factors.

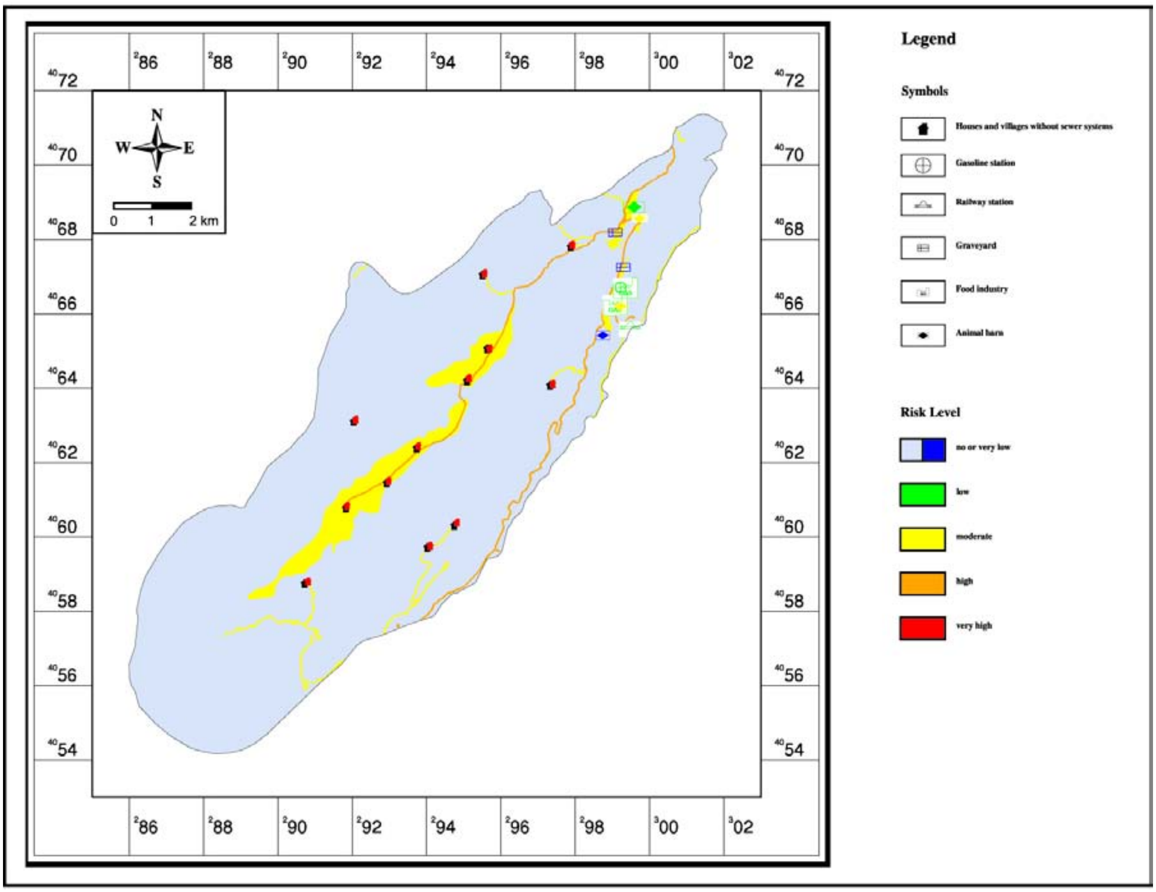

Fig. 62. Risk map of the Sierra de Líbar constructed by overlaying the PI vulnerability map (Fig. 56) and the classified hazard map (Fig. 61).

In the villages the risk level is definitely lower due to the lower classification of the hazards and the partly lower vulnerability of the more marly rock sequence. The point-hazards do not come up with high Hazard Index numbers - as was shown in the previous subchapter. The total risk of the groundwater therefore remains low to moderate in these parts of the catchment.

2.2.7 Conclusions

The intrinsic vulnerability maps for the Sierra de Líbar carried out by two methods (PI and COP) developed in the framework of COST 620 Action show a high to very high vulnerability in the catchments of swallow holes and on outcrops of karstified rocks where the epikarst is highly developed. The COP method discriminates between areas with different degrees of vulnerability within the Jurassic carbonate rock outcrops taking into account the thickness of the unsaturated zone, the degree of surface karst development and different slope gradients. The maps obtained using both methods show lower vulnerability degree on the outcrops of Cretaceous marl and clayey Flysch formations.

The specific vulnerability maps of Sierra de Líbar for faecal coliforms and aromatic hydrocarbons illustrate different behaviours of contaminants due to the different contaminant properties and theirs interactions with the lithology. Thus, for bacteria, the specific vulnerability assessment is the same as that for the intrinsic vulnerability where the latter is “high” or “very high” as the rapid transit time precludes significant degradation. Conversely, for areas with moderate or low intrinsic vulnerability, the specific vulnerability decreases to low or very low degree respectively, because of the clay and the slow flow of groundwater in these areas, which facilitate the sorption and filtration capacities of the medium, and greater time for die off to occur. On the other hand, for aromatic hydrocarbons, the vulnerability is decreased in practically the whole area. In most of the cases, the attenuation is at least by one grade due to the importance of vitalization to the degradation process.

Sierra de Líbar is a strongly karstified aquifer but it is a rural area with a small number of hazards, which have low weighting values. The villages in the northeastern part and the feedlots in the poljes of the central part are the most important hazards. Infrastructure and industrial activities are poorly represented. Consequently the hazards found in the test site are classified as having “No” or a “Very Low” hazard level. Nevertheless, point contamination episodes have been detected in the past due to livestock husbandry. So, the combination of hazard and vulnerability maps is very useful to achieve a risk assessment and, thus, avoid contamination of the groundwater in the future.

The groundwater contamination risk map for Sierra Líbar shows a spatial classification different to those of the intrinsic vulnerability map. Thus, in the marginal mountains of the poljes, where the intrinsic vulnerability is high, a low risk exists due to the absence of hazards. However, in the floor of the polje and at the edge of the mountains where the houses without sewer system are placed, a very high risk has been calculated. In the poljes of the central sector the low hazard assessed for feedlots leads to moderate risk levels due to the high vulnerability. In the villages the risk level is lower due to the lower classification of the hazards and the lower vulnerability of the Cretaceous marls.

The Sierra de Líbar case study is a good example of the fact that the final risk map does not include all the information that might be required by an end user. For example, a land-use planner using the final risk map will know that the existing situation at the margins of the mountains poses low or moderate risk to groundwater. However, to ascertain the risk, which might be posed by future developments, the planner would also require access to the intrinsic vulnerability map and appropriate specific vulnerability maps.

2.2.8 References

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2.3 Engen, Swabian Alb, Germany

– Comparative application of the German GLA method, the Swiss EPIK method and the PI method of intrinsic vulnerability mapping, and hazard mapping –

2.3.1 Geographical and Geological Overview

The test site is a drinking water protection area of the city of Engen and neighbouring communities. It belongs to the Hegau landscape in the Swabian Alb and covers an area of 36 km^2 (location see Fig. 55). The altitude ranges between 470 m in the southern lowland and 690 m in the in the northern hills. The mean annual temperature is 8,1°C (478 m).

The area is located within the transition zone of the South German cuesta landscape, formed by Mesozoic sedimentary rocks, and the Tertiary Molasse foreland basin of the Alps (Fig. 63). The area is made of gently (2–3°) SE-dipping Upper Jurassic (Oxfordian-Tithonian) carbonate formations with a total thickness of 300–400 m. The predominant rock types are marl and limestone, which is present in bedded and massive facies. These formations are partially overlain by Oligo- to Miocene Molasse sediments and, locally, volcanites. Glacial and fluvio-glacial deposits of the Riss and Würm Ice Age (alluvial gravel terraces, glacial gravel, moraines) cover large parts of the area. In the Holocene, detritus formed on slopes and alluvial sediments were deposited in the valleys (SCHREINER 1992).

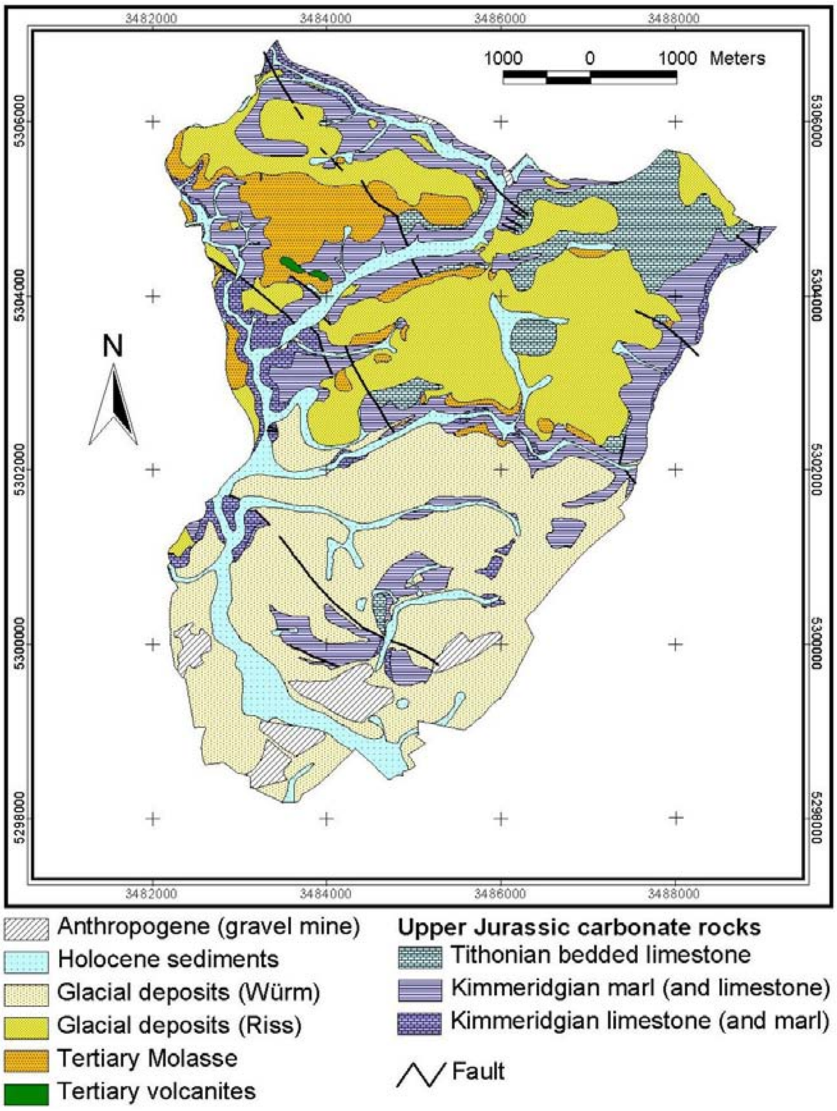

Fig. 63: Geological map of the Engen area.

The soil distribution reflects the geology and landscape history. The soils on Jurassic carbonate rocks (rendzina) are characterized by low to medium effective field capacities (eFC) (50-140 mm) and high hydraulic conductivities (40-300 cm/d). On the old glacial deposits, there are a large variety of deep, loamy soils with medium to high eFC (90-200 mm) and very low conductivity (< 1 cm/d). Soils on young glacial deposits show favourable hydraulic properties: medium to high eFC (90-200 mm) and moderate conductivity (10-40 cm/d). The soils on alluvial sediments are often characterized by a low to medium eFC (50-140 mm) and a very high hydraulic conductivity (> 300 cm/d).

In large parts of the area, the karstified Jurassic rocks are covered with sediments and soils. Exokarst features are rare and often not noticeable. The only relevant karst landforms are dry valleys. However, due to the widespread low permeability sediments, there are many surface waters. Most of the valleys are not permanently dry but the watercourses sink underground in different places, dependent on the respective hydrologic condition.

Many karst cavities are visible in quarries, most of them filled with Cretaceous to Tertiary bean iron ore loam. Since the Pliocene, some of them have been reactivated. The Oxfordian and Kimmeridgian limestones form the main karst aquifer, which has been studied in detail within the framework of the investigation of the Danube-Aach-System (e.g. HÖTZL 1971).

Locally, there are higher aquifers above the main karst aquifer. In the north of the Engen area, there are four perched aquifers in Tithonian limestone. In the south, the karst aquifer is covered by sand and gravel with an interstratified clayey layer. Tracer tests proved that karst water rises up into the granular aquifer.

2.3.2 Intrinsic Vulnerability

2.3.2.1 GLA method

DICKEL et al. (1993) applied the German GLA method (HÖLTING et al. 1995; brief description see annex) in the Engen test site. The results are summarised in this section.

The effective field capacities of the soils were taken from the soil map. The grain size distribution of the subsoils and the properties of the bedrocks (lithology, fracturing, karstification) were assessed on the basis of geological maps, field observations and lab-analyses. The thickness and distribution of the layers was determined by intersecting the geological map, the DEM and the groundwater contour lines using GIS operations.

The GLA map (Fig. 64) shows the protective function of the layers above the groundwater surface in the Upper Jurassic karst aquifer and, vice versa, the vulnerability of that resource. The perched aquifers are considered to protect the underlying karst aquifer. In the southern lowlands, artesian pressure was proved in the karst aquifer. A high protective function was consequently assigned to this area, although the karst aquifer is overlain by a highly vulnerable granular aquifer. The elevated areas in the north are also characterized by a high to very high protective function due to the large depth to groundwater table and the high thickness of the protective cover respectively. A very low to moderate protective function was calculated for most of the valleys because the depth to groundwater table is reduced there.

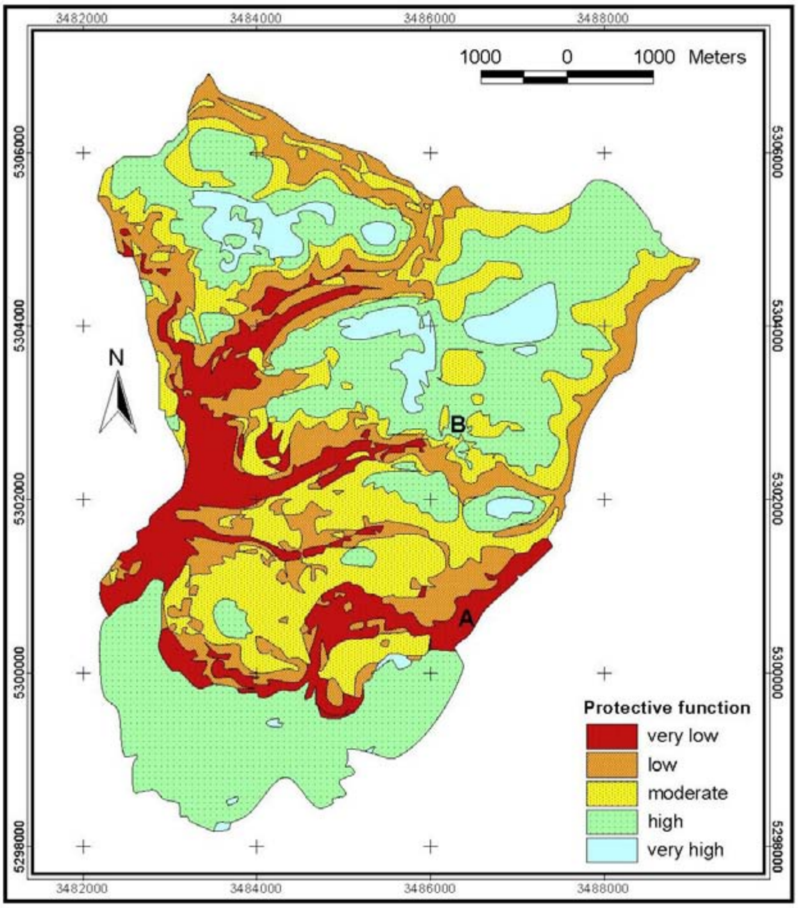

Fig. 64: Map of the protective function of the layers above the groundwater surface (vulnerability map) using the GLA method (Dickel et al. 1993).

2.3.2.2 EPIK method

Sturm (1999) and Klute (2000) applied the EPIK method (DOERFLIGER & ZWAHLEN 1998; brief description in the annex) in the test site. The method requires the evaluation of the epikarst (E), the protective cover (P), the infiltration conditions (I) and the karst network (K).

The epikarst (E) was assessed using topographic maps and field observations. The class E_1 is restricted on a few small dolines; E_2 was assigned for the dry valleys; the rest of the area was classified as E_3 (Klute 2000).

EPIK demands for relatively simple information on the protective cover (P), and so the detailed available database had to be generalized. The test site was subdivided into areas with or without low permeability formations; their respective thickness was determined using GRID data. The soil thickness was taken from the soil map. This information was re-combined using the EPIK classification. As large parts of the area are covered with sediments and soils, the class P_4 predominates by far; the classes P_2 and P_3 are present in the valleys (Sturm 1999).

The first step to evaluate the infiltration conditions (I) is to identify swallow holes and sinking streams. In the northern and central Engen area, all streams infiltrate into the karst aquifer – permanently or temporary, totally or partially – and were consequently classified as sinking streams (class I_1). The delineation of the catchments was problematic as the topographic catchment of a sinking stream is often larger than the effective hydrologic catchment. A coverage showing the slope angle (<10 %, 10-25 %, >25 %) was created using a DEM, which had to be transformed into the TIN format in order to obtain more precision (Klute 2000).

EPIK distinguishes between arable areas and meadows/pastures, but does not consider forests and settlements. However, for the Engen area, it is more applicable to distinguish between forest and non-forest areas. The classification into I_{2-4} was done by intersecting the coverages showing the hydrologic catchments, the slope gradients and the land-use/vegetation.

As the Engen area is a part of the Danube-Aach-System, there is clear evidence for a well developed and connected karst network. Thus, the entire area was classified as K_1 (Sturm 1999).

The EPIK vulnerability map (Fig. 65) was created by intersecting the four coverages and calculating the protection index F. Large areas are classified as moderately vulnerable; the dry valleys and the bordering slopes are zones of high to very high vulnerability (dependent on the gradient and land-use); the sinking streams are very highly vulnerable. Altogether, this is a largely plausible distribution of vulnerability zones. However, all areas with diffuse infiltration and without epikarst features are classified as moderately vulnerable, although the protective cover may be very thin.

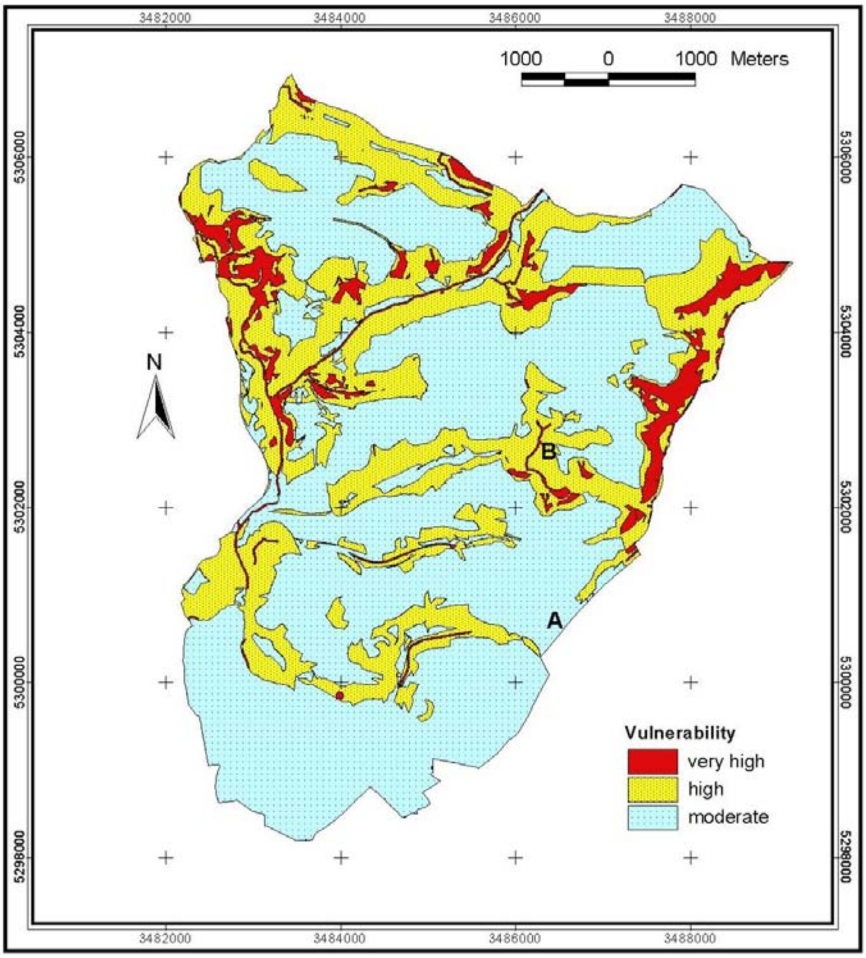

Fig. 65: EPIK vulnerability map of the Engen area.

2.3.2.3 PI method

The following steps were carried out in order to determine the I factor (Fig. 66):

- 1st step: determination of the dominant flow process as a function of soil properties

The dominant flow process is assessed on the basis of the topsoil permeability and the presence of low permeability layers ($k < 10^{-6}$ m/s): Surface flow has to be expected on low permeability soils; subsurface flow takes place in highly permeable soils with low permeability layers; infiltration predominates if low permeability layers are absent. The digital soil map contains data on the permeability of the soils in different depths (0-30, 30-60, 60-100 cm) and the underlying bedrock. The dominant flow process was determined by intersecting the coverages 'topsoil permeability' and 'depth to low permeability layers' (Klute 2000).

- 2nd step: determination of the I' factor

The intensity of lateral surface and subsurface flow also depends on the slope gradient and the vegetation/land-use. Gentle slopes and forests favour infiltration, while steep slopes and agricultural land-use favour lateral flow. The I' factor is determined by intersecting the coverages 'dominant flow process', 'vegetation' and 'slope gradient'. In the test site, surface or subsurface flow has to be expected in settlement areas, on steep slopes bordering the valleys and on areas covered with clayey sediments.

- 3rd step: determination of the I factor

Lateral surface and subsurface flow is relevant for groundwater vulnerability only if the water enters the underground at another place, e.g. via a swallow hole. Consequently, the I map (showing the degree to which the protective cover is bypassed) is obtained by intersecting the I' map (showing the intensity of lateral flow) with the surface catchment map (showing the sinking streams and their catchments). The latter was created on the basis of a digital map showing all swallow holes and sinking streams. The 10 m and the 100 m zones were created with the buffer command and the catchments of the sinking streams were delineated automatically from the DEM. The I map shows that lateral flow components, which bypass the protective cover, have to be expected along the valleys in the northern part of the area, while the protective cover is not likely to be bypassed in the southern part of the area.

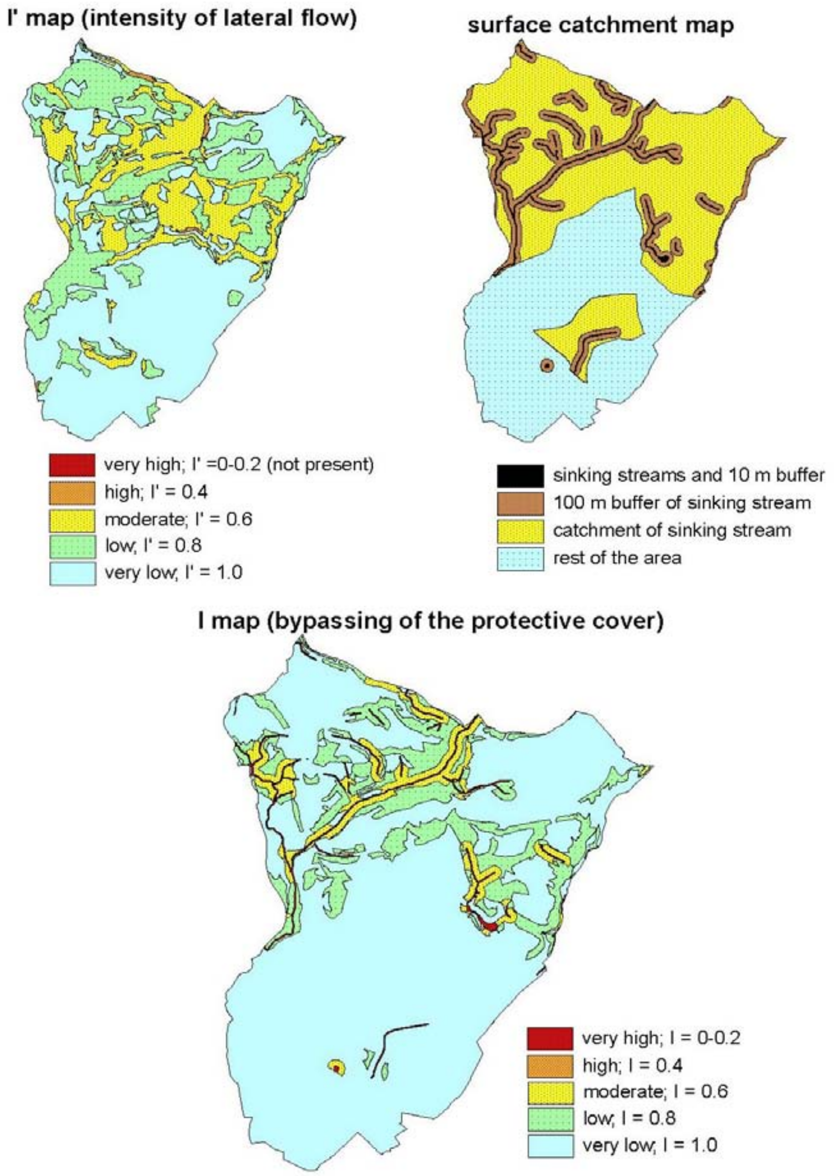

Fig. 66: Determination of the I map (showing the degree to which the protective cover is bypassed by lateral surface and subsurface flow) by intersecting the I' map (showing the intensity of lateral flow) and the surface catchment map (showing the sinking streams and their catchments).

The P factor is calculated using a slightly modified version of the GLA method described above. Therefore it was possible to use the existing GLA vulnerability map for the Engen area as a basis for the construction of the P map. However, two significant changes were made:

- The score ranges of the total protective function are much wider in the PI than in the GLA method. As a consequence, the areas of very high natural protection disappear completely on the P map and the areas with very low protection become much smaller.
- The PI method always takes the groundwater surface in the uppermost aquifer as the target. Therefore, all areas with higher aquifers above the karst aquifer had to be re-evaluated. A bold line on the P and the PI map shows the border of these higher aquifers.

The final PI vulnerability map was obtained by intersecting the P and I maps (Fig. 67). The protection factor π was calculated by multiplying the P and I factors. The range of values for π was subdivided in five classes of natural protection and vulnerability respectively.

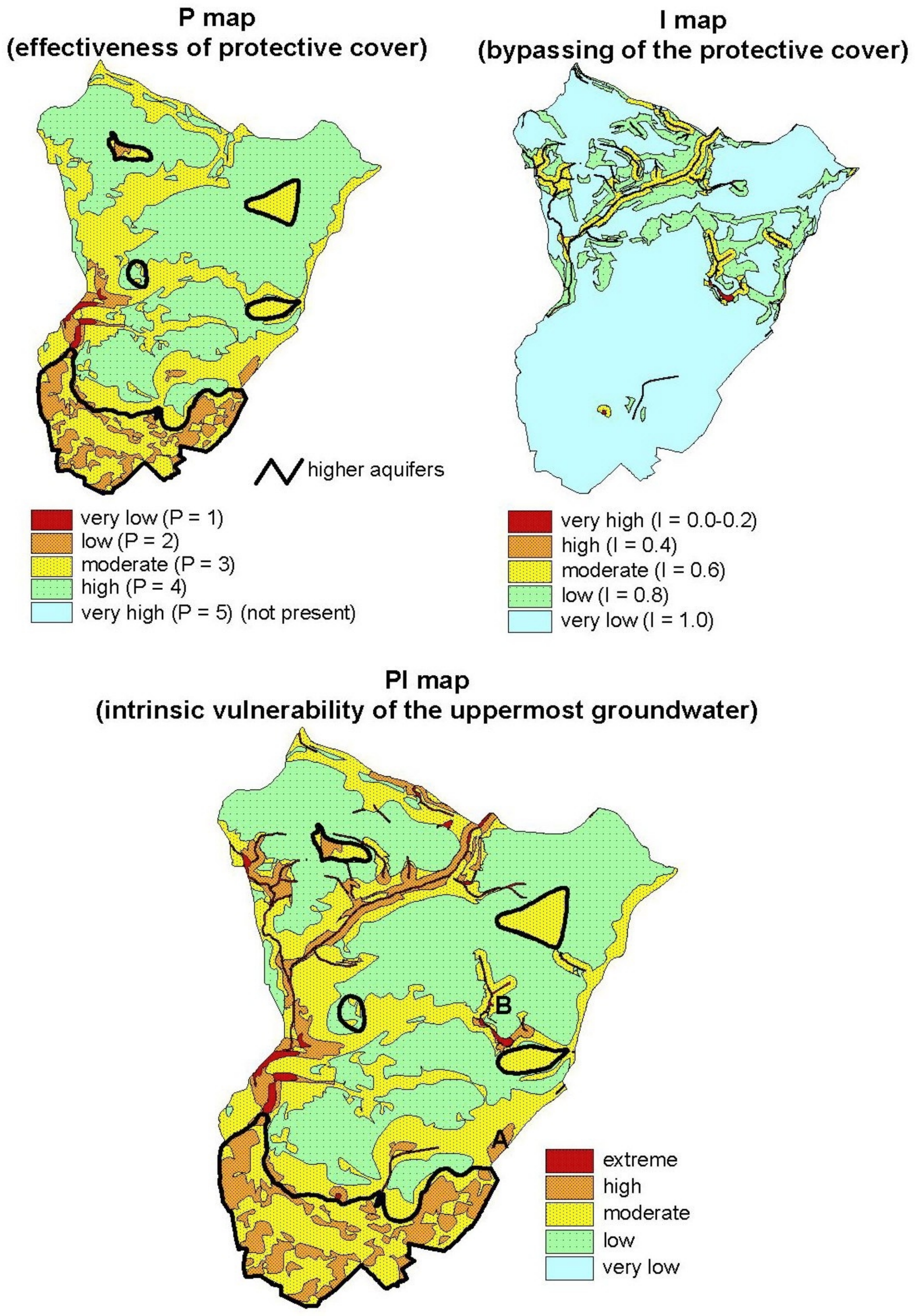

Fig. 67: The PI vulnerability map of the Engen area was created by intersecting the P map (showing the protective function of the overlying layers) and the I map (showing the degree to which the protective cover is bypassed by lateral surface and subsurface flow).

On the PI vulnerability map, most areas range between high and low vulnerability. Only the swallow holes, the sinking streams and some outcrops of karstified limestone turn out to be extremely vulnerable. A high to moderate vulnerability was assigned to large parts of the valleys, the perched aquifers and the granular aquifer in the southern lowlands. The elevated areas, which are covered by glacial deposits and Tertiary sediments, show a low vulnerability. Areas with very low vulnerability are not present.

2.3.2.4 Comparison and discussion of the maps

Comparing the three vulnerability maps, it is noticeable that the valleys are always assessed to be more vulnerable than the bordering hills. However, there are different reasons for that common result: on the GLA map, the valleys are vulnerable, because the overlying layer thickness is reduced there; on the EPIK map, the valleys are classified as epikarst features (E) and form the catchments of sinking streams (I); on the PI map, both the reduced thickness (P) and the concentrated infiltration (I) are taken into account.

In detail, there are significant differences between the three vulnerability maps, which will be demonstrated by means of two examples (see points **A** and **B** on the three maps): Point **A** is classified as extremely vulnerable on the GLA map, because the protective cover is thin and permeable. EPIK considers the same point to be moderately vulnerable due to the absence of epikarst and the presence of diffuse infiltration. On the PI map, the point is highly vulnerable, because the protective cover is thin but the infiltration is diffuse. Point **B** is extremely vulnerable on the EPIK and PI map, as it is located near a sinking stream. However, there are thick impervious layers below this point, and so it is classified as moderately to lowly vulnerability on the GLA map.

2.3.3 Hazard Mapping

2.3.3.1 Overview

Forest covers 50 % of the area, 44 % are used for agriculture and 6 % are settlements and industrial parks. Engen is the biggest settlement with approximately 6500 inhabitants. Several villages with only a few hundred inhabitants are situated within or at the margin of the area. The area is crossed by a railway line and several roads: the motorway A 81, the national roads B 491, B 31 and B 33, several smaller roads and a network of farm and forest tracks.

2.3.3.2 Description of hazards

The hazards within the Engen test site were identified according to the hazard inventory proposed by COST 620 and mapped on topographic maps at scale 1:25,000. First all information on the infrastructure, like for example roads, railway lines and graveyards was derived directly from the topographic map. Some information on the location of gasoline stations and industrial plants was available in mercantile directories and on the municipal Internet homepage. Information on houses, which are detached from the municipal sewage system, was available from the city council. Further hazards were identified during a field survey. According to their spatial extension, three types of hazards were distinguished – polygon, line and point hazards. Polygon hazards result from urbanisation (leaking sewer pipes) (5.2 % of the total area), industrial parks (1.4 %), car parking areas (0.1 %), gravel pits (0.5 %), feedlots (0.7 %) and intensive agriculture (27.6 %). Line hazards include 12.6 km of railway line and almost 100 km of roads and farm tracks outside the settlements. Of these, 7.2 km are motorway, 12.9 km main roads with up to 10,000 cars per day, 19.8 km country roads with up to 1,000 cars per day, 1.3 km minor roads with less than 100 cars per day and 56.9 km farm and forest tracks with less than 10 cars per day. The most common point hazards in the Engen test site are houses detached from the public sewage system, gasoline stations (private and public) and farms. Most industrial hazards are located in three industrial parks. The factories are small to medium sized and of different age and technical standard. Graveyards and recreational activities are situated close to the settlements, whereas most of the dwellings with their own water treatment or waste water storage are more or less isolated farms.

2.3.3.3 Determination of the Hazard Indices

The *weighting factor*, which describes the harmfulness of the hazard to the groundwater, was determined according to the values proposed by COST 620. Farms can only be mapped as one single hazard at the given scale, although they often include several different hazards (e.g. animal barn, manure heap and closed silage). Thus, only one hazard (manure heap) was chosen to represent farms. In addition to the individual factories, industrial parks were mapped as a general hazard (industrial plants, non-mining) and assigned an average weighting factor of 55. The *ranking factor* was assessed considering the range of possible technical specifications of each hazard type.

Urbanisations with leaking sewage pipes were ranked according to the number of inhabitants of the settlement using a logarithmic scale from <1,000 to >1,000,000 inhabitants. In doing so, the ranking of this hazard leads to a Hazard Index between 24 and 48 (Hazard Index Class 2). The number of citizens in the townships was also used for the ranking of the respective graveyards. Houses detached from the public sewage system were rated considering the number of inhabitants. Gasoline stations were ranked according to their size and number of pumps, ranging from single, private pumps (Qn = 0.8) to highly frequented, large gasoline stations (Qn = 1.2).

Roads outside the settlements were ranked considering the number of cars making use of these. A logarithmic classification ranging from <10 cars per day to >10,000 cars per day was used for this hazard. The same scheme was also applied for car parking areas. Railway lines and stations were ranked according to a similar scheme considering the number of passenger and freight trains. 160 trains per day use the railway line crossing the test site. A ranking factor of 1.0 was applied for the two railway stations, a factor of 1.2 for the railway line.

Recreational facilities (campground and sport stadium) were ranked according to their size. A ranking factor of 0.8 was applied for the small campground, factors of 0.8 and 0.9 for the sports fields in Engen.

The two active gravel pits were rated according to their spatial extension. A factor of 0.9 was applied to the smaller pit, a factor of 1.0 to the larger one. For industrial plants a possible range from small family enterprises to large industrial plants was presumed. The same scheme was also applied for industrial storage.

Only one hazard (manure heap) was chosen to represent farms. Ranking was made according to the size of the farms respectively the number of livestock. Feedlots were rated considering the number of cattle per hectare (LU = livestock units), ranging from < 0.5 LU/ha to > 2.0 LU/ha. For sheep, a factor of 0.1 LU/animal was applied. The calculated numbers in this area might be too high, since the area used as a feedlot was not always clearly distinguishable.

All agricultural area in the test site was classified as intensive agriculture. However, crop rotation with rapeseed apparently being practised on many fields, only a low ranking factor was applied (0.8). Greenhouses were also rated according to their size, with the greenhouse in Engen taken as an average sized greenhouse (Qn = 1.0).

The *reduction factor* considers the probability for a contamination event to occur. For most of the hazards in the Engen test site no information was available concerning their technical status and level of maintenance and therefore R_f was defined as 1 (no reduction). In case of very new industrial plants or gasoline stations a lower reduction factor was used. Houses with private wastewater storage or treatment system were rated according to the type of storage or treatment (ranging from 0.6 = biological treatment plant to 1.0 = latrine). As the highway A81 is equipped with its own drainage and security system, a reduction factor of 0.5 was used.

Tab. 31 shows the hazards found in the Engen test site together with their possible ranges of the Hazard Index and Hazard Index Class. The influence of the ranking and reduction factor on the Hazard Index Class is limited. Most of the hazard index values vary between two Classes. Such variation was obviously smallest when no information on the technical standard was available and a reduction factor of 1 was applied. However, the hazards in the Engen test site represent Hazard Index Classes ranging from 1 to 3 whereas very low and low Hazard Levels dominate.

Tab. 31: Hazard inventory of the Engen test site with the possible range of the Hazard Index values and corresponding Hazard Index Classes.

Hazard	HI	HIC	Hazard	HI	HIC
Urbanisation with leaking sewer pipes and sewer systems (1.1.1)	14-42	1-2	Gravel and sand pit (2.2.2)	12-36	1-2
Septic tank, cesspool, latrine (1.1.3)	18-54	1-2	Metal processing and finishing industry (2.4.3)	20-60	1-3
Gasoline station (1.3.6)	22-66	1-2	Chemical factory (2.4.6)	28-85	2-4
Road, unsecured (1.4.1)	16-48	1-2	Food industry (2.4.10)	16-48	1-2
Car parking area, unsecured (1.4.6)	12-36	1-2	Containers for hazardous substances (2.6.2)	24-72	1-3
Railway line (1.4.5)	8-24	1	Feedlot (3.1.2)	12-36	1-2
Railway station (1.4.7)	12-36	1-2	Manure heap (3.1.4)	16-48	1-2
Campground (1.5.2)	12-36	1-2	Intensive agriculture area (3.2.4)	8-24	1-2
Open stadium (1.5.3)	8-24	1	Greenhouse (3.2.6)	8-24	1-2
Graveyard (1.6.2)	10-30	1-2			

2.3.3.4 Graphical interpretation

The graphical interpretation of the hazard map was done in a GIS, using standardised polygon, line and point symbols (*shadesets, linesets, markersets*) (Fig. 68). In order to improve the visibility of the hazard symbols they were supplied with a white background hiding the topographic map. Some groups of hazards, like for example gravel and sand pits, were mapped as points and polygons in order to improve the cartographic representation.

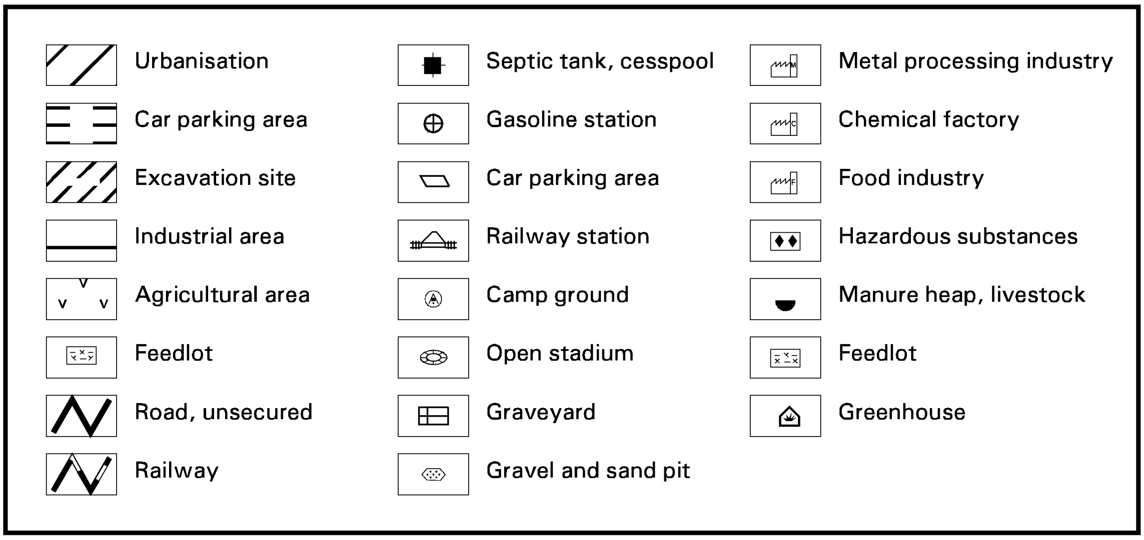

Fig. 68: Symbols representing the hazard in the test site.

The unclassified map (Fig. 69) shows the inventory of hazards in the test site in red, independent of the Hazard Index. The classified hazard map (Fig. 70) shows the mapped hazards in different colours according to the Hazard Index Class.

2.3.3.5 Usefulness of the hazard map for the test site

The hazard map of the Engen area was mapped at scale 1:25,000 and printed at scale 1:50,000. At this scale, only a general overview of the hazards and their location can be given. Especially in areas with a high concentration of hazards, like in industrial areas, the currently applied representation is not ideal for this scale of the map. For further investigations and decision-making purposes, more detailed maps, for example of the industrial areas, would be necessary.

Since for many hazards the Hazard Index Class is almost independent of the ranking and reduction factor, the graphical interpretation does not show finer differences between the individual hazards. For a future combination of the Hazard Map with the Vulnerability Map, it would be sensible to use the Hazard Index instead of the Hazard Index Class in order to maintain the influence of the ranking and reduction factors.

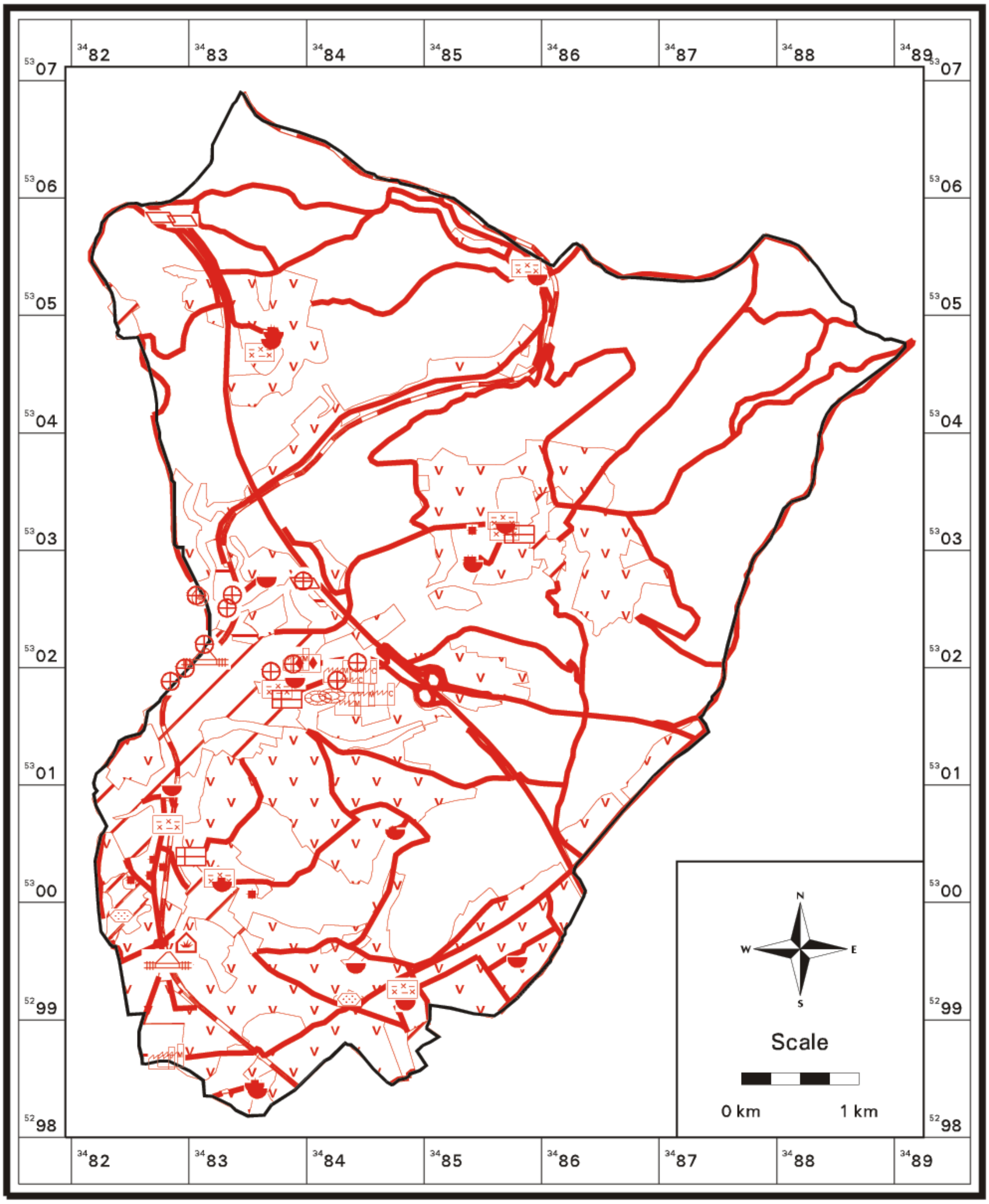

Fig. 69: Unclassified hazard map of the Engen test site showing the hazard inventory.

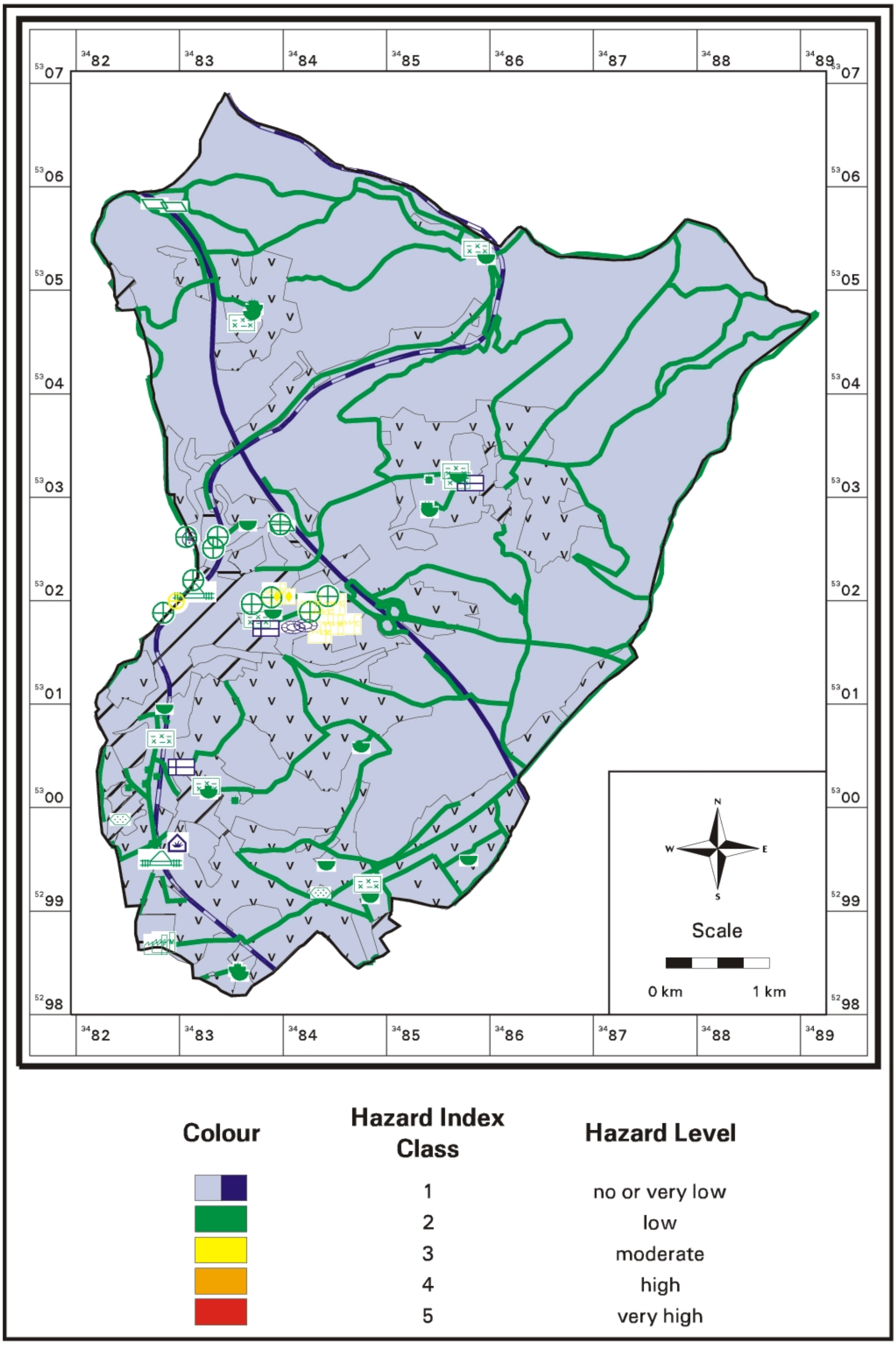

Fig. 70: Hazard map of the Engen test site.

2.3.4 Risk mapping

2.3.4.1 Overview

Many definitions of risk and risk assessment schemes have been developed over recent years as a result of the lack of information concerning the potential of human activities to pollute groundwater (FETTER 1999, FOSTER & HIRATA 1988, MORRIS & FOSTER 2000). Additionally, the European Commission emphasises the importance of risk assessment within the European Framework Directive (EUROPEAN PARLIAMENT AND COUNCIL OF THE EUROPEAN UNION 2000).

At the start of its work COST Action 620 prepared unambiguous definitions of risk and risk assessment to avoid subsequent misunderstandings. These definitions are given in Chapter 6 where the concepts of risk assessment are also explained. The risk map of the test site at Engen was constructed by the superposition of the intrinsic vulnerability and hazard maps, which used an individual assessment scheme based on an individual classification. The assessment scheme is explained in Chapter 6 in Part A of this report, and a further application to the Sierra de Líbar is given in Part B, where the classification of the risk map is discussed in detail.

2.3.4.2 Risk assessment

The risk assessment scheme used for this map is based on the intrinsic vulnerability map constructed using the PI method (GOLDSCHEIDER et al. 2000) and the hazard map (see previous sections) and focuses on risk assessment for the groundwater resource. By overlaying the vulnerability and hazard maps and multiplying the specific assessment value of the vulnerability by the assessed value of the hazard a new value is calculated. This new value describes the risk of groundwater contamination dependent upon the hazard characteristics and the nature of the pathway to the groundwater that is given by the vulnerability map.

Using the equation:

$$R = 1/HI \cdot \pi$$

where
R : risk value
HI : Hazard Index
π : PI-Factor (vulnerability)

the proposed risk assessment is classified taking into account the classes of the vulnerability and the hazard index (Tab. 32).

Tab. 32: Classification of the risk map concerning the classes of the vulnerability and the hazard map.

π - factor	Hazard Index	1/HI	π • (1/HI)	Risk Class	Risk Level	Colour
4 - 5	0 - 24	> 0.042	> 0.167	1	no or very low	blue
3 - 4	24 - 48	0.042 - 0.021	0.167 - 0.063	2	low	green
2 - 3	48 - 72	0.021 - 0.014	0.063 - 0.028	3	moderate	yellow
1 - 2	72 - 96	0.014 - 0.010	0.028 - 0.010	4	high	orange
0 - 1	96 - 120	< 0.010	< 0.010	5	very high	red

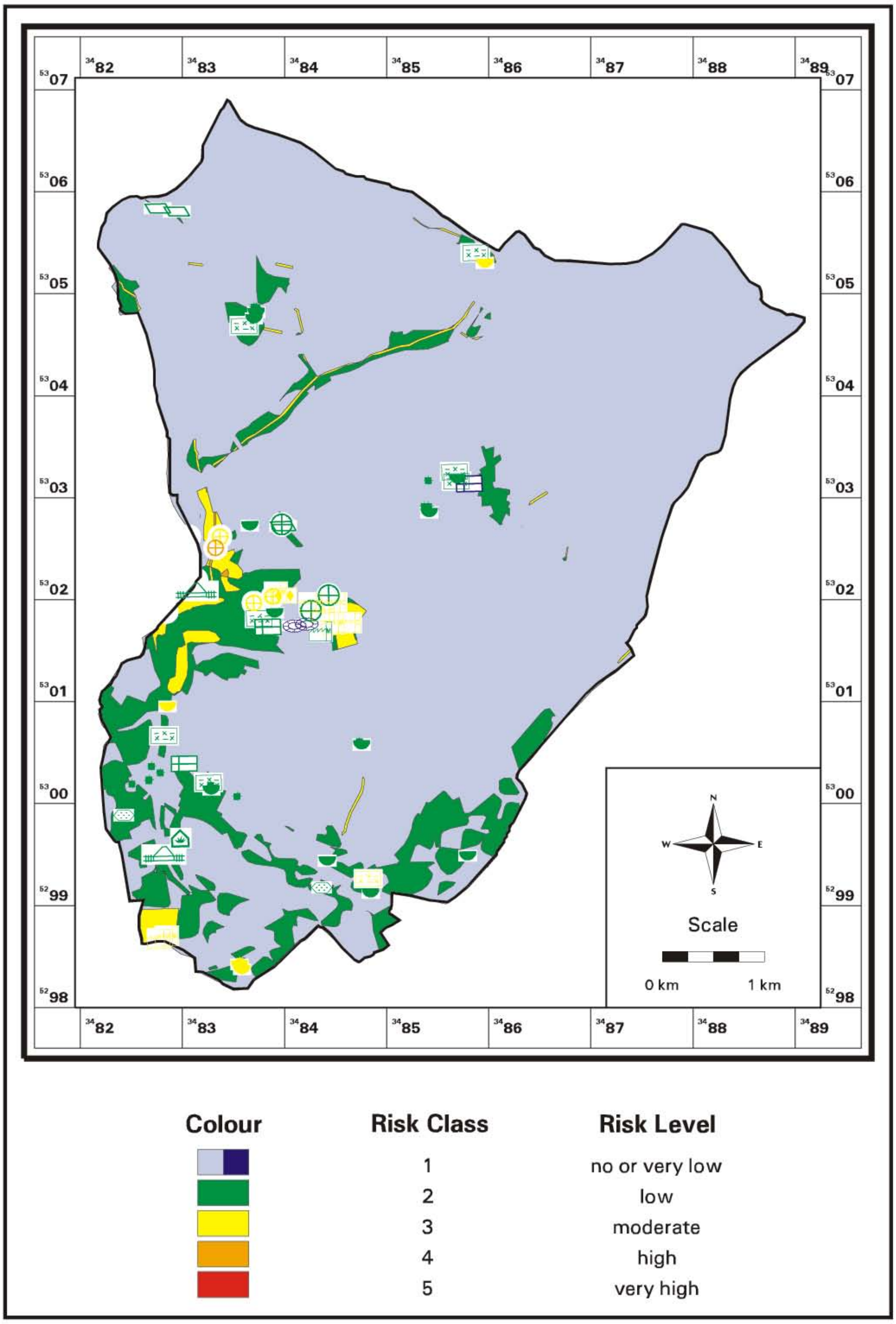

Fig. 71: Risk map of the Engen test site generated by overlaying the vulnerability map (PI method) and the hazard map. The legend for the hazard point symbols is given in Fig. 68.

Following this classification, a hazard classified as low or very low could produce a "high" or even a "very high" risk to groundwater if the vulnerability is very high (Fig. 71). In contrast, a hazard classified as high does not produce a "very high" risk when vulnerability is low as the risk classification in this case depends strongly on the vulnerability.

In the risk map of the Engen test site, point hazards kept their original spatial extension whereas linear hazards (roads and railway lines) were treated with a 10 m buffer to take into consideration the real spatial dimensions of such objects. Furthermore, the graphical interpretation of point hazards is much easier if the symbols from the hazard maps are adopted.

2.3.4.3 Results

The risk map of the Engen test site shows mainly zones of “very low” and “low” risk. “Moderate” and “high” risks are only locally present and “very high risk” is absent (Fig. 71).

If we look at the percentages of the risk distribution we find that the major part of the risk classes are occupied by class 2, which means that under the existing vulnerability conditions of the test site around 57% of all hazards represent a “low” risk to groundwater (Fig. 72). Approximately 26% of the hazards are considered to cause “moderate” risk and only around 4% represent “high” risk levels. It is noticeable that 14 % of the hazards cause “no or very low” risk level but they occupy around 86 % of the test site surface. Therefore the risk map is in general characterised by “very low” to “moderate” risk levels. However, some differences in risk distribution are present. While the northern part contain mainly “low” risk levels the western and southern region indicate hazards with “moderate” to “high” risk levels. As the hazard level varies only slightly across the region while the vulnerability varies significantly, the variations in risk levels are mainly due to the higher vulnerability in the western and southern part of the test site.

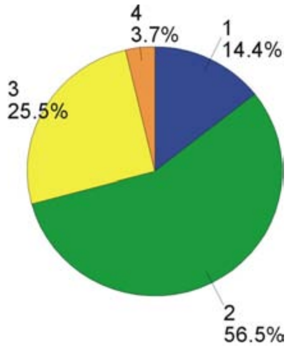

Fig. 72: Pie chart of the percentages of risk classes in the Engen test site.

2.3.4.4 Conclusion

Characterised by “low” and “moderate” risk levels, the risk map of the Engen test site shows that the problematic areas with relatively high risk levels are located in the western and southern part. Large areas are classified as “very low” due to the absence of hazards but also due to low vulnerabilities. These areas could consequently be interesting for future development as they are preferable in view of ground water protection. New land use developments can easily be checked for problematical risk levels by constructing potential risk maps, also at a detailed scale, to assess the risk to groundwater contamination of these planned activities.

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2.4 Vaulion test site, Jura Mountains, Switzerland

– Intrinsic resource and source vulnerability mapping using the 1D VULK simulation tool; Specific vulnerability mapping for resource protection for different types of contaminant –

2.4.1 Introduction

Intrinsic vulnerability assessment using the simple 1D "VULK" transport solver (Jeannin & al. 2001) was tested on a small hydrogeological basin in the western part of Switzerland. Although the development of this quantitative method has not been fully completed at the time of the writing, preliminary results are presented in this contribution. The C factor of the European approach (Daly et al. 2002) taking into account runoff processes has not yet been considered. Nonetheless, this approach suggests it has significant potential as a tool to check, and better constrain, the empirical vulnerability mapping method on well-documented test sites. The current project investigating this subject (Swiss national fund FN-20-68066.02) will further refine the development of this technique, and ultimately propose a fully operational vulnerability mapping method.

Specific vulnerability maps for resource protection for individual contaminants in an agricultural area have also been compiled. The implementation uses the available intrinsic map in conjunction with attenuation maps. These latter maps are derived using the specific vulnerability method, which has been developed within the framework of COST Action 620. The Vaulion hydrogeological basin acted as test site for the development of the method, and permitted verification of both the applicability of the approach and the plausibility of the first results.

2.4.2 Geological and hydrogeological setting

The Vaulion test site is located in the faulted region of the Jura Mountains in the western part of Switzerland (Fig. 55). The area of interest is a small syncline of Cretaceous limestone and marls (Fig. 73). The upper part of the aquifer consists of Baremian limestone including thick-bedded Urgonian at the top and smaller-bedded lower Barremian (Aubert & Dreyfuss 1963). Yellowish Hauterivian limestone constitutes the lower part of the aquifer, defined at its base by a 20 m thick impermeable marl layer. These Hauterivian marls delineate a 3.5 km^2 hydrogeological basin feeding five main springs located in a small valley. Three of these springs are used by the municipality of Vaulion as a drinking water supply. Tracing tests and water balance calculations suggest that most of the water infiltrated on this area is discharged by the springs, which excludes significant exchange with underlying Vallanginian or Malm aquifers (Perrin 2002). Karstification is well developed within both Baremian and Hauterivian limestones, as shown by development of caves up to 1 km long. Three successive karstic networks have been created by the progressive lowering of the karstification with time, and are linked to the presence of marl layers (Wittwer 1990). The Urgonian network is a fossil system while lower Barremian and upper Hauterivian networks are active. The structure of the syncline and speleological observations imply that the system is a shallow karst.

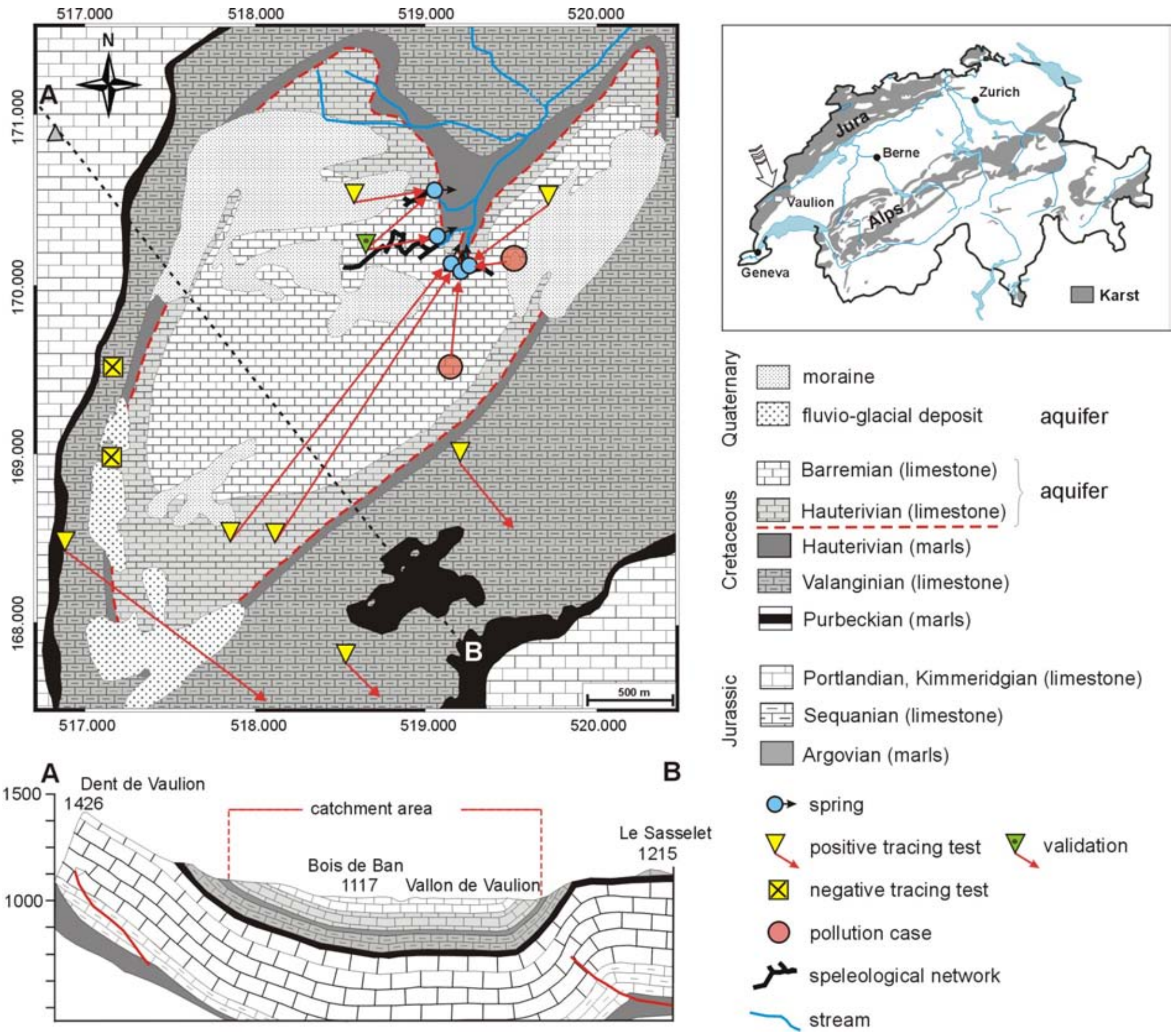

Fig. 73: Geological overview and location of the Vaulion test site. Cross section modified after Aubert & Dreyfuss 1963, geological map compiled after Aubert & Dreyfuss 1963, Custer & Aubert 1935 and Digout 2002

Non consolidated Quaternary sediments consisting of moraine and fluvio-glacial deposits cover about 20% of the area and reach a thickness of up to 10 meters. The fluvio-glacial deposits in the southeastern part of the basin are composed of sand and gravel and contain small perched aquifers while the moraine is more loamy and less permeable.

2.4.3 Intrinsic vulnerability

2.4.3.1 Methodology

The use of the quantitative based VULK tool is proposed as an alternative to vulnerability methods using matrix and relative weighting. VULK first requires schematisation of the hydrogeological system into a multi-layer conceptual model (e.g. subdivision of the O factor of the European approach, proposed by Daly & al. 2002). For each layer, thickness, velocity, dispersivity, and dilution parameters must be evaluated for the whole area investigated. Calculations are made either by using a single or a double porosity assumption (Jeannin & al. 2001). VULK provides the breakthrough curve for an input of concentration = 1 and of any duration, by simulating the transport through a layer. The curve is then used as input in the following layer and so on. The response to a contamination event can be thus evaluated in terms of transfer time, attenuation and duration, either for the resource or for the source. A more detailed overview of this approach is presented in this volume (Part B, section 1.3).

Evaluation of each of the model parameters requires detailed knowledge of the hydrogeological setting. Detailed test site studies, including numerous tracer tests as calibration or verification and sensitivity analysis are performed at present to refine this approach.

2.4.3.2 Field data

The conceptual model for the Vaulion test site consist of four layers based on the classification according to the European approach, with subdivision in topsoil, subsoil, unsaturated karst (corresponding to the O factor) and saturated karst (K factor):

- Topsoil and subsoil have been mapped in terms of thickness and groundwater velocities (Digout 2002). Manual drilling and geophysical radio magneto telluric measurements have provided an accurate evaluation of these parameters for the whole study area (Fig. 74). Topsoil thickness ranges between 10 and 100 cm (four classes), and subsoil between 1 and 10 meters (six classes). Two classes of velocity are considered for the topsoil (0.5 for brown soil and rendzina and 0.1 m/hour for peaty soil) and for the subsoil (0.01 m/h in the moraine and 0.02 m/h in the fluvio-glacial deposits). These values are based on observations of the structure and composition of these formations and on infiltration tests performed in this region. The velocities considered are, however, close to saturated conditions and have to be regarded as worst-case scenarios. Dilution factors are only considered in fluvio-glacial deposits (value = 1/20), which act as small perched aquifers.
- Unsaturated karst thickness is deduced by subtracting the base level of the aquifer from the topography (NMT) and by also accounting for the presence of Quaternary deposits. The thickness of the unsaturated karst ranges from 10 to 70 meters and four classes are considered (Fig. 74). Velocities in the unsaturated karst are assigned based on karstic geomorphology observations and data collected on other test sites where tracing tests have been performed in the unsaturated zone. Karstification is, in general, well developed with karrenfields frequently present. However, few dolines or locally very karstified structures are observed, which accounts for relatively homogeneous velocities attributed to the whole area. Three classes of velocity are proposed: 20 m/h for strongly karstified zones such as dolines and depressions, 15 m/h for the inner part of the syncline, and 20 m/h for the outer part of the syncline.
- The parameters required for the saturated karst are distance to the spring, velocity, and dilution factor. Distance to the spring is calculated by taking flow toward the synclinal axis into account. Velocities assigned to the saturated zone are 100 m/h in the center of the syncline and 50 m/h on the outer part. These values reflect relatively high water level

conditions, since velocities could be up to ten times slower in low water conditions as demonstrated by tracing tests. The dilution factor for the five sub-basins is deduced from the spring discharge data, by taking into account relatively high water conditions (about 1.5 times the mean annual discharge). Providing the absence of significant drowned saturated karst (see section 2.4.2) dilution factor is estimated by the ratio of an injection of 1 m^3/h of water on the water catchment area to the discharge of the spring. This implies a certain type of input scenario that will influence the transit time and the attenuation factor calculated.

Results presented are not a direct vulnerability index, but a value for transit time and attenuation. The parameter of duration of a contamination, which could be also of interest in the definition of vulnerability (part A, section 2.2) is not taken into account in this example but could be easily determined on the basis of the breakthrough curve, calculated with the VULK code.

2.4.3.3 Map processing

Combination of the maps of thickness or distance and velocities for each layer has been performed with using a GIS (Fig. 74). Transfer time to the source and resource (tt = d/v) can be easily calculated.

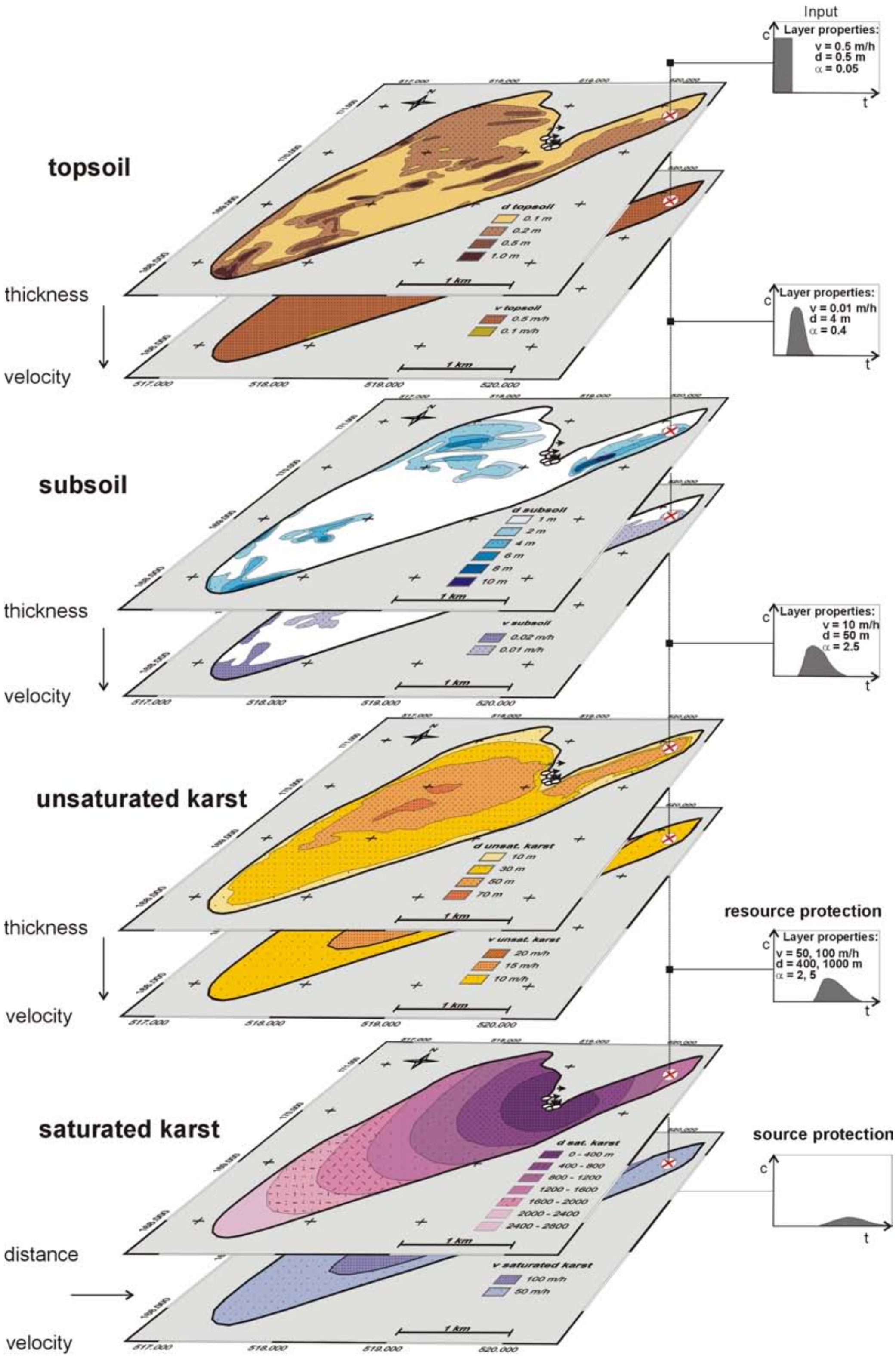

Fig. 74: Combination of velocity, thickness and dispersivity layer properties used for the VULK calculation to assess resource and source vulnerability.

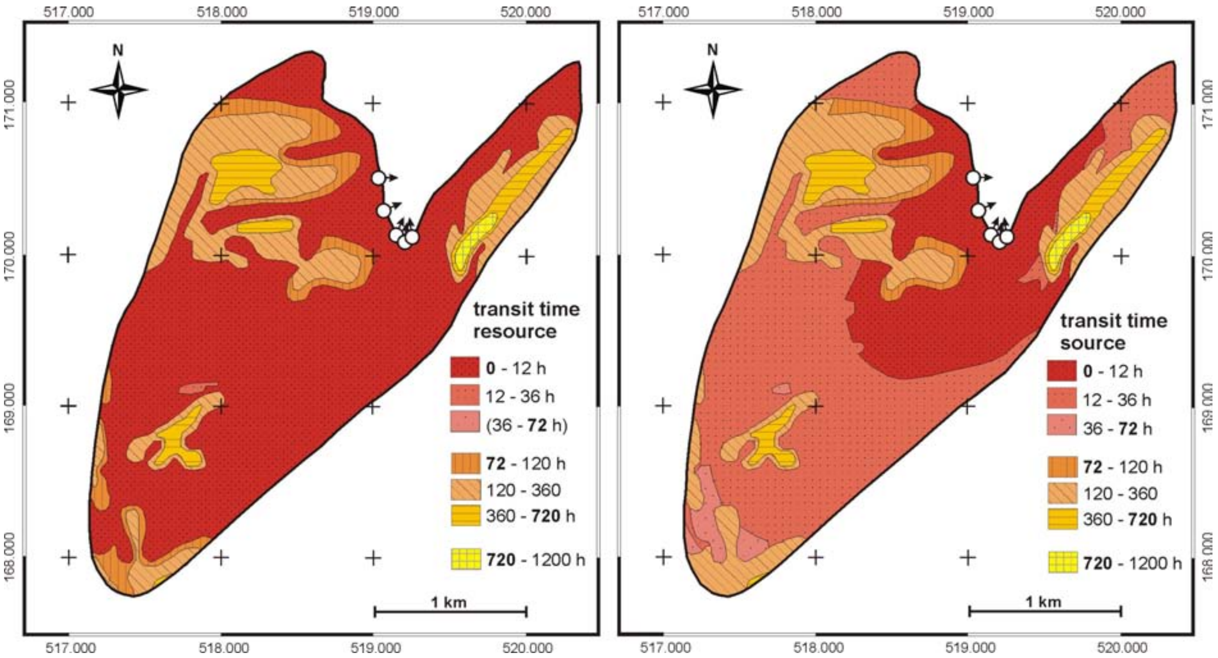

Fig. 75: Transit time maps for resource and source.

Classes for transit time (3 days, 30 days, 1 year,...) proposed by COST Action 620 (part A, section 2.2) are subdivided in three sub-classes in order to better illustrate the contrasts on the map (e.g. 12, 36 and 72 hours for the 0 to 3 days class).

Simulations with the 1D VULK tool have been performed for each polygon created by the intersection of the layers (Fig. 74). As a first step, a single porosity assumption was adopted and dispersivity coefficient (α) was considered function of thickness or distance (d) of the layer ($\alpha = 1/10*d$ for topsoil and subsoil and $\alpha = 1/20*d$ for unsaturated and saturated karst). This assumption is relevant considering scale effect in geological media and its influence on transport processes (Neuman 1990, Gelhar & al. 1992). It has to be pointed out that a variation of the α coefficient by a factor 5 will only modify attenuation by about 10%, which mitigates the uncertainty on this parameter. Results of the simulation using VULK have been introduced in the GIS to draw attenuation map for the resource (Fig. 76).

The attenuation factor, corresponding to the inverse of the relative concentration of the breakthrough curve maximum, is adopted to present the results. Classes proposed by COST Action 620 (section 2.2) are divided here in two sub-classes (e.g. 1-4, 4-10 for attenuation factor class 0-10, corresponding to a relative concentration of 1 to 0.1).

As the limits of the sub-basins catchments have not been delineated accurately enough so far, the attenuation map for the source is not presented. Forty points that can be attributed with certainty to one of the sub-basins have been calculated and plotted on a graph presenting the relationship of transit time with attenuation (Fig. 76). Red arrows (points A and B) highlight that the decrease of vulnerability in the saturated zone is mainly due to attenuation.

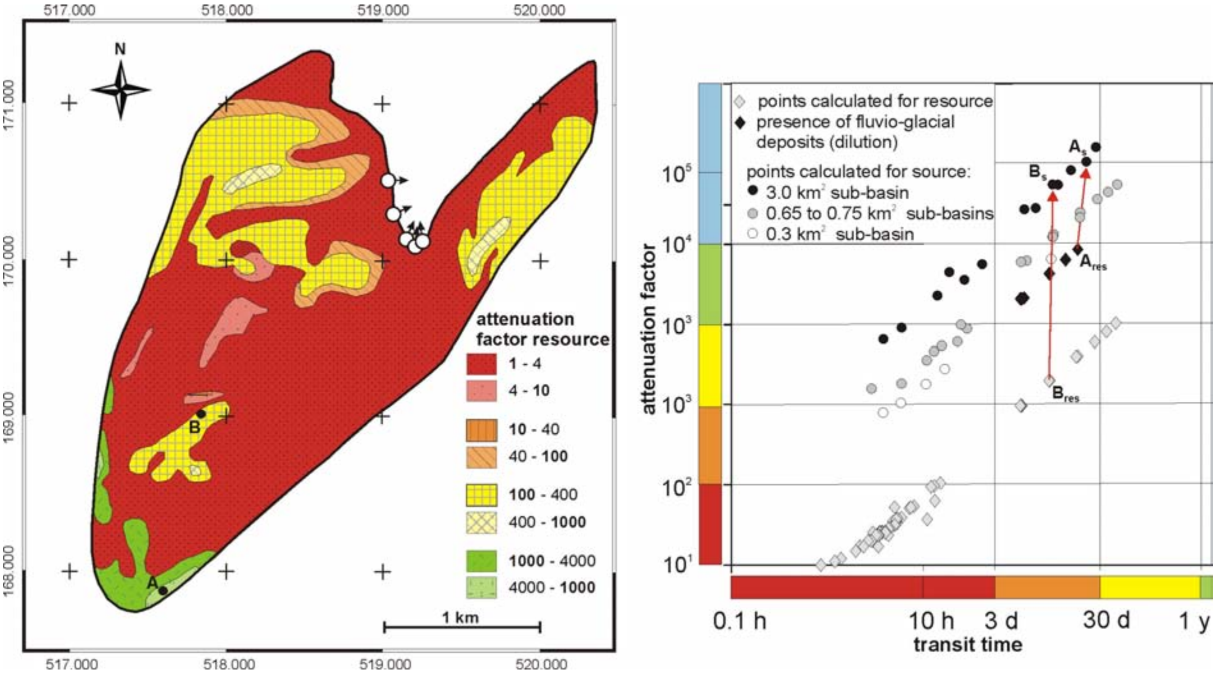

Fig. 76: Attenuation map for resource. Correlation between transit time and attenuation for the calculations made for resource and source.

2.4.3.4 Results

Transit time maps show the critical importance of Quaternary deposits compared to the other layers (Fig. 75). Relatively thin soil and unsaturated karst as well as strong karstification considered on the whole area account for very rapid flow, determined for the rest of the basin. The influence of saturated zone is limited with maximum transit time of 40 hours in this layer. Minimum transit times are 1 hour for the resource, respectively 3 hours for the source and maximum values are 1003 and 1015 hours. However, the velocity parameters used reflect a worst-case scenario, and ongoing investigation could result in less pessimistic values, being applied in some areas.

As expected, attenuation for resource is strongly correlated with transfer time, as long as no significant dilution occurs. Resource intrinsic vulnerability could therefore either be assimilated into the results of an attenuation map, or transit time map, as a first step. For significant dilution, this case no longer applies as reflected by high attenuation values determined for the perched saturated fluvio-glacial deposit in the southwestern part of the catchment (Fig. 76). Concerning source attenuation, dilution accounts for a significant decrease in concentration, depending on the discharge of the spring and thus on the size of the sub-basin (Fig. 76).

2.4.3.5 Conclusions

Vulnerability assessment using a 1D transport tool has been shown to be possible without requiring significantly more data than empirical methods. The attribution of discrete values for velocity and dispersivity within each layer may seem problematic and to suggest significant potential error at first, but this approach has the advantage of clearly showing uncertainties that are inherent and not quantifiable using empirical methods.

As surface run-off is limited, and no sinkhole is observed within the Vaulion test site, the use of a provisory approach not accounting for surface hydrology is relevant.

A vulnerability index is not proposed so far, but only transit time and attenuation maps. A high correlation exists between these two parameters, except for the case of significant dilution. This allows, at the moment, consideration of both maps, and preferably using the transit time map to reflect resource vulnerability.

The high intrinsic vulnerability values suggested are logical, since intrinsic vulnerability does not reflect a mean behaviour of pollutants, but only the non-realistic purely conservative transport of a substance not influenced by any additional specific attenuation process. Only by taking into account specific properties of contaminants can an intrinsic map be turned as an efficient predictive tool for water management.

2.4.4 Specific vulnerability

2.4.4.1 Methodology

The Vaulion test site was also used for the application of the specific vulnerability method, which has been developed in the framework of COST Action 620 described in this volume (see section 4.6 of part A). The qualitative evaluation procedure provides one specific attenuation map for each contaminant concerned. These maps, representing the S factor of the European approach, are combined with an already existing intrinsic vulnerability map. In the present case, the VULK resource transit time map was used as intrinsic vulnerability map. Together, they portray resource specific vulnerability maps (Fig. 77).

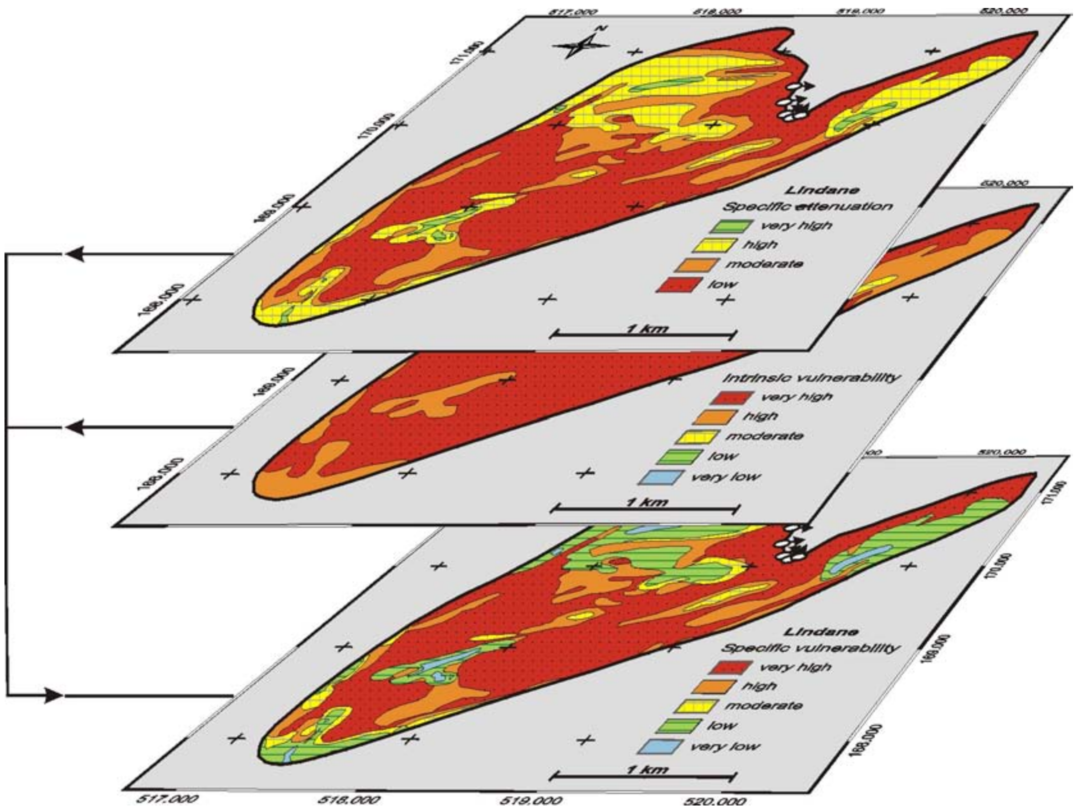

Fig. 77: Superimposition of specific attenuation and intrinsic vulnerability maps for preparing specific vulnerability maps.

The attenuation classes of the specific assessment are linked to the intrinsic vulnerability classes according to the scheme displayed in Fig. 78. Significant specific attenuation increases groundwater protection and downgrades vulnerability. For example, a specific attenuation classified as moderate, according to the applied method, can reduce vulnerability by one class. High specific attenuation downgrades vulnerability by two classes, and the protective effect of very high specific attenuation may decrease vulnerability up to three classes. In contrast, a low specific attenuation effect does not lead to a change of vulnerability. It is obvious, that the intrinsic VULK map, here representing the O factor of the European approach, cannot maintain its quantitative character. Instead, it has to be converted into qualitative classes before being merged with qualitative specific data.

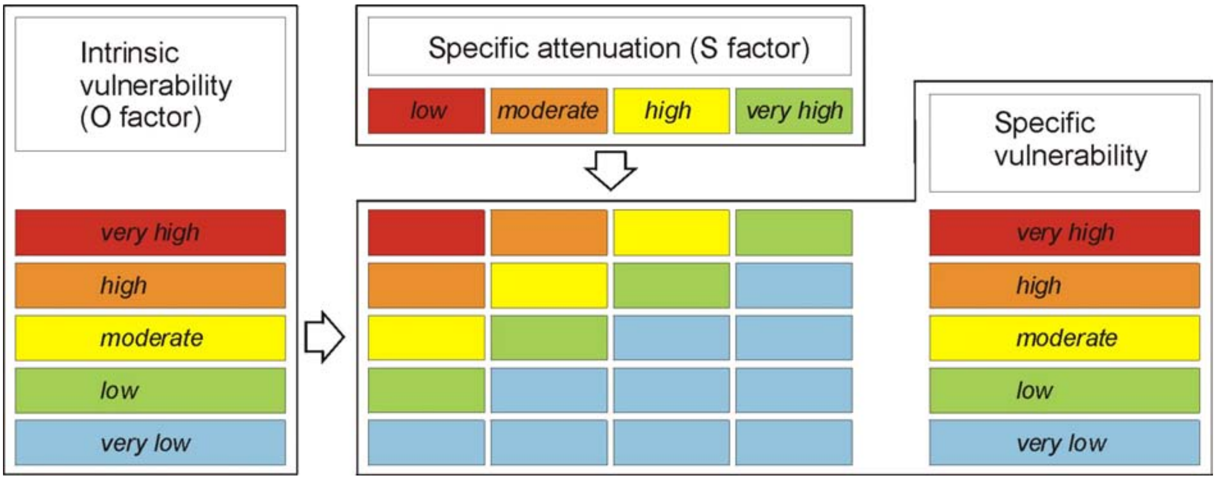

Fig. 78: The link between VULK transit time classes and specific attenuation classes for preparing specific vulnerability maps.

The specific assessment requires information about the nature of the contaminants concerned, and also additional field data. The collected contaminant and layer characteristics are transformed into parameter scores according to the property ranges given in the method guidelines.

2.4.4.2 Contaminants

Four individual substances were selected for vulnerability mapping based on the agricultural cultivation in the basin. They represent contaminant groups, which frequently cause problems for karst groundwater quality: Pesticides, fertilisers and microbiological contaminants.

- Atrazine is a widespread pesticide of the triazine group and its use in Switzerland is only permitted for maize production. It is not significantly attenuated in the subsurface, and consequently is often found in groundwater. This is due to its low sorption potential to both organic and clayey material and to moderate biodegradation half-lives.
- Lindane belongs to the group of chlorinated hydrocarbon pesticides. It is no longer deployed in Switzerland, but still used in many other countries. Lindane displays a higher attenuation potential than atrazine, having a moderate sorption potential in conjunction with moderate biodegradability. Under reducing conditions, lindane may even be highly biodegradable. Both pesticides are non-volatile.
- Nitrate is a very common fertiliser. It is often regarded as a contaminant displaying quite conservative behaviour in the subsurface. However, nitrate may be attenuated due to its high reduction potential.
- Cryptosporidia are pathogenic protozoa that may be derived from livestock or farmland. They are relatively persistent and may survive in the subsurface for months or up to a year. Due to its size, cryptosporidium is poorly qualified for adsorption. However, its size allow for filtration processes.

2.4.4.3 Field data

Layer parameters were estimated based on field observations and general standard layer characteristics. The three existing layers (topsoil, subsoil, unsaturated karst) show different physical-chemical and hydraulic properties, resulting in contrasting vulnerability for each specific map.

- The topsoil layer has high organic matter content and medium clay content. Some preferential flow may occur due to vegetation and superficial cracks, but diffuse flow remains the predominant infiltration process. It is for this reason, that the topsoil was rated as oxygen-rich, apart from some spots of peaty soil, where reducing conditions are assumed. The topsoil matrix is suspected in possessing medium pore diameters.
- The subsoil sediments consist of material with low organic matter content and medium (fluvio-glacial) to high (moraine) clay content. The higher clay content also accounts for a lower matrix aperture in the moraine (medium) than in the fluvio-glacial deposits (high). Diffuse flow dominates in both subsoil types. Balanced redox conditions are assumed.
- The unsaturated karst is highly karstified with an important preferential flow regime, which favours high dissolved oxygen content. Diffuse flow in medium sized matrix pores is a subordinate part of total flow. Neither organic matter content nor clay content is significantly high.

2.4.4.4 Specific vulnerability maps

The specific vulnerability assessment was performed by means of a GIS and results in different specific vulnerability maps for the four observed contaminants (Fig. 79 and Fig. 80).

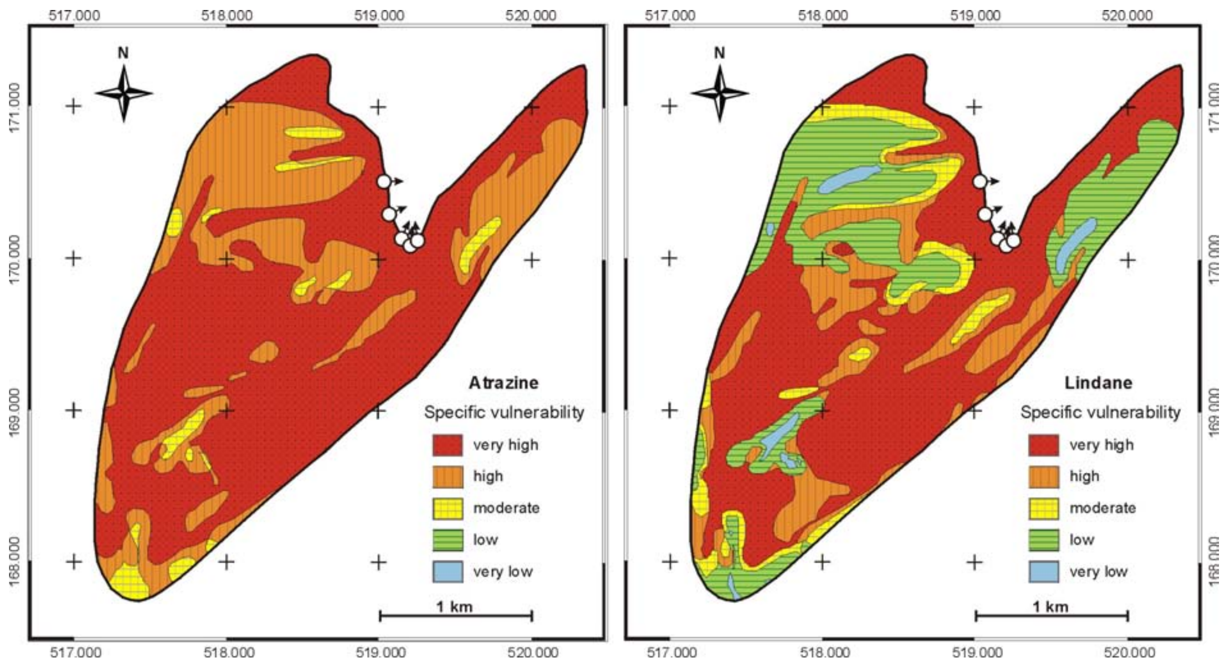

Fig. 79: Atrazine and Lindane specific vulnerability maps for karst groundwater resource protection.

A comparison of the two pesticides shows that the atrazine specific map generally has a higher vulnerability than lindane specific map (Fig. 79). Atrazine is attenuated only if there is a thick topsoil layer of at least 0.5 m, which is assumed to result in a partial loss by biodegradation. Subsoil and karst layers do not contribute significantly to atrazine attenuation due to the lack of organic material. In contrast, groundwater protection against lindane contamination benefits from adsorption processes in both topsoil and subsoil, as well as biodegradation in the topsoil. Lindane specific vulnerability is thus assessed as being reduced in zones underlain by Quaternary deposits, but also in zones with a soil cover of at least 0.2 m. The atrazine vulnerability map differs only slightly from the intrinsic vulnerability map, due to the unfavourable attenuation characteristics of this compound. On the other hand, due to a greater degree of attenuation, the lindane vulnerability map differs more notably from the intrinsic vulnerability map.

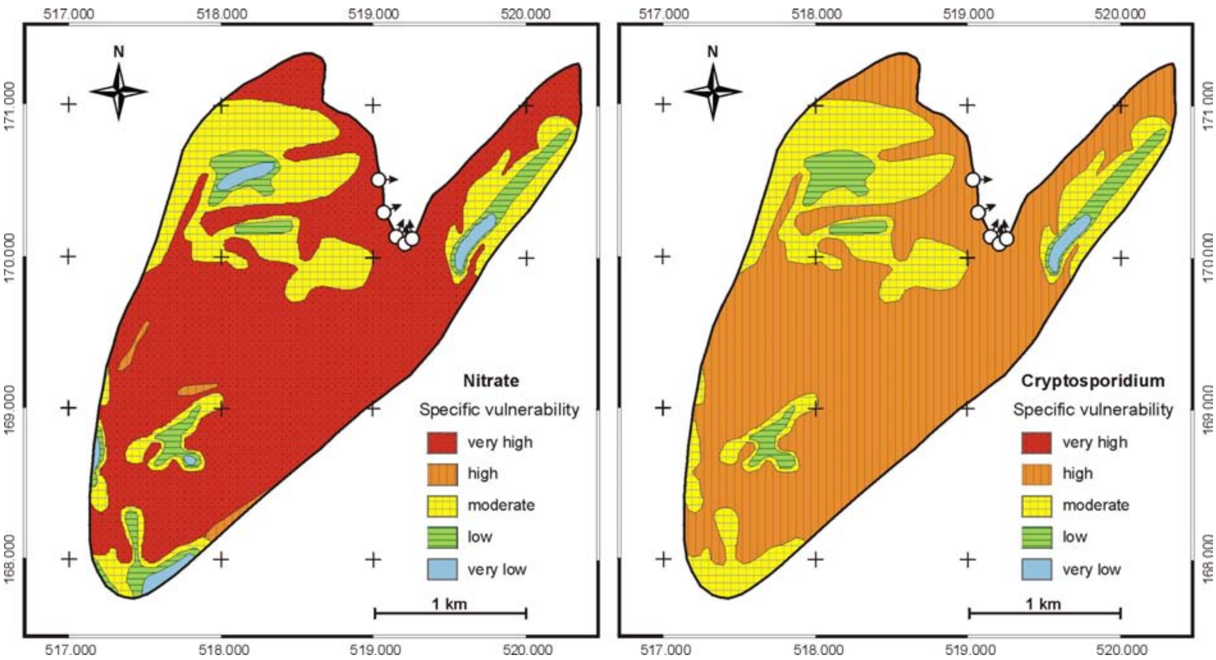

Fig. 80: Nitrate and Cryptosporidium specific vulnerability maps for karst groundwater resource protection.

The specific behaviour of nitrate is determined by its ability to be reduced in layers with relatively low oxygen content. This is suggested for moraine and fluvio-glacial deposits (Fig. 80). The topsoil supports reduction only in the peaty soil zones, which have a high oxygen deficit. Since reduction is the only relevant process for nitrate loss, the specific map is a combination of the intrinsic map and a reduction process map.

Filtering in the matrix of topsoil, moraine subsoil and unsaturated karst influences the fate of Cryptosporidium. Neither significant adsorption in spite of present clayey and organic material, nor sufficient die off is evaluated, due to the suggested properties of this microbiological contaminant. Hence, cryptosporidium do not suffer appreciable attenuation by specific processes other than filtration. The corresponding specific map (Fig. 80) is a reflection of the distribution of layers with important diffuse flow through fine-grained media (topsoil, moraine). However the karst layer is also taken into account by a small additional filtration contribution, although this is limited to the subordinated matrix flow in the limestone.

The Vaulion catchment as a whole is more vulnerable to atrazine and nitrate than to lindane with particularly high contrasts notable between the central karst zone and areas containing Quaternary sediment cover in the northern and southern part of the basin. Cryptosporidium shows a more even vulnerability distribution.

2.4.4.5 Conclusions

The applied specific vulnerability method accounts for both the layer properties and the characteristics of individual contaminants. Three parameters determine process effectiveness, and associated specific attenuation evaluation, namely the contaminant process indices, the layer process indices and the rate of diffuse flow. The latter parameter may become important to restrict process effectiveness within thick layers with appreciable bypass by preferential flow component (e.g. filtration in the karst layer). However, it is normally not the limiting factor for relatively thin media, and/or media where flow is predominantly diffuse (here: topsoil, subsoil). This makes the method applicable to a wide variety of karst settings, and generally to all kinds of contaminants, thus permitting meaningful specific vulnerability maps to be produced.

The specific vulnerability assessment of the Vaulion basin provides final maps of different vulnerability distribution patterns, highlighting the activation of different processes depending on the contaminant concerned. Furthermore, they generally show the effect of a topsoil and subsoil cover on the protection of karst groundwater to contamination, and the lack of attenuation processes where these layers are missing. However, even suitable layers may become insufficient, if they do not interact with reactive and/or degradable contaminants. Conversely, the mapping results illustrate that a pronounced soil layer over a karst system may already be responsible for substantial attenuation. Intrinsic assessment does not display vulnerability decrease in such a situation often found in karst environments. Overall, specific vulnerability mapping can help to compile a more differentiated image of contaminant attenuation and to improve the compatibility of landuse and groundwater protection strategies.

2.4.5 Tracer testing

A tracer test was performed injecting several tracers. The three tracers used for validation of the intrinsic and specific approaches were uranine, sulforhodamine B and nitrate. However, this first validation test must be considered carefully, as it was realised during low water, implying low velocity conditions in the saturated zone, and significant water deficit in the unsaturated zone.

10 m^3 of tracer solution were injected by sprinkling over a 40 m^2 karrenfeld area covered by about 10 cm of soil. Total simulated precipitation was 250 mm over a 7 hours period, corresponding to a rainfall intensity of 36 mm/h. Such conditions are not realistic but are not believed to have created unreasonable hydraulic conditions within the unsaturated zone. The tracer mass injected was calculated by taking into account tracer molarity in order to obtain comparable number of molecules for each tracer.

A flow towards two springs was observed as shown in Fig. 73. Results for the spring located further to the south, where the recovery was ten times higher, are presented (Fig. 81). The tracer recovery during the period of investigation was very low with 3 % for uranine, and 1 % for sulforhodamine B and nitrate. This can be partially explained by a significant storage in the unsaturated zone, especially in the epikarst. Moreover, diffuse injection will result in low tracer recovery relative to a concentrated injection (Perrin & al. 2002).

The tracer test shows contrasting results for the three injected substances in terms of contaminant specific behaviour. Uranine, regarded as a nearly conservative tracer, may be used for intrinsic vulnerability validation and VULK calibration. Reactive and/or degradable tracers are suitable to validate specific vulnerability maps. For instance, sulforhodamine B may represent a sorbable contaminant. Nitrate application data in combination with spring monitoring may provide valuable information about specific behaviour of this contaminant.

Simulation of the uranine curve using VULK code considering a restitution of 100% shows that the use of double porosity assumption is necessary to get an acceptable fitting. Velocity (15 m/h instead of 100 m/h) and dilution (0.3 instead of 0.018) parameters had also to be adapted to the low water conditions of the experiment.

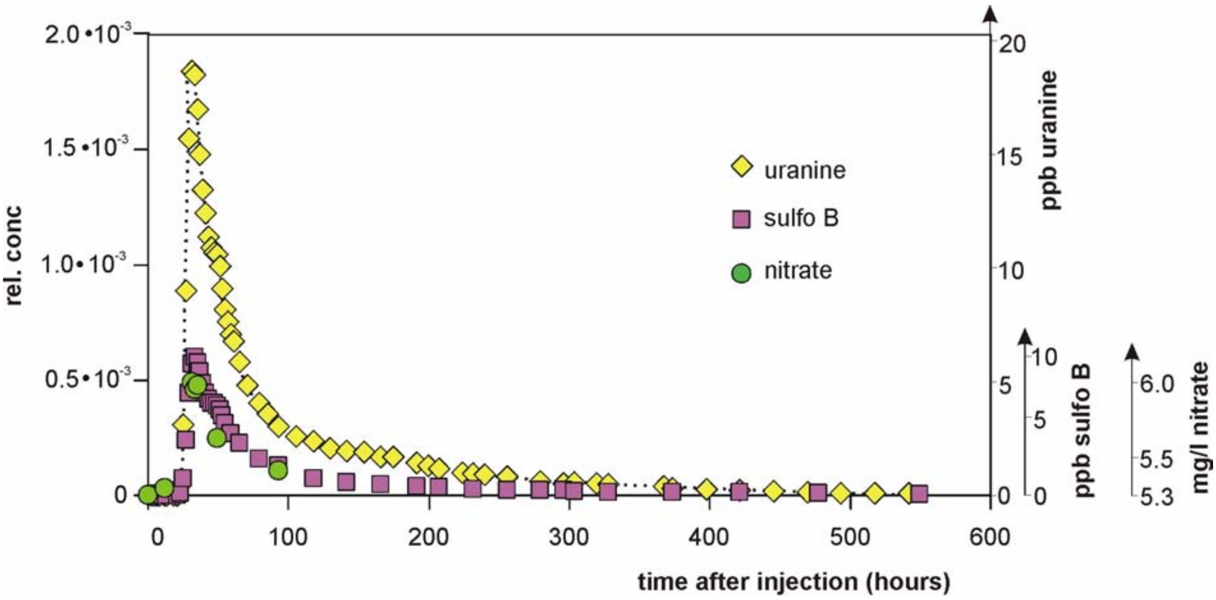

Fig. 81: Breakthrough curves of the tracer test.

2.4.6 Final conclusions and further development

The least vulnerable zones are generally situated where Quaternary subsoil deposits occur. This is the result of two influences: Intrinsic characteristics cause high flow transit time and hydrodynamic dispersion. On the other hand, many contaminant specific processes are considered as being likely to occur in this kind of material, causing retardation and degradation. Both methods account for these aspects and allow for the production of plausible maps of transit time or attenuation (intrinsic vulnerability) as well as of specific vulnerability.

The simulation of intrinsic contaminant transport in karst using 1D simulation tool gives very promising and coherent results for groundwater vulnerability assessment. Tracer tests provide good perspectives for a more accurate calibration of the model parameters. An additional module, taking into account surface contaminant transport by run-off (C factor), is under development and will make the method applicable to any type of hydrological setting. Another challenge is the implementation of the S factor into the VULK model, thus providing a tool for a quantitative specific vulnerability assessment. Retardation and degradation effects may be directly linked to transit time and attenuation factor. Classes of vulnerability based on transit time and attenuation factor have yet to be to be defined.

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2.5 Nassfeld, Southern Alps, Austria

– Comparative application of the new Austrian Approach (VURAAS) and the PI method of intrinsic vulnerability mapping, and hazard mapping

2.5.1 Geographical and Geological Overview

The VURAAS method (**Vu**lnerability and **R**isk analyses for **A**lpine **A**quifer **S**ystems) was applied in an Alpine karst region in the Southern Alps of Austria and compared with the PI method (GOLDSCHEIDER et al., 2000). The test site (about 8,3 km²) around the mountain Trogkofel is located in the Carnian Alps in the Skiing Region of Nassfeld (Fig. 55).

The altitude ranges between 1120 m and 2280 m within the test site. The average slope gradient is around 30°. The mean annual precipitation reaches some more than 2260 mm and the mean annual temperature is about 3,6 C. Snow cover lasts on average 193 days per year.

On the eastern part of the mountain, there are extensive winter tourism activities. New ski courses are planned and built. Anthropogenic impact on the environment is investigated with the help of the VURAAS method for mapping intrinsic vulnerability.

On the western part of the mountain, human influences can be neglected. The geological setting is relatively homogeneous on both sides; therefore the areas can be compared.

The geological formations predominantly consist of Permian limestone, which are underlain by schist and sandstones. The carbonates are fractured and slightly karstified. In some parts, scree or moraines overlie the hard-rock formations. At the contact of the limestone with the "low permeability rocks" many springs are situated at an altitude around 1600 m above sea level. Several (karst) springs are used for the public water supply.

2.5.2 Intrinsic Vulnerability – Comparison of the VURAAS and PI method

2.5.2.1 Overview

The VURAAS method includes the description and evaluation of the input factors (P), infiltration (I) and exfiltration (E) (CICHOCKI et al., 2001). The general concept is shown in the flow chart below (Fig. 82). The approach is based on existing methods but includes modifications for high Alpine karst regions. The weighting and rating of the parameters were adapted to the special features in the Alps. For instance, the heterogeneous geology and the complex tectonics require modifications of existing methods. Measurements at selected springs and creeks were included in the VURAAS method, which are suitable to enable more information to be obtained about the whole system. A significant weight was given to those parameters, which describe the hydrological system.

It has to be emphasized that only the results of the I-Map (VURAAS method) and the PI-Map (PI method) are shown here to compare each other. The maps for the input and the validation with the factor exfiltration of the VURAAS method are not taken into account.

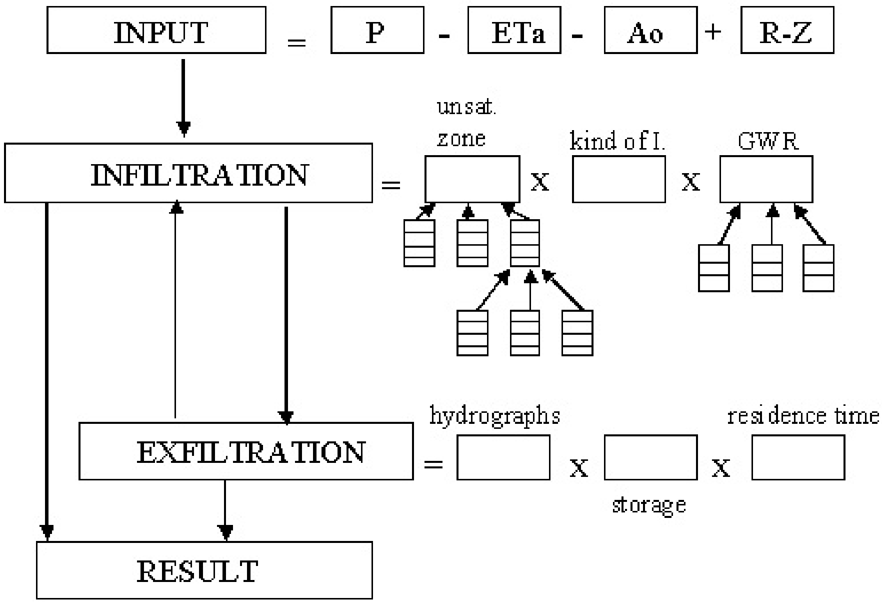

Fig. 82: Assessment concept of the VURAAS method

The external stress to the system is described by the factor P. It is described below to complete the methodology of the VURAAS method and the PI method. After that several parameters and data describing the core-factors O (overlaying layers) and C (concentration of flow) of the European Approach were either mapped in the field or developed from existing information. In the VURAAS method the factor I includes the core factors O and C. In the PI-Method the factor P is equal to the factor O and the factor I with C, respectively. Finally the karst network was characterised by the factor E when applying the VURAAS method.

2.5.2.2 Precipitation regime – P-map

The parameter P (VURAAS) was evaluated with the water balance equation. Therefore the equation is transformed as follows to obtain the amount of water that can infiltrate (= input P):

$$P = A - B - C + (R \pm \Delta S) \tag{1}$$

P:	input to infiltration water
A:	precipitation [mm/a]
B:	ETp (HAUDE 1954); landuse-map (SACCON 1999)
C:	surface and subsurface runoff (recession curve analysis)
$(R \pm \Delta S)$:	retention / conduction in or drainage out of the area and differences in the water storage in the unsaturated zone

Following the PI method the parameter R was evaluated by the water balance equation. As the recharge exceeds 400 mm/y, a multiplication factor of 0,75 was used in the equation of the total protective function (PTS).

2.5.2.3 Overlaying layers – O-map

The unsaturated zone is described in the PI method with the factor P. In the VURAAS method the unsaturated zone is included in the factor I (infiltration), beside the parameter A (type of infiltration), and the parameter GWN (groundwater recharge). The total points for the factor I are obtained by using the equation:

$$I = UZ \times A \times GWN$$

UZ : unsaturated zone
A : type of infiltration
GWN : groundwater recharge

In the VURAAS method, the unsaturated zone is described similar to the PI-method but with a proper rating and weighting system.

The type and thickness of the soils was obtained from 60 hand auger short holes (7-8 holes per km²). A representative profile for each type of soil was described and soil material was analysed for the calculation of the effective field capacity (eFC) and saturated hydraulic conductivity (Tab. 33). The taxonomy of soils of Austria was used for the nomenclature.

Tab. 33: Type of topsoil and calculation of points applying the VURAAS method and PI-method.

Type of topsoil	eFC [mm] up to 1m	thickness [m]	points/m VURAAS : PI	total points VURAAS : PI
Staupodsol	79	0,6	0,8 : 50	0,8 : 50
pseudovergleyter Kalkbraunlehm	38	0,2	0,8 : 10	0,72 : 10
Pseudovergleyter Podsol	46	0,5	0,5 : 10	0,5 : 10
Anmooriger Nassgley	166	0,3	1 : 125	0,9 : 125
Rendzina	36	0,15	0,5 : 10	0,45 : 10

Seven subsoil types of unconsolidated material can be distinguished in the VURAAS method (Tab. 34). The thickness of the subsoil was estimated by geological cross sections and the type of subsoil was mapped in the field. Within the test site, the very low vulnerability moraines can be differentiated from the very highly vulnerable block deposits.

Tab. 34: Type of subsoil and calculation of points applying the VURAAS method and PI method.

Type of subsoil	points/m VURAAS : PI	thickness [m]	total points VURAAS : PI
block deposits and scree of limestone	0,7 : 5	5-10	0,7 - 25-50
scree of limestone	1 : 10	1 - 20	0,9-1 : 10-200
scree (no carbonat)	2 : 90	1 - 5	1,8 : 90-450
quaternary deposits with block deposits	0,75 : 75	5-20	0,75 : 375-1500
side moraines, ski-runs, alluvial deposits	3 : 120	5-20	3 : 600-2400

The lithology was classified in 5 different classes (Tab. 35). The geological units and the structural geology were mapped in the field. For the non-karst bedrock and unsaturated karst bedrock, several cross sections were used to estimate the thickness of each layer. Some data are taken from the literature. The structural geology was characterised by field mapping and interpreting aerial images.

Tab. 35: Type of non-karst and karst rocks and calculation of points applying the VURAAS and PI method.

Type of non karst+ karst rocks	points/m VURAAS : PI	thickness [m]	total points VURAAS : PI
sandstone	1,2 : 15	10	1,08 : 150
claystone - sandstone	1,4 : 20	70	1,26 : 1400
claystone	1,4 : 20	200	1,4 : 4000
limestone sligthy karstified	0,8 : 5	70-100	0,72 : 350-500
limestone karstified	0,4 : 2,5 (L*F)	100-400	0,4 : 250-1000

Tab. 36 includes the score ranges of the unsaturated zone and the protective function (PTS) of the overlaying layers: The scores in the VURAAS method vary from 0,5 to 1,25 points, which means that the area can be divided into 4 different classes of vulnerability (very low to high effectiveness of the protective cover). The scores range between 56 to 3712 points when applying the PI method. The effectiveness of the protective cover is consequently classified between "low" and "high".

Tab. 36: Effectiveness of the protective cover.

Score VURAAS method	Score PTS	effectiveness of the protective cover	P - factor
<2,5 = 0,5	0-10	Very low	-
2,6-5,0 = 0,75	>10-100	Low	2
5,1-7,5 = 1,0	>100-1000	Medium	3
7,6-10,0 = 1,25	>1000-10000	High	4
>10,1 = 1,5	>10000	Very high	-

2.5.2.4 Concentration of flow - C-map

Applying the PI method, the I-map was created reflecting the degree of bypassing the overlying strata due to surface and subsurface flow processes.

In the VURAAS method, an infiltration map is produced which includes the O-map and C-map of the proposed European Approach. In addition to the description of the unsaturated zone, one layer was set up showing the type of infiltration – point, linear and planar infiltration are distinguished.

Furthermore the groundwater recharge (GWN) was estimated by describing the water retention on the surface, the density of springs and the characterisation of the water storage in the underground. On one hand, the values for the GWN were obtained by empirical observations during the field mapping; on the other hand methods from isotope hydrology and hydrogeochemistry were applied for the assessment of the storage capacity.

2.5.2.5 OC-map

A GIS-based production of the vulnerability map was performed showing the core factors O (overlaying layers) and C (concentration of flow) of the European Approach or P and I respectively (PI method) in the test site.

Tab. 37: Classification of the vulnerability applying the VURAAS method and the PI-method

I - Map (VURAAS method) represents the OC - Map of the European Approach		P-Map (PI method) Effectiveness of Protective Cover P-factor		I-Map (PI method) Degree of Bypassing I-factor	
Extreme	**0-1**	Very low	-	Very high	-
High	**>1-2**	**Low**	**2**	High	-
Moderate	**>2-3**	**Moderate**	**3**	**Moderate**	**0.6**
Low	**>3-4**	**High**	**4**	**Low**	**0.8**
Very low	**>4-5**	Very high	-	**Very low**	**1**

The resulting values for the I-map applying the VURAAS method are between 0,5 and 2,5 points. All five classes of vulnerability are presented within the test site (see Tab. 37 and Fig. 83). The scores of the PI method are ranging from 1,6 to 4, thus the test area can only be divided in three classes of vulnerability applying the PI-method.

The classes of Vulnerability are much wider (from very low to extreme vulnerable) applying the VURAAS method. The geological situation within the high alpine test site requires some modifications of the assessment scheme of the factor P of the PI-method.

The factor E of the VURAAS method is used for validation of the I-Map, which is described below.

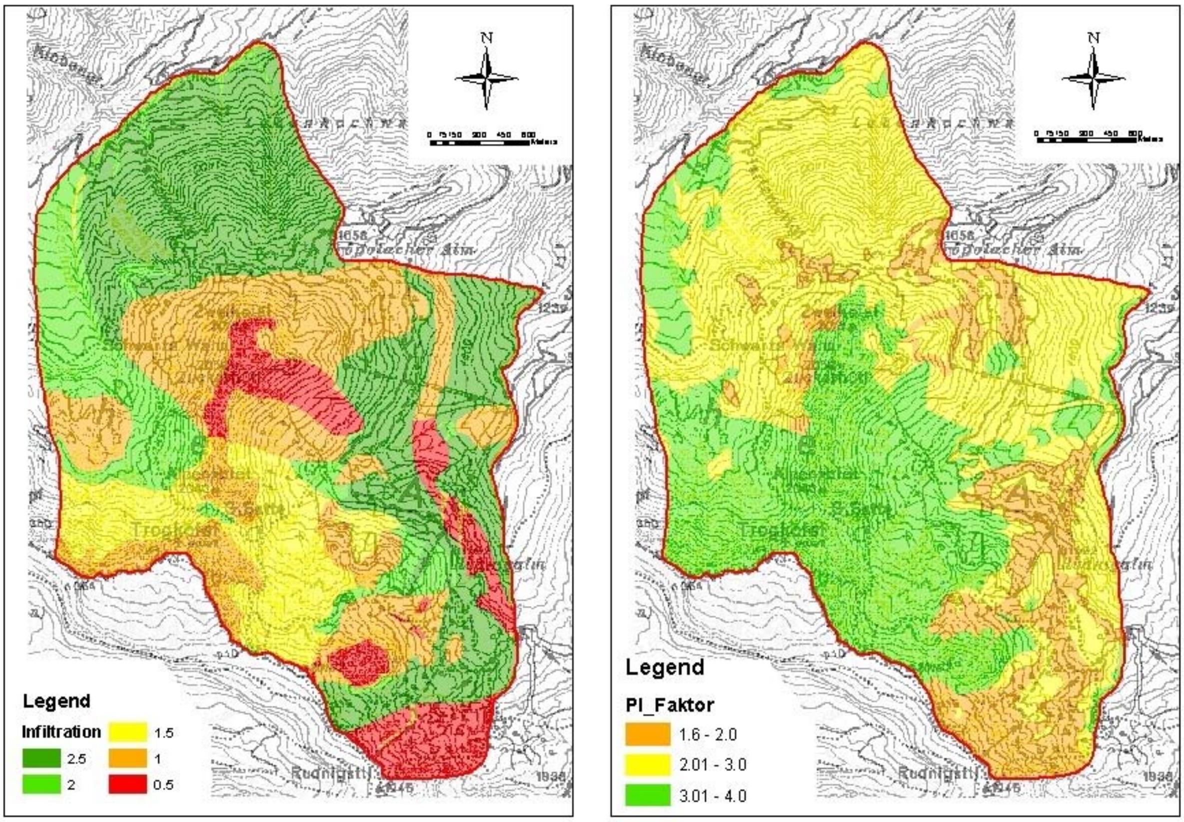

Fig. 83: Resulting I-map (VURAAS method) and PI-Map (PI method)

2.5.2.6 Karstification – Factor K (factor E - exfiltration in VURAAS method)

In the VURAAS method, information from the whole catchment area was used to analyse the exfiltration. The total points for the quality assurance depend on the availability of data, which characterise the water storage in the whole hydrogeological system. If data concerning the output of water are not available, the K-factor will be neglected for the total calculation.

The factor K is expressed by three parameters, which describe the system dynamics: (a) the fluctuation of the hydrograph, (b) the coefficient after MAILLET (1905) by recession curve analyses and (c) the mean residence time of water.

A long term observation of springs with measurements of discharge, hydro-geochemistry, electric conductivity, water temperature and measurements of environmental isotopes (^{2}H, ^{18}O) are carried out and interpreted to assess the dynamics and storage capacity of the whole aquifer. Thus, the fluctuation of the hydrograph, the water temperature and conductivity related to the discharge gives information about the storage of water in the underground water body.

Recession curves are evaluated with respect to the storage capacity, which is expressed by Maillet's coefficient (α-value). The mean residence time of water in the underground is usually determined by tritium (^{3}H) analysis.

2.5.2.7 Discussion

- The results of the methods used in the test site should be checked by a validation method (e.g. with VULK or factor E of the VURAAS method).
- In applying the PI method in high Alpine areas, the scores for the unsaturated bedrock (both karst and non-karst) are leading to an overestimation of the effectiveness of the protective cover in some cases (see Fig. 83), because of the thickness of layers of some hundred meters. On the other hand the soil seems to be less important in the calculation of the PI method.
- Uncertainties in the determination of the thickness of each layer are a problem in this strongly heterogeneous area because of a lack of data for their determination. We expect an increasing thickness of the subsoil with decreasing altitude of the slopes. Furthermore, we assume a mean thickness for homogeneous areas.
- How can the tectonic structure be handled, e.g. the dipping of layers? Complex structural geological conditions are difficult to describe in a simple model.
- The location of the groundwater table/surface is not easy to determine in Alpine karst systems.
- The individual methods should be calibrated with respect to comparable classes.

2.5.3 Hazard Mapping

2.5.3.1 Introduction

The Nassfeld pilot area for hazard mapping represents the same area as for mapping intrinsic vulnerability. The aerial extent is 8,3 km². Around 46% is covered by forest, 25% is low vegetation ground, 13% is alpine pasture, 6% represent rock outcrops, 4% ski courses, with the remainder representing gravel, pasture and low vegetation.

The eastern part of the test site is characterized by winter tourism infrastructure: several ski runways exist on the Trogkofel and Zweikofel mountains and further runways are recently planned and built. Two Alpine farmhouses are used in winter as ski huts.

Several karst springs are situated at an altitude of around 1600 m; some of them are locally used by alpine farmhouses, some supply a tourist centre in the vicinity (Sonnleitn village) or are used for the public water supply down in the Gail river valley. Vulnerability and hazard mapping in this area clarifies whether there exists a threat of groundwater contamination posed by human activities within the catchment area.

2.5.3.2 Description of hazards

The types of hazards within the Nassfeld test site were identified and evaluated according to the inventory of hazards set up by WG3. The mapping scale is 1:10.000. Tab. 38 includes all the hazards that were identified and mapped during a survey conducted in summer 2002.

Tab. 38: List of hazards mapped in the test site Nassfeld

	HAZARDS	Qn Ranking	Rf	Hazard Index HI	Hazard Index Class
1	**Infrastructure development**				
1.1	*Waste Water*				
	septic tank, cesspool, latrine	0.9	0.5	16	1
	treatment plant without biological stage	0.8	0.5	8	1
	runoff from paved surfaces	1.2	0.9	27	2
	waste water discharge into surface water cources	0.8	0.6	19	1
1.3	*Fuels*				
	storage tank, above ground	0.8	0.5	16	1
1.4	*Transport and traffic*				
	road unsecured	0.85	0.8	27	2
	car parking area, unsecured	0.8	0.8	19	1
1.5	*Recreational installation*				
	skiing course	1.2	0.9	22	1
1.6	*Diverse hazards*				
	transformer station	0.8	0.5	14	1
2	**Industrial activities**				
2.2	*Excavation sites*				
	gravel and sand pit	0.8	0.5	12	1
2.7	*Diverting and treatment of waste water*				
	waste water pipelines	1	0.6	39	2
3	**Livestock and Agriculture**				
3.1	*Livestock*				
	animal barn (shed, cote, sty)	1.1	1	28	2
	manure heap	0.9	1	36	2
	slurry storage tank or pool	0.8	1	36	2
3.2	*Agriculture*				
	stockpiles of fertilisers and pesticides	0.8	0.6	19	1

In this time 15 different kinds of point, line and polygon hazards were distinguished. Nearly all point hazards are around the alpine farmhouses or beside the ski lifts buildings. Initial information about these hazards was obtained by asking the mountain farmers and the operators of the lift systems, respectively. The next step was to map all kinds of point hazards in the field, for example, car parking areas, septic tanks, treatment plants, storage fuel tanks, cesspools, animal barns, manure heaps, among others (see Tab. 38). Information on the livestock and agriculture, such as Livestock Units (LU) or size of the animal barn, are summarized in the land-register "Almkataster Nr.1053 and 1050" and were used in the hazard assessment.

Linear hazards like forest tracks were delineated using the topographic map. During the winter skiing time (from November until May) the forest tracks are covered by natural and mechanical snow and have the function of a skiing course. The existing and planned waste chanical snow and have the function of a skiing course. The existing and planned waste water pipelines were printed in a map undertaken for an "Environmental Protection Assessment", executed by law for several Nassfeld projects.

Polygon hazards like skiing courses are also shown on this map mentioned above, which was the basis for delineating them in our hazard map. Fertilisers, which mainly consist of organic substances, are spread on the ski runways during the summer months. The recommended amount of fertilizer for skiing tracks is 1200 to 1700 kg/ha.

2.5.3.3 Determination of Hazard Index (HI)

The *weighting factor* (H) for the 15 different hazards was obtained using the values proposed by WG3. The latter represent the degree of harmfulness of different hazards to the groundwater.

The *ranking factors* (Q_n), which can modify the H values by a maximum of 20%, were determined by evaluating the relative size of the hazard in comparison to the "average occurrence". Therefore any information about the quantity of substances relevant to the respective hazard types was collected. As mentioned above, most hazards are near the two alpine farmhouses. The ranking factor of 1,2 for the runoff from paved surfaces is due to visible manure of the stable spread on the paved surface. The size of animal barns and the structural state of the stable building is reflected in a Qn-value of 1,1. The number of live stock units allows a lower value of Qn; also for the cesspools and the manure heap, a Qn value of 0,9 was given. The low estimated frequency of cars on the forest tracks and the number of tourists who visit the alpine farmhouses justify the low ranking factor of 0,8 for the unsecured roads and the car parking areas. The great expansion of skiing courses is responsible for a higher Qn value of 1,2. On the other hand the stockpiles of fertilisers and pesticides at the margin of the skiing runways are estimated to be less hazardous. Therefore a Qn of 0,8 was given.

The *reduction factor* (R_f), which describes the probability of contamination, was empirically estimated. Theoretically the reduction factor may range from 1 to 0. The estimated values for the test site Nassfeld are shown in Tab. 38.

The Hazard Index ranges from 8 to 39. It has to be emphasised that hazards within the test site are not continuously present during a year. For example, fertilizers are only spread in June. Livestock is mostly present only during this same period.

2.5.4 Graphical Interpretation

The graphical interpretation allows the point, linear and polygon hazards to be distinguished separately. In a GIS-program standardised Marker Sets were used which are similar to the ones proposed by WG3. The signature and symbols used in Fig. 84 are presented in the legend below.

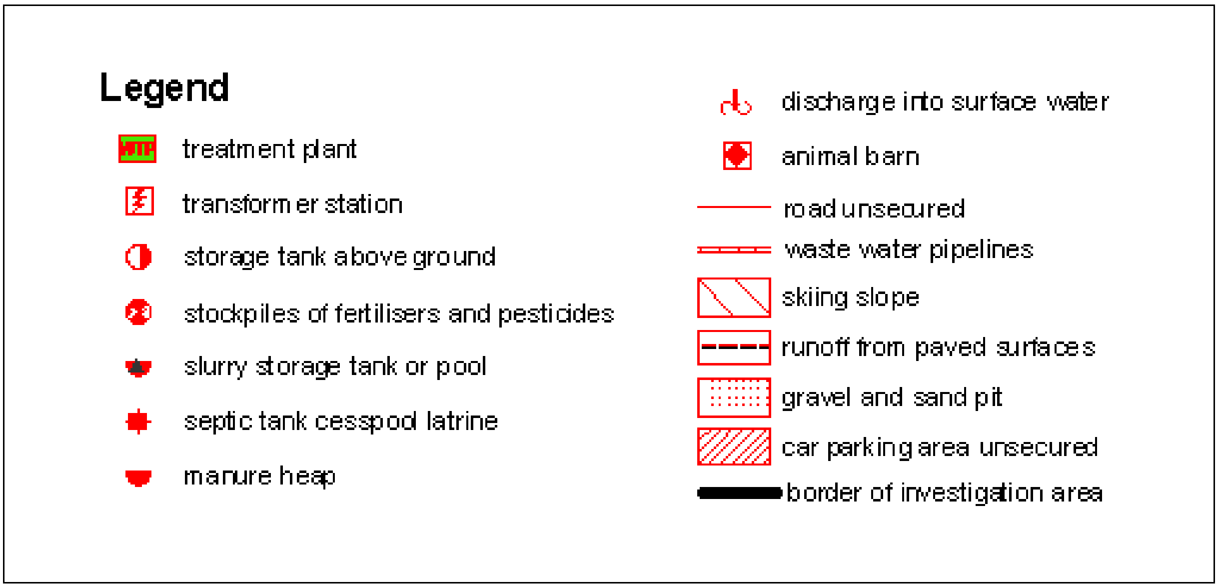

Fig. 84: Legend of the Marker Sets used in the test site Nassfeld

The unclassified map on the topographic base shows all the hazards in the test site in red, independent of the Hazard Index (Fig. 85). In the classified hazard map (Fig. 86), the hazards are plotted in different colours according to the Hazard Index Class. Most of the area is shown in blue, representing no or very low hazard level, while the green symbols are indicating a low hazard level.

2.5.5 Usefulness of the hazard map for the test site

The hazard map was mapped in a scale of 1:10.000 and printed at a less detailed scale (1:20.000). The possibility for a graphical interpretation of line and polygon hazards is quite sufficient but for decision making in the area around the alpine farmhouses, a more detailed scale should be preferred for a better localisation of the potential sources of contamination.

The type of hazards and their corresponding degree of harmfulness may partly change in the future because of infrastructure development that is being planned at the moment. Pipes taking the wastewater elsewhere for treatment will enable such treatment within the test site to cease. On the other hand, new ski runways, further ski transport systems (ski lifts and cable cars) and cesspools are planned which will increase the number of hazards.

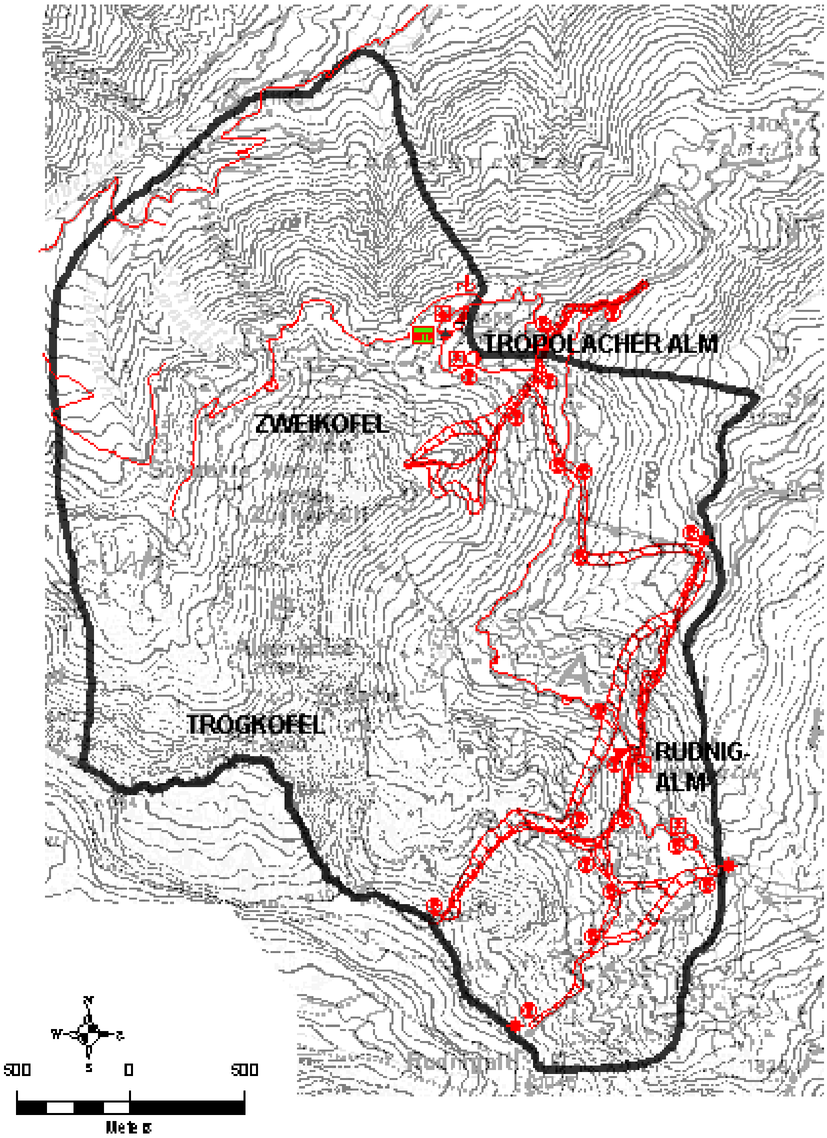

Fig. 85: Unclassified hazard map of the test site Nassfeld (all hazards symbols in red)

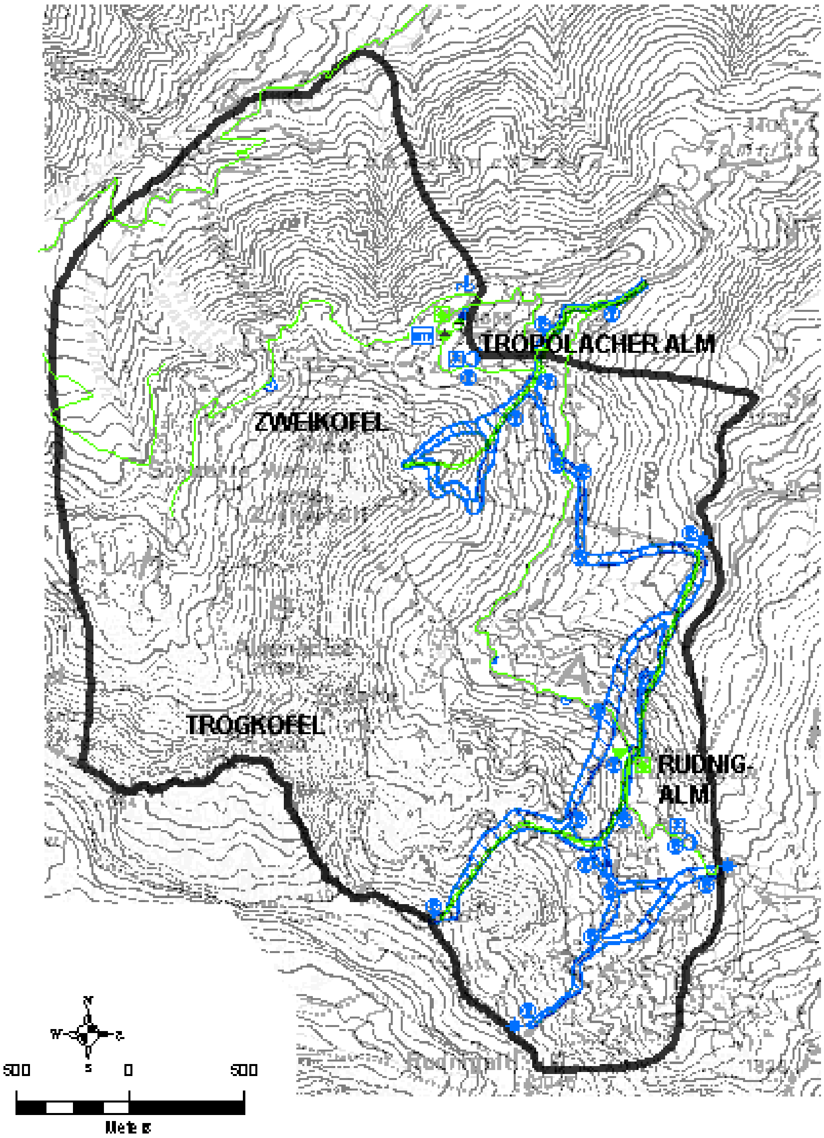

Fig. 86: Classified hazard map of the test site Nassfeld (colours according to the Hazard Index Classes)

2.5.6 References

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2.6 Zöbelboden, Northern Calcareous Alps, Austria

– Intrinsic vulnerability mapping using the Time-Input method, and hazard assessment

2.6.1 General

The Time-Input Method (Kralik & Keimel 2003) and the hazard assessment as proposed by COST Action 620 were applied in a well-studied forested dolomite karst area in the front range of the Austrian Northern Calcareous Alps (Reichraminger Hintergebirge) 50 km South of Linz. The total area of 5 km^2 was split into a fine grid of 20x20m cells. The altitude of the steep mountain ridges ranges between 500-950 m. The monitoring sites are divided in plateau and slope areas. The natural mixed mountain forest (beech, fir) covers 85% of the area. The rest are bush and grassland.

Annual rainfall ranges from 1500 to 1800 mm and depends strongly on local relief (slope, orientation). Monthly precipitation ranges from 100 mm (October) to 230 mm (July). The coldest monthly temperature (900m) is -0.9°C (January), the highest is 15.5°C (August). There are 188-198 days with temperatures higher than 5°C. At altitudes of 900 m snowfalls occur between November and May with an average duration of snow cover of about 4 months, although this is very variable (Mirtl 1996).

Human impacts are timber logging, hunting, mountain biking and mountaineering.

2.6.2 Geology and soil (sediment) characteristics of the investigated area

Tectonically the Northern Calcareous Alps of Austria form part of the east alpine orogeny, with clearly north-facing imbricate and folded structures, originating from Cretaceous and Tertiary orogenies. The project area belongs to the Reichraming nappe and is part of the northvergent Kreuzeck anticline.

The main type of rock is Norian (Triassic) dolomite (Hauptdolomit) with a thickness up to 500 metres. In some small areas the dolomite is overlain by limestone (Plattenkalk) and Upper Jurassic – Lower Cretaceous marls, limestone and radiolarites (see Fig. 87). The dolomite is hydrogeologically a fractured aquifer with limited karstification along bedding planes and fault zones indicated by initial doline structures on the plateau of the Zöbelboden.

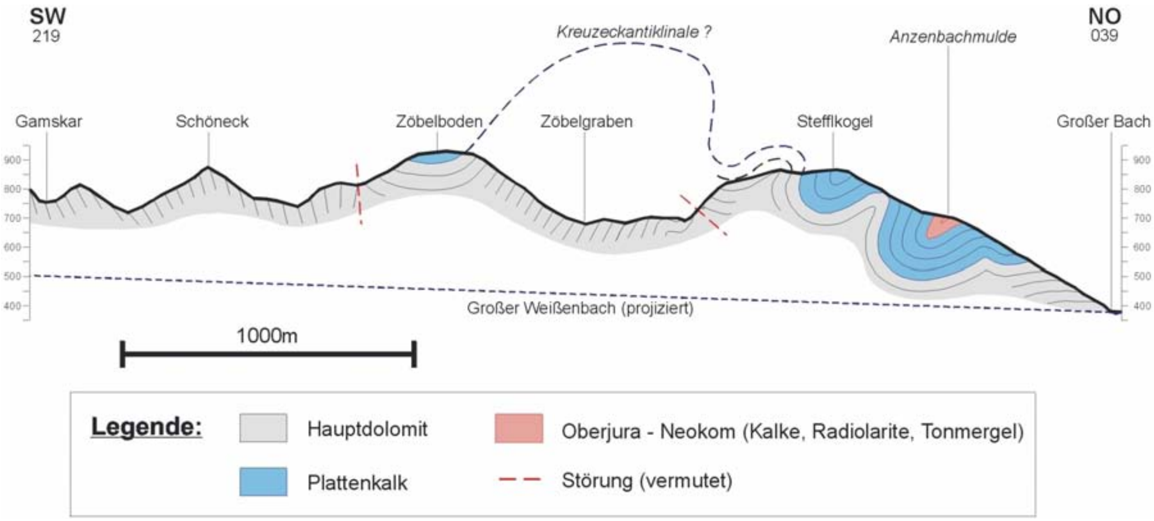

Fig. 87: Cross-section of the Zöbelboden dolomite massif

The occurrence and distribution of soil types mostly depend on the local relief (inclination). According to FAO-nomenclature the plateau contains mainly medium thick (0.3-0.6m) Cambisol (relictic brown loams most likely formed from weathered dolomite (the whole region had been part of the periglacial zone) and the partly steep (30-45°) slopes show mainly thin (0.05-0.30m) Rendzic Leptosol (colluvially influenced Rendzinas).

Below the soils coarse-grained dolomite scree (talus) covers the plateau of the carbonate massif and lower parts of the slopes of the Triassic carbonate rocks. These up to several meters thick layers are caused by earlier fluvio-glacial activities or recent ongoing erosion. They form the "aquifer" for a rapid interflow discharge predominantly during rainstorms and long lasting rain events.

2.6.3 Groundwater vulnerability assessment

2.6.3.1 Acquisition of assessment data

The official Austrian geographical map 1:25.000 was enlarged to a scale of 1:5.000 and the investigation area of 5 km^2 provided with a grid of 20 x 20 m. The geological 1:50.000 map of the Austrian Geological Survey and a detailed 1:10.000 (Leithner 1997, Keimel, 1999) hydrogeological map of this area were used as geological background information. These also provided the basis to estimate the thickness of the layers and to delineate areas with the dip of bedding planes towards and away from the groundwater. Usually, soil information was obtained by assigning typical morphologies such as, hilltops, plateaux, depressions, trenches, steep and gentle slopes and soil assemblages. This information was obtained from aerial photographs. The mean evapotranspiration decreases from 35% (forest) and 23% (scrub and grassland) to 7% (bare rock). Therefore, vegetation can be simplified into three classes (Katzensteiner 1999).

In addition to the aforementioned interpretation, six days fieldwork were undertaken in order to obtain the necessary additional data and to verify results.

2.6.3.2 Evaluation of main factors (residence) Time and Input (groundwater recharge)

Discharge, temperature, electrical conductivity, pH and major ions were measured periodically at twenty springs and surface waters (small sub-catchments). These data were combined with measurements of a main on-line station with a weekly sampling for chemistry. This allows the identification of sub-catchments with excess or a deficit of the nominal discharge. Likewise, those sub-catchments may be identified with highly variable water composition and rapid travel-times of at least part of the water input.

The significant lower surface runoff of the southern sub-catchments reflects the importance of the higher evapotranspiration due to greater solar radiation input. Excess discharge from the southeastern and eastern springs and surface runoff from their sub-catchments indicate rapid groundwater transport from the plateau area and the north-facing catchment areas along tectonic fault zones.

Springs at higher altitudes (700-800 m) are very dynamic (high relative standard deviations) in water temperature and conductivity. These southeastern and eastern springs show a medium response after storms, whereas the northern springs close to the receiving stream are very constant.

Oxygen-18, Deuterium and Tritium model calculations indicate mean residence times of some weeks in agreement with the vulnerability assessment. Only the northern springs have ages of several months.

Four tracer experiments (Haseke 2000) on top of the plateau close to the fault zones and karstification structures (removed soil covers) indicate a short residence time of 1-2 days during and after heavy rainfall as previously determined by the TIME-INPUT method.

2.6.3.3 Discussion of Time and Input assessment

The results of this study obtained by the use of the Time-Input assessment scheme highlight the vulnerability of groundwater in particular above faults and along the lowest parts of the slopes closest to the groundwater. Most springs emerge in this area In this strongly tectonised bedded dolomite formation some fault zones seem to be responsible for rapid travel-time to groundwater as demonstrated by tracer tests.

Tab. 39: Attribution of total bulk infiltration (travel-times) to time classes

Time-Classes	Time-Intervals	Bulk infiltration times in seconds	Vulnerability classes
1	<12 hours	<43200	**extreme**
2	12-24 hours	43200 - 86400	
3	1-2 days	86400 - 172800	**high**
4	2-4 days	172800 - 345600	
5	4-7 days	345600 - 604800	
6	1-2 weeks	604800 - 1209600	**medium**
7	2 weeks -1 month	1209600 - 2592000	
8	1-3 month	2592000 - 7776000	**low**
9	3-6 month	7776000 - 15552000	
10	>6 month	>15552000	

The classification of the travel-time of infiltration from the land surface to the groundwater surface into ten classes certainly indicates tendencies rather than accurate estimates. It could also be grouped into three vulnerability classes: High (travel-times 1-4 days), medium (1-4 weeks) and low vulnerability (> months) during or after a series of major rainfall events (Fig. 88 and Tab. 39).

The input (groundwater recharge) classes (Tab. 40) have to be adapted to the climatic conditions to obtain the modified time classes expressing the degree of vulnerability (Tab. 39).

Tab. 40: Correction factors for the Input (groundwater recharge by the amount of infiltrating water).

GW-recharge by infiltrating waters	Correction Factor Q
0-200 mm	1.50
200-400 mm	1.25
400-600 mm	1.00
600-800 mm	0.75
800-1000 mm	0.50
>1000 mm	0.25

However, the classification of medium vulnerability of the most parts of the investigation area with some minor parts of extreme and high vulnerability, is reasonably confirmed by the evaluation steps.

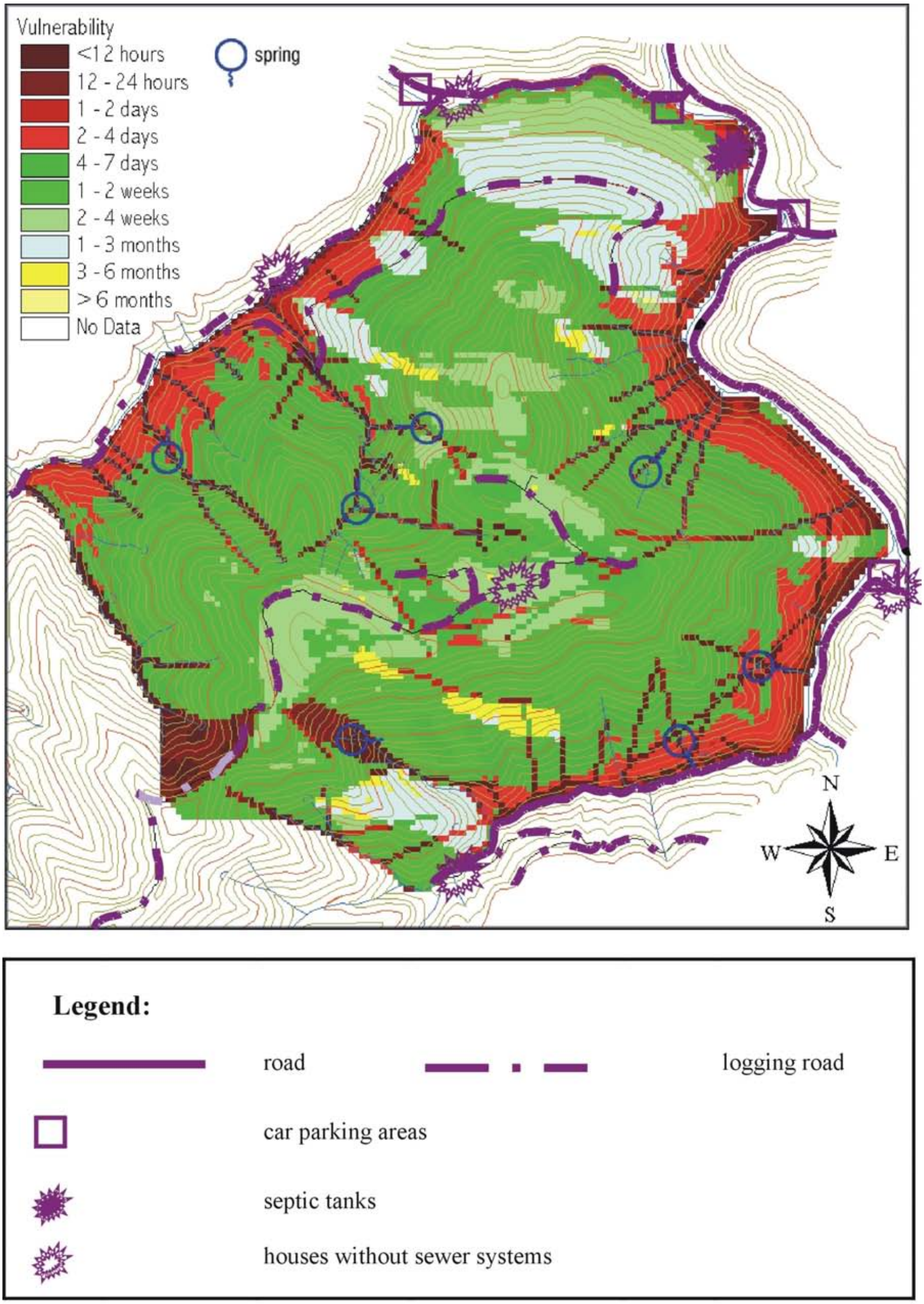

Fig. 88: Combined groundwater vulnerability (Time-Input method) and hazard map of the test site.

2.6.4 Hazard assessment

2.6.4.1 Introduction

Even or because the investigation area is situated in a national park, a hazard assessment method as proposed by COST 620 was applied in order to evaluate the risk of groundwater contamination posed by human activities. The detailed hazard classification and assessment schemes are given in chapter 5 on hazard mapping in part A of this report.

2.6.4.2 Description of hazards

Two different kinds of hazards could be identified in the Zöbelboden test site: linear and point hazards.

Linear hazards are shown in Fig. 88 as unsecured roads surrounding the test area and two logging roads (in the North and in the centre of the Zöbelboden). They represent a potential source of contamination by transport and traffic (hazard type no. 1.4). Trucks transporting timber frequently use the unsecured roads. In summer respectively weekends a high number of tourists are using the roads visiting the national park “Calcareous Alps”.

The point hazards are concentrated around houses or parking lots. The houses in the NE of the test area are mainly used for recreation and vacation, the others are mainly used by forest workers and hunters. The first mentioned houses comply with 1.1.4 (septic tanks) described in the list of hazards. In comparison the houses used by forest workers and hunters, which have no sewer systems belong to 1.1.3. That means that wastewater represents the most probable possibility of contamination in this case. Although there are no differences according to the weighting value H, distinctions can be made by the reduction factor R_f (Tab. 41).

The parking lots indicate another possibility of contamination by transport and traffic (1.4). Here leaky fuel tanks or tripping oil can pose a threat to the ground water.

2.6.4.3 Determination of Hazard Index (HI)

The weighting value **H** was taken from the list of hazards of and gives a factor running from 0 (not harmful) to 100 (extremely harmful) showing the harmfulness of a hazard to the groundwater.

The ranking factor $\mathbf{Q_n}$ between 0.8 and 1.2 can the weighting value up to ± 20 % by multiplying it to H. It shows the quantity of toxic substances in comparison to the average.

The reduction factor $\mathbf{R_f}$ from 1 to 0 is an empirical number. If the factor is 0 it means that there is no possibility to contaminate the groundwater, whereas 1 would mean that there is no information whether the groundwater can be contaminated or not.

Multiplying these three values results in the Hazard Index (HI), which describes the degree of harmfulness of each hazard. All hazards mapped in the test area have relative low (> 24 – 48 points) or even very low hazard levels (< 24 points) (Tab. 41). As a result the hazards in the test area can be divided into the two hazard index classes 1 and 2 (very low and low) as proposed by COST action 620 (see chapter 5, part A).

Tab. 41: List of hazards mapped in the Zöbelboden test site

HAZARDS	Weighting Value H	Ranking factor Q_n	R_f	Hazard Index HI
Road, unsecured	40	0.85	0.8	27
Logging road	40	0.8	0.8	26
car parking area	30	0.85	0.8	20
septic tank	45	1	1.0	48
houses without sewer systems	45	1	0.8	36

The signature and symbols for the different type of hazards are shown in the groundwater vulnerability and hazard map (Fig. 88).

2.6.4.4 Discussion

Hazard and Risk assessment: Due to the introduction of hazard information in the groundwater vulnerability map (Fig. 88) sites of increased risk of groundwater contamination around houses, roads and parking areas can be shown. The houses and roads around the Zöbel massif are located above highly vulnerable areas without protective cover and quick transfer times to the groundwater. However, due to relatively small hazards, high dilution (1500-1800 mm precipitation) and small distances to the creeks the risk of intensive and long lasting groundwater contamination is extremely low. A slightly higher risk of groundwater contamination exists along the logging road crossing the broad and intensive tectonised fault zone in the Southwest of the investigation area.

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2.7 Veldensteiner Mulde, Franconian Alb, Germany

– Application of the PI method of intrinsic vulnerability mapping using a GIS-supported multiple regression approach to assessing overlying layer thickness –

2.7.1 Geographical and Geological Overview

The Veldensteiner Mulde is a tectonic depression in the Northern Franconian Alb, S Germany (Fig. 55). The subsurface catchment of the wells and springs covers an area of about 250 km^2, out of which a test area of 63 km^2 was mapped in this study (Fig. 89). Within the test area, the relief drops gently from 600 m above sea level at the dolomite knolls in the west down to 400 m at the bottom of the Pegnitz valley in the east. Two thirds of the test area is forested, the rest is mainly used for agriculture. The motor highway A9 crosses the area from S to N. The mean annual temperature ranges from 7 to 8 °C. The mean height of annual precipitation is between 800 and 900 mm.

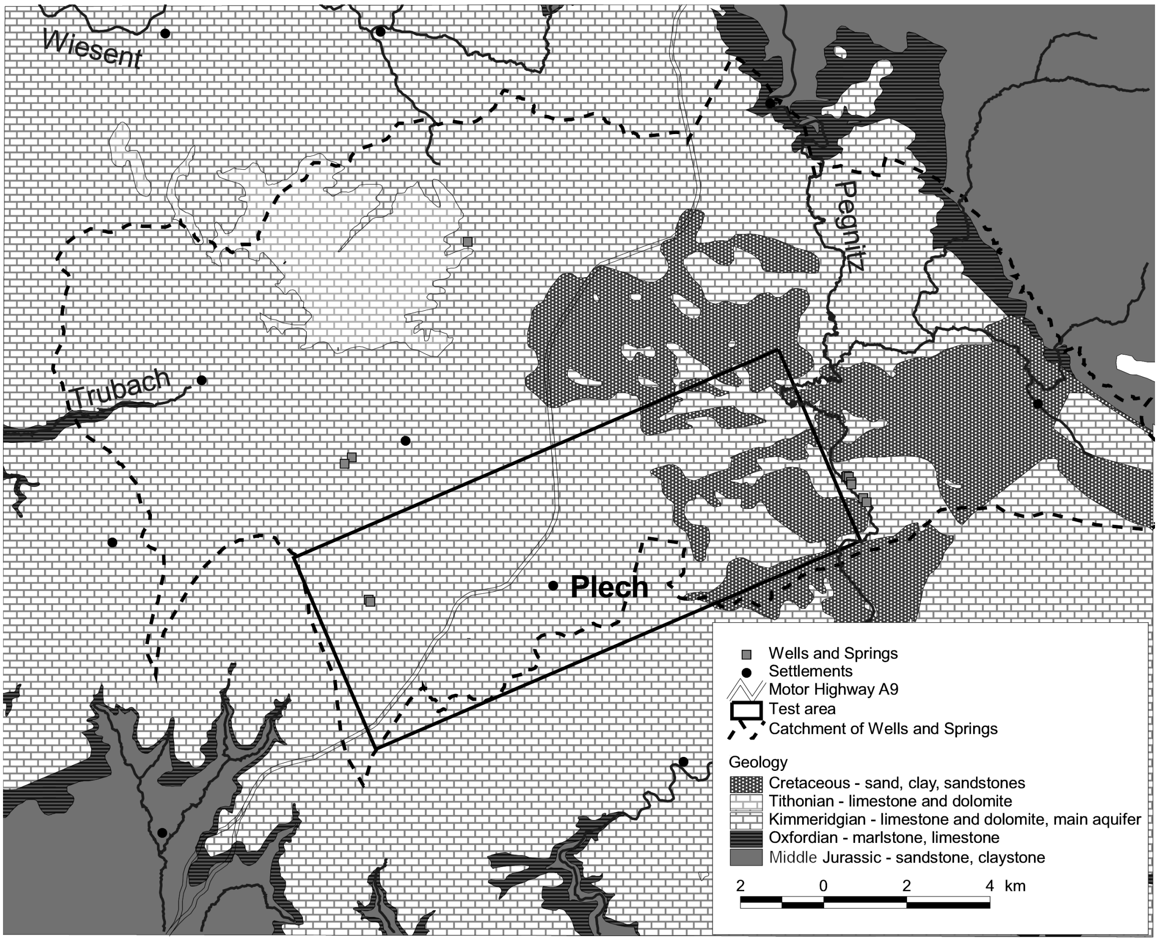

Fig. 89: Generalised geographic and geologic overview of the Veldensteiner Mulde with an approximate outline of the catchment of wells and springs and outlines of the water protection areas.

Among the karstified German highlands, the Northern Franconian Alb is exceptional for its geologic and paleo-geographic evolution. Carbonate formations of about 250 m in thickness were sedimented the Upper Jurassic (Oxfordian-Tithonian), both as bedded and massive facies. They converted diagenetically to dolomite for a large part. After the Jurassic sea had retreated, an intense karstification phase set in under tropical conditions in the Lower Cretaceous period. This resulted in a structured relief with cone and tower karst features, subsurface channels, dolines, and poljes (Fig. 90). The karst relief was subsequently covered with sandy to clayey sediments in the Upper Cretaceous period.

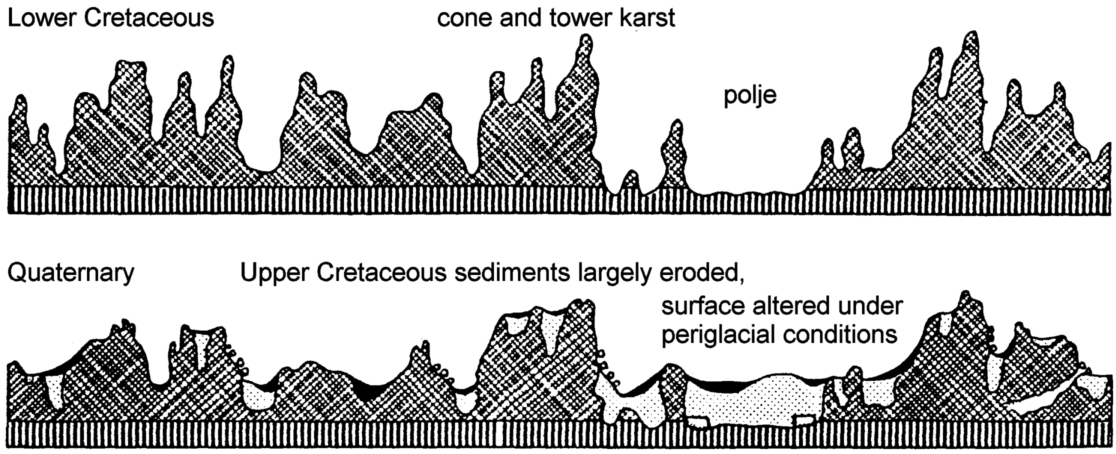

Fig. 90: Schematic profile for the Northern Franconian Alb in the Lower Cretaceous and in the Quaternary period. In the Upper Cretaceous the cone and tower karst relief had been covered completely with sandy to clayey sediments. Modified after PFEFFER (1986:81).

In the Tertiary, erosion of the Cretaceous material partially re-exposed the Jurassic bedrock, into which the Pegnitz cut its valley throughout the Pleistocene. However, Cretaceous sediments of considerable thickness can still be found in tectonic depressions and in paleokarst hollows and basins. They are mainly unconsolidated except for the Auerbacher Kellersandstein, a sandstone formation that is locally present in remnants. During the Ice Ages, Loess deposition and solifluction further altered the relief (Fig. 90), providing a complexly amalgamated soil parent material (Tillmann & Treibs 1967, Pfeffer 1986).

The current landforms can thus be interpreted as having experienced two karstification phases, the first of which accounts for a fossil Cretaceous relief. In the second phase, which is still continuing, new dolines and ponors have formed, partly reactivating old subsurface channel systems. Some local surface watercourses sink into those ponors. Being most active during snowmelt, they generally dry out in summer. In the test area, the Pegnitz as an allogenous stream is the only permanent surface water (Spöcker 1952).

2.7.2 Intrinsic Vulnerability Mapping (PI Method)

The PI method (Goldscheider et al. 2000) was applied in the Veldensteiner Mulde in order to assess and map the karst groundwater vulnerability. The Veldensteiner Mulde holds a regionally important groundwater resource that has been used for drinking-water supply since the beginning of the 20th century.

The project was carried out at the University of Karlsruhe (Department of Applied Geology, supervised by Prof. H. Hötzl, Prof. D. Burger, and Dr. N. Goldscheider) in cooperation with GeoTeam Ltd., Bayreuth. Maps and data were provided by GeoTeam and by the Bavarian department of geology. The Bavarian department of land survey provided a digital elevation model (DEM). A comprehensive report can be found in Schmidt (2001).

For the vulnerability map in Veldensteiner Mulde, the PI method was applied at a scale of 1:25.000 as outlined in Goldscheider et al. (2000), except for some minor modifications (Schmidt 2001): For the unsaturated karst rock, a factor describing the effective porosity was implemented to account for the retarding effect of rock porosity on percolating water. Porosity estimates were deduced from facies maps. Furthermore, continuous instead of discrete classification schemes were used to avoid rounding errors. According to the project objectives, the target for vulnerability mapping was always the groundwater surface in the main aquifer. Local perched aquifers above the karst aquifer were consequently not treated as groundwater bodies, which need to be protected, but as an additional protection for the underlying karst groundwater (as suggested in the German GLA method, Hölting et al. 1995).

Sensitivity analyses proved that the P factor was most sensitive to the thickness of the Cretaceous sediments, simply because of its wide range of possible values (from a few centimetres to 120 m) and the uncertainty attributed to it. Within the Veldensteiner Mulde, layer thickness data was available for more than 100 points from drillings and geoelectric measurements. However, spatial interpolation was considered an unsuitable approach due to the extreme fossil relief of the underlying karstic bedrock. Instead, a multiple regression approach was applied (Chatterjee & Price 1995), estimating the log-transformed depth to the Jurassic rock, by means of variables extracted from maps and the DEM.

In a step-by-step forward selection, the following five variables out of about 20 were found to be correlated to at the 95 % level, according to their t statistics:

1) the log-transformed distance to the nearest Jurassic rock feature as shown in the geologic map (positively correlated);
2) the slope as calculated from the DEM (negatively correlated);
3) the relative area around a point showing Jurassic rock outcrops on the geologic map within a ring of 500 and 550 m inner and outer radius, respectively (positively correlated); this variable reflects the characteristic geometry of karst depressions which formed during the Lower Cretaceous;
4) the depth to the top of the Middle Jurassic formation as calculated from the difference between DEM and bedding plane contour lines (positively correlated);
5) a point's membership in the relief class "basin" as derived from the DEM in a relief analysis (positively correlated indicator variable).

A regression equation was defined from these variables. The explained variance was R^2 = 0.585. In a 10 m raster map, the predicted thickness was used to calculate the protective function of the Cretaceous layers for each raster cell. As a byproduct, the predicted thickness was also used to reconstruct the topography of the Cretaceous landscape (Schmidt & Goldscheider 2002).

There was no soil map available for the test site. However, it was possible to estimate the effective field capacity (eFC) on the basis of data about grain size distributions in more than 160 hand auger samples (AG BODEN 1996: 297). Based on this data, regions of eFC classes as suggested for the topsoil score T (Goldscheider et al. 2000) were then outlined by combining the geologic map with DEM plan and profile curvature classes.

The thickness of the Jurassic sequences and the Upper Cretaceous sandstone series were derived from bedding plane contour line maps and literature values. A map of the karst groundwater contour lines was available but had to be updated to comply with more recent measurements. Subsequently, the total thickness of the unsaturated zone was estimated by subtracting the groundwater level from the DEM elevations.

For the I factor, soil layer permeability values were determined from the soil data mentioned above, using the German soil mapping instructions (AG BODEN 1996). Next, the corresponding dominant flow process was attributed to each of the sampling points according to the PI method. Three soil groups with different frequency distributions of dominant flow processes could be distinguished. After intersecting the soil groups with the slope and land-use classes, I' values were assigned according to the frequency of dominant flow processes within each soil group.

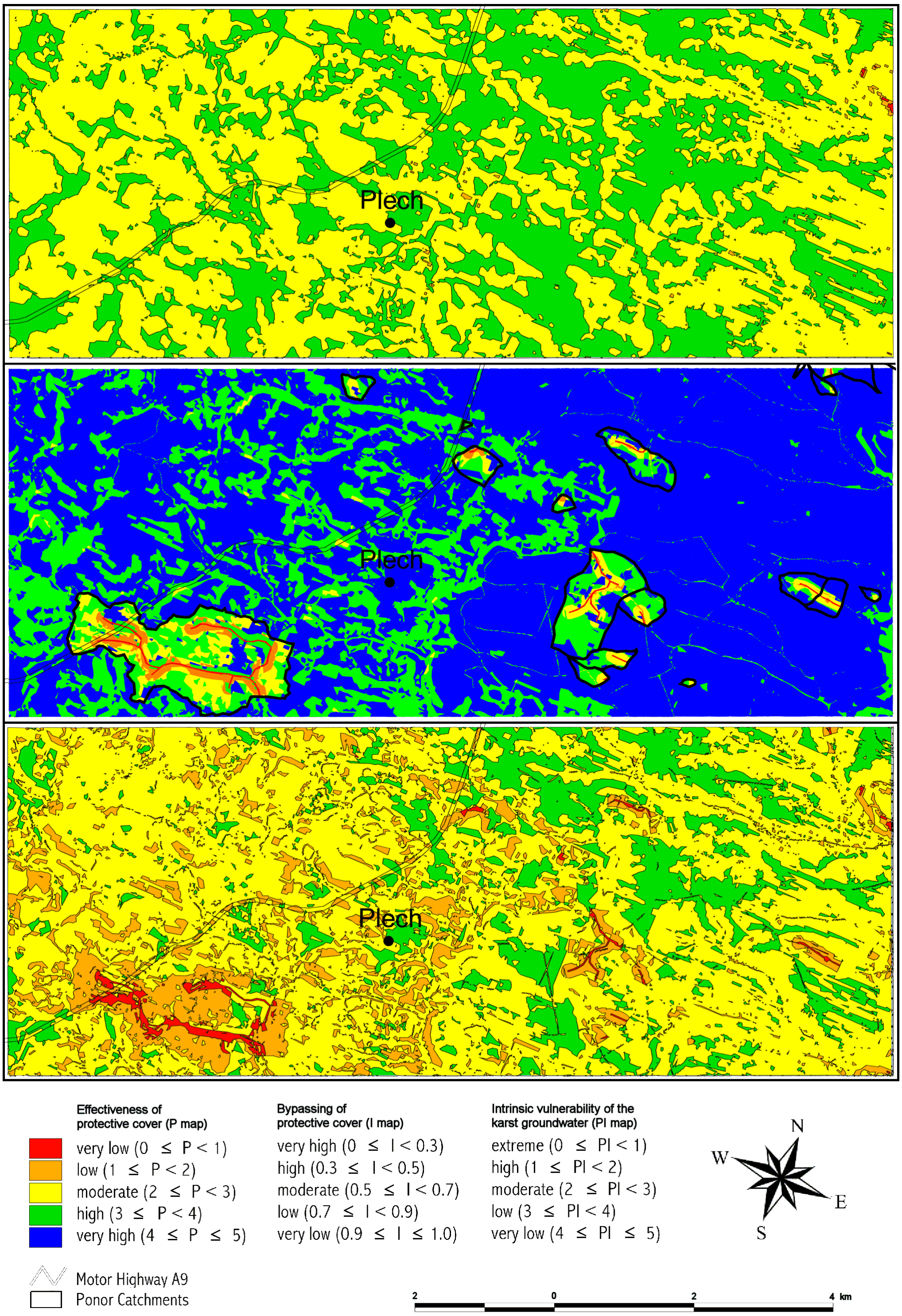

Fig. 91: P, I, and PI map for the test area in the Veldensteiner Mulde.

In the test area, 16 karst features were classified as ponors. Although the topographic maps show no surface watercourses other than the Pegnitz, dry valley troughs upstream of ponors had to be included in the buffer zones because they can hold temporary sinking streams. The I factor map was generated by intersecting the I' map with the buffer zones. The PI map was created by intersecting the P and I map and multiplying the P and I factor (Fig. 91).

2.7.3 Discussion

On the P map, two classes predominate, indicating a moderately to highly effective protective cover. The spatial pattern approximately reflects the distribution of Jurassic rock outcrops (P = 2) and of areas covered with younger sediments (P = 3). Lower P values may occur in valleys due to a shallow depth to groundwater table.

According to the I map, the degree of runoff bypassing the protective cover is low to very low for most of the test area. This is because the ponor catchments make up only a small proportion of the total area. Furthermore, no fast runoff components need to be expected on forested areas and on sandy soils from Cretaceous sediments. On the other hand, agricultural dry valleys draining to ponors may exhibit a moderate to very high degree of bypassing according to the applied method.

The PI map shows some areas of extreme groundwater vulnerability which closely match the areas of very low I factors. Especially in the western part of the test area only small patches of "low" vulnerability remain. The reasons are the predominating agricultural land-use with a higher tendency towards runoff generation, and the higher proportion of Jurassic outcrops with a lower P factor.

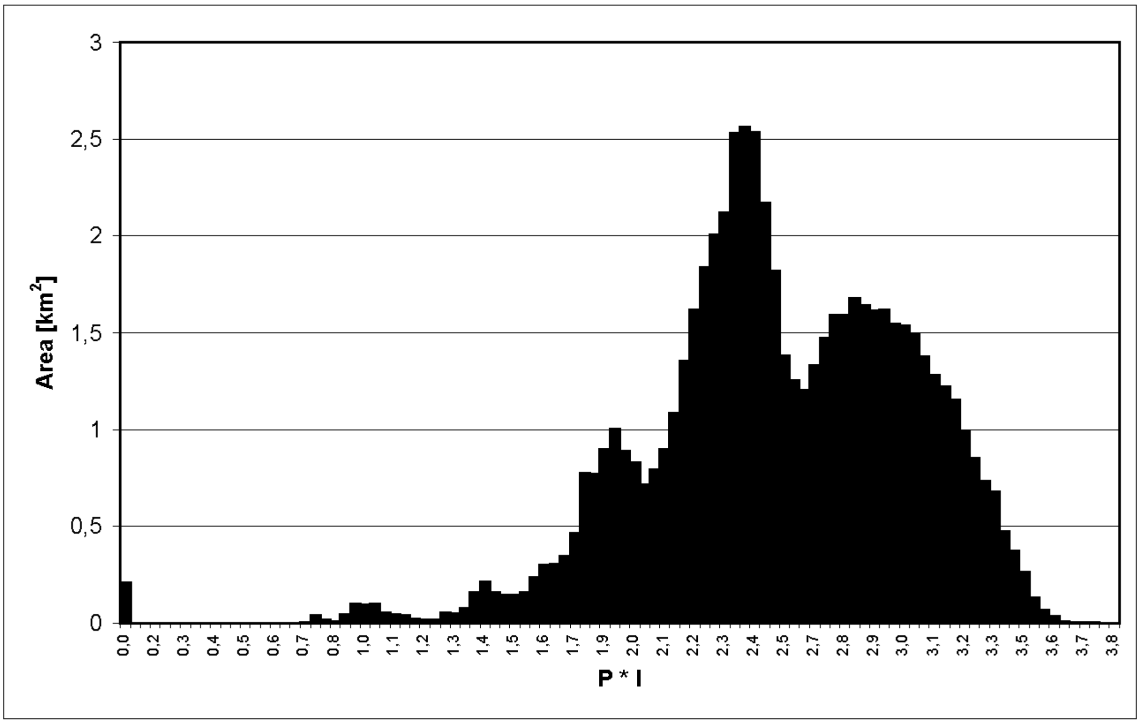

Fig. 92: Histogram for the continuous PI function, created as 190 classes of equal interval.

When comparing the P map with a vulnerability map according to the GLA method (Hölting et al. 1995), the latter suggests much higher groundwater vulnerability, because the class boundaries are different. However, any a priori classification of P, I, and PI is arbitrary. In order to better reflect possible natural breaks, a histogram-based classification may be proposed instead. Its several peaks suggest that it is composed of several overlapping vulnerability groups. This leads to the question whether it is more appropriate to adjust the class boundaries as to match the local minima of the histogram. Although such a proceeding would render the results of different studies incomparable, it should be discussed as an alternative classification approach. In the present study, the histogram suggests class boundaries at PI = 1.2, 2.0, and 2.6 (Fig. 92).

There are many uncertainties associated with the results, some of which are due to the local conditions, while others are methodical. Despite the regression approach, the protective function of the Cretaceous sediments still remains an important factor of uncertainty. Some of the dry valley troughs have sinks, which makes it difficult to delineate the ponor catchments. Since the vulnerability concept is not based on physical-mathematical process descriptions, the relative importance of infiltration and surface runoff, as expressed by the choice of an I factor value, cannot easily be justified without hydrologic modeling.

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2.8 Néblon basin, Belgium

– Intrinsic vulnerability mapping using the PI method

2.8.1 General

The Néblon basin is located in Belgium in the region of Condroz (Fig. 55). Geologically, it belongs to the part of Devonian Carboniferous pleats formations of the eastern edge of the Dinant synclinorium that crosses Belgium from West to East. This region is characterised by typical alternation of shales and sandstones anticline crests (Upper Devonien or Famennian) and calcareous syncline depressions (Lower Carboniferous or Dinatien) (Fig. 93). The geological formations are made of terrigeneous detrical facies of Famennian age, carbonated rocks of carboniferous and terrigeneous detrical sediments of Namurian age. Locally, ancient paleokarsts are filled by Tertiary sandy clay sediments. The region is also covered with loess formation.

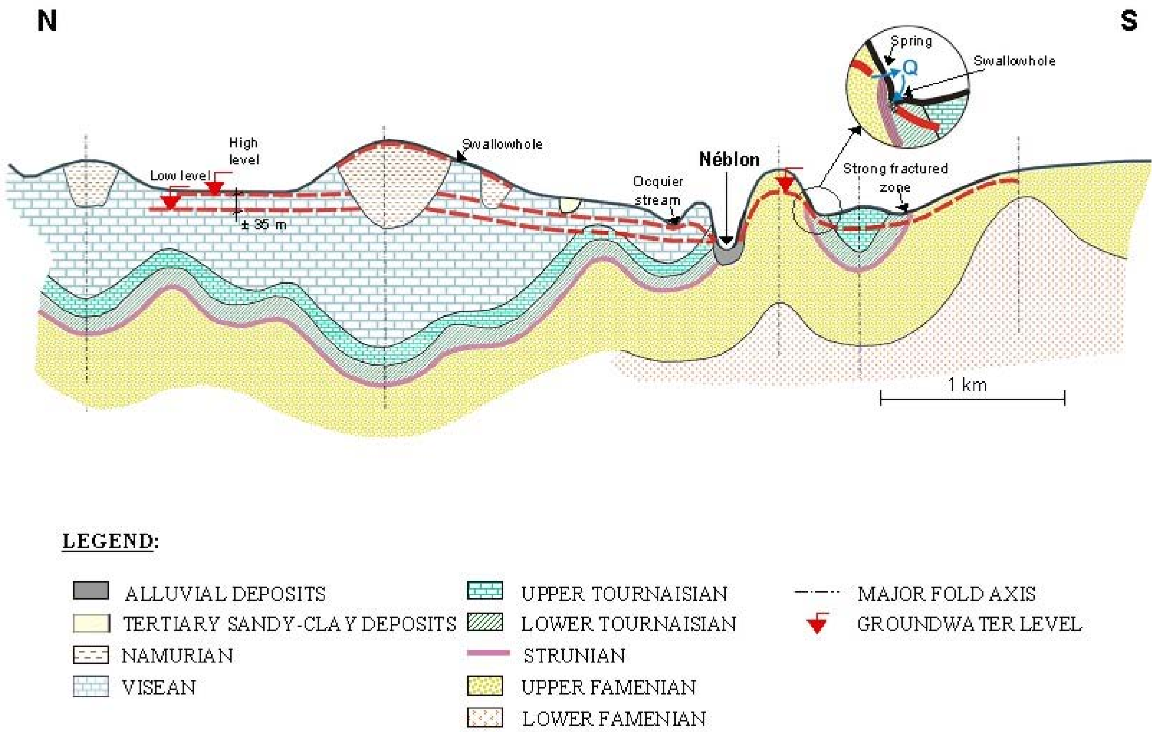

Fig. 93: Schematic N-S geological and hydrogeological cross-section in the Néblon basin (Derouane et al. 1995, Gogu 2000).

2.8.2 Geological characteristics of the investigated area

The region is intensively faulted and the main structural features can be classed in three groups:

- thrust longitudinal faults and fractures, oriented E-W, gently dipping to the S
- transverse faults , oriented NNW-SSE to N-S
- oblique faults, oriented NE-SW.

Some of the intensively faulted zones correspond to dry valleys that are supposed to facilitate groundwater drainage to the Néblon River.

In the southern part, a boundary of the basin consists in shaly formations of a Famennian anticline. The northern, eastern and western limits are located mostly in the Visean limestones. Each of these boundaries corresponds to a groundwater divide between the studied Néblon basin and two other neighbour basins. Consequently, these limits can show large spatial and time variations.

Three aquifers can be distinguished:

- the karstic aquifer in Visean and Tournaisian limestones and dolomites, exploited by the local water company CILE (Compagnie Intercommunale Liègeoise des Eaux);
- the faulted sandstone aquifer in the Upper Famennian;
- locally, the perched aquifer in the Namurian faulted sandstone.

The basin is moderately karstified. Several karstic features can be identified: a couple of dolines, dry valleys, three major swallowholes and a few resurgences (Fig. 94). A few undeveloped karstic caves are also noticed along the cliffs in the Néblon valley as well as some dolines.

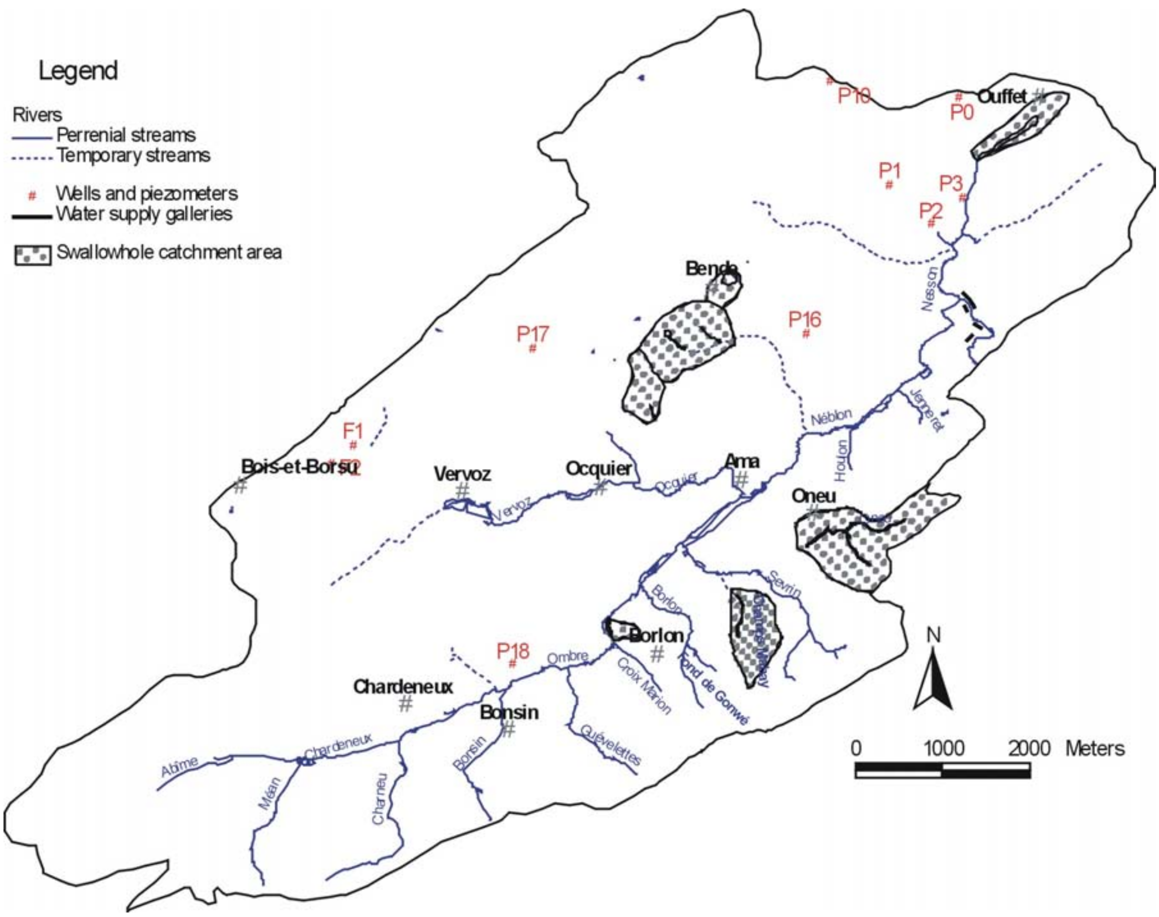

Fig. 94. Map of the basin, hydrologic network, piezometers, swallow holes and sub-basins feeding the swallow holes (Gogu 2000).

2.8.3 Vulnerability mapping using the PI method

2.8.3.1 Determination of the P parameter

Topsoil – T: The different types of soil are clayed silts, silty clays, silts, silty loams, sandy loams, silty sands, silty loamy sands, and sandy silts. In PI, the topsoil parameter must be quantified taking account the effective field capacity (mm/dm). In Germany, this parameter can be derived using the standard tables of the German Pedological Handbook (AG Bodenkunde, 1982). Then the effective field capacity is multiplied by the thickness of the soil horizon. The far more detailed Belgian soil map was used here and correspondences with the German classification had to be studied accurately. The result parameter is almost constant on the area except where the soil is significantly less thick.

Subsoil – S: The parameter is quantified taking account of the lithology of the subsoil horizon. In our case, the non-consolidated lithology is made of alluvial sediments, loess and sand-clay sediments. The thickness of these sediments is estimated on basis of precedents studies (Di Clemente and Laurent, 1986, CILE – LGIH - INIEX Report, 1986, Hallet et al, 2000).

Lithology – L: Most of the different lithologies are foreseen in the method tables. For few geological units, the indexation must be adapted. The thickness of the different units take the topographic elevation and the position of the piezometric heads into account. The thickness is reduced in the perched Namurian aquifer.

Fracturation – F: In the limestones, several zones were distinguished:

- limestones with strongly developed epikarst and showing swallowholes and dolines;
- limestones strongly fractured or strongly karstified (e.g. dry valleys);
- limestones moderately fractured or karstified

Other lithologies were supposed to have a "slightly jointed" structure

Recharge – R: The parameter was quantified on the basis of previous balance studies. Different zones can be distinguished with the recharge respectively estimated as entering into three classes: 300-400mm/y, 200-300 mm/y, 100-200 mm/y.

The P parameter is obtained combining the previous described parameters (Goldscheider et al. 2000 and chapter on the PI method in this report). The map (Fig. 95) shows that the protection is higher on the Famennian and Strunian shales and sandstones while the limestones have mostly a moderate protection.

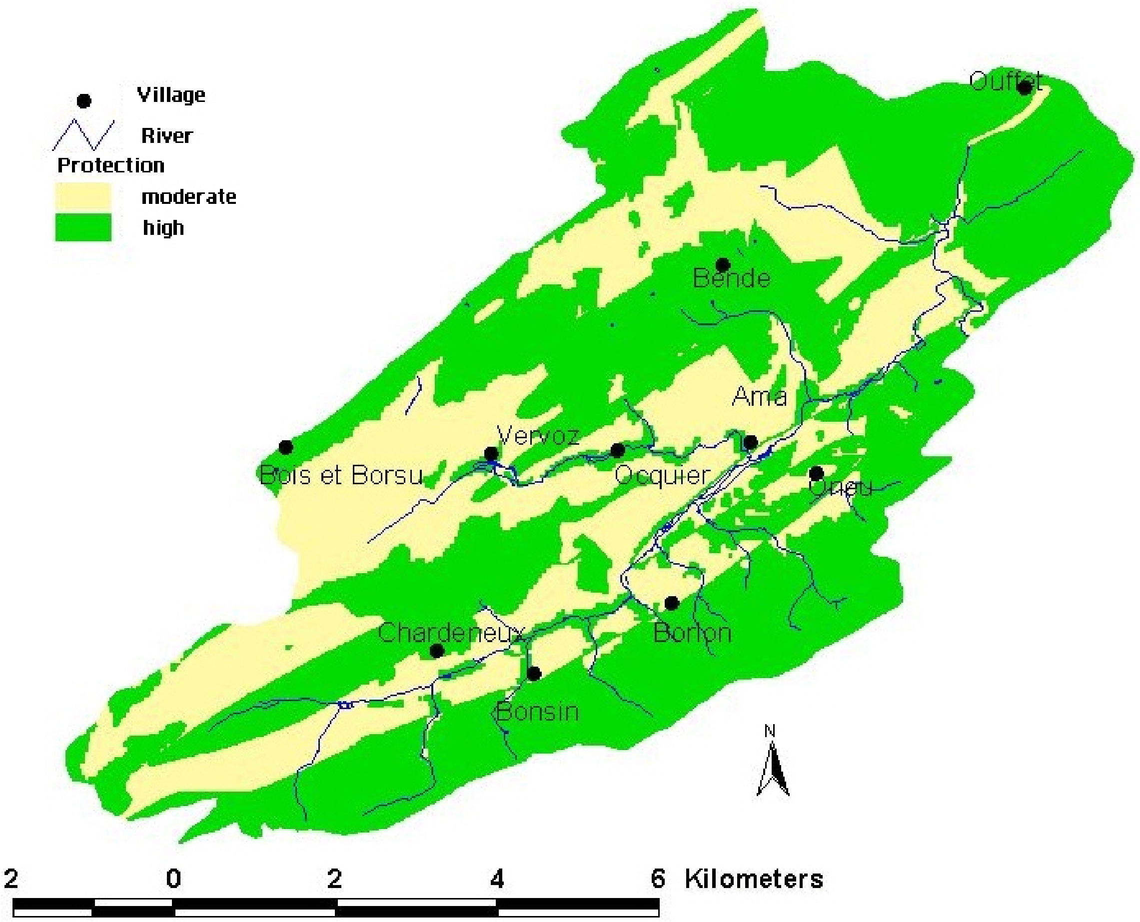

Fig. 95: Map of the parameter P of the PI method.

2.8.3.2 Determination of the I parameter

The dominant flow process is assessed on the basis of the topsoil permeability and the presence of low permeability layers. Permeability of the topsoil has been estimated on the basis of saturated hydraulic conductivity tables as proposed by Mermoud (1998).

The I' parameter is assessed by intersecting the coverages of dominant flow process, the vegetation and the slope gradient. On the basis of the landuse map, it can be noticed that ~24% of the area is covered with forest, while almost 76% represent fields,meadows and pastures. The slope gradient is calculated on the basis of a DTM with 30m pixels. The more important gradient corresponds to cliffs along the Néblon river. The lowest gradient is generally situated in the northern and eastern part of the basin.

The I map (Fig. 96) is obtained by intersecting the I' map and the surface catchment map. This one is created on the basis on a digital map containing all the swallowholes and the sinking streams. The 10 m and 100 m "buffer zones" around these features are introduced. The map shows that flow components which bypass the protective cover have to be expected in the catchment areas of the swallow-holes and sinking streams and mostly in the central zone, while protective cover is not likely to be bypassed in the N-W part of the investigated area. A large impact of the slopes must be noticed: the map is parcelled out according to this factor.

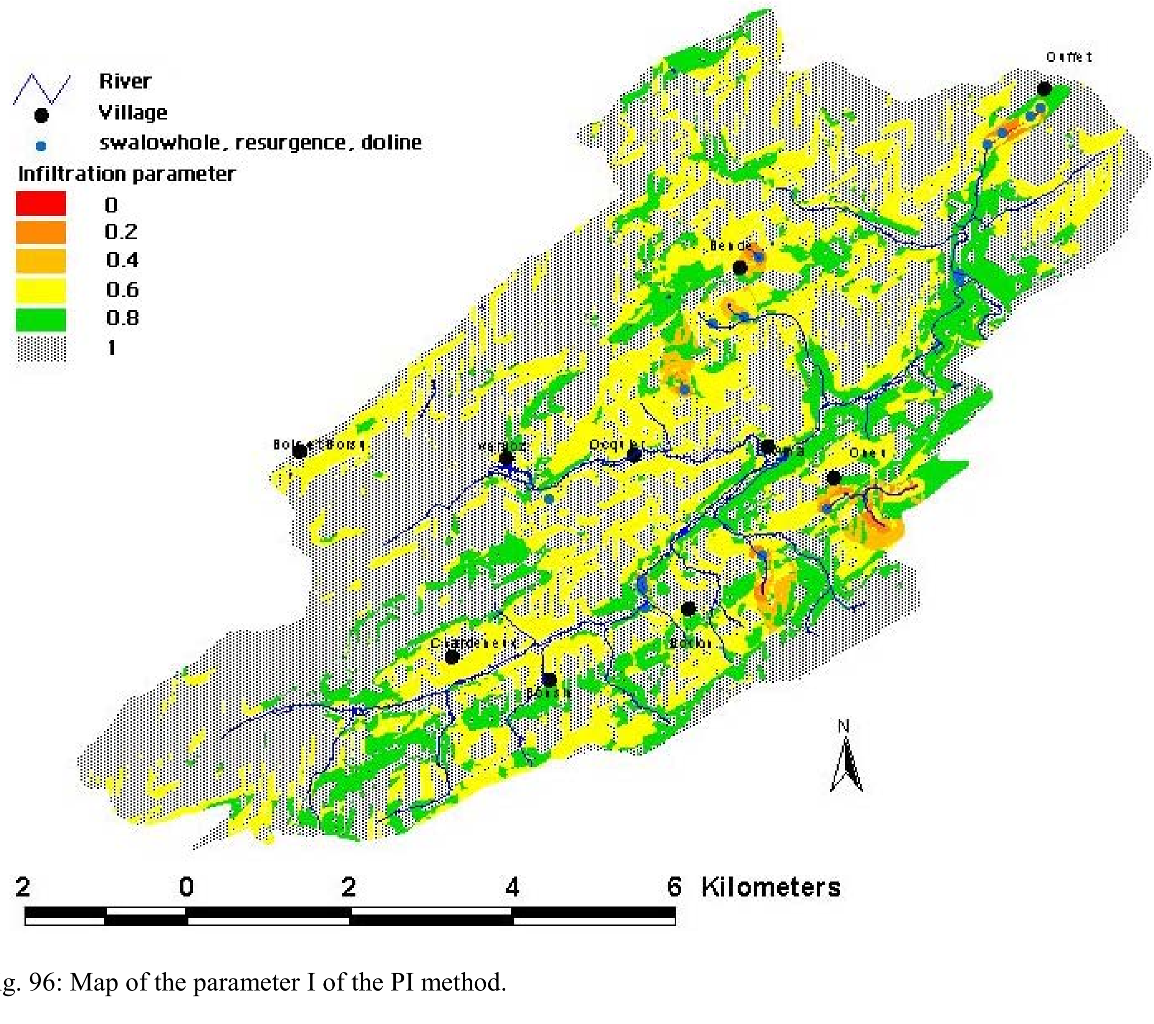

Fig. 96: Map of the parameter I of the PI method.

2.8.3.3 The PI vulnerability map

The PI vulnerability map (Fig. 97) is obtained by intersecting the P map with the I map. The most vulnerable zones are located near the swallow-holes (100 m). The slope impact must be noticed: for geological formations with the same characteristics, the greater the slopes gradient, the more vulnerable the formation appears. The vulnerability of geological formations classed as strongly fractured is moderate to high. The dolines are mapped in the more vulnerable zones. The PI map is so parcelled out that its application for land-use management seems to be questionable.

A sensitivity analysis was performed showing the importance of having a good knowledge of topography, with a detailed DTM. In a gently hilly country as the Néblon basin, the parcelling out seems inevitable. These observations are valid only if the slope computation accuracy is acceptable. This reliability depends on the DTM uncertainty and the size of the pixels. In this case, the level error between two successive pixels is estimated at 1 m maximum.

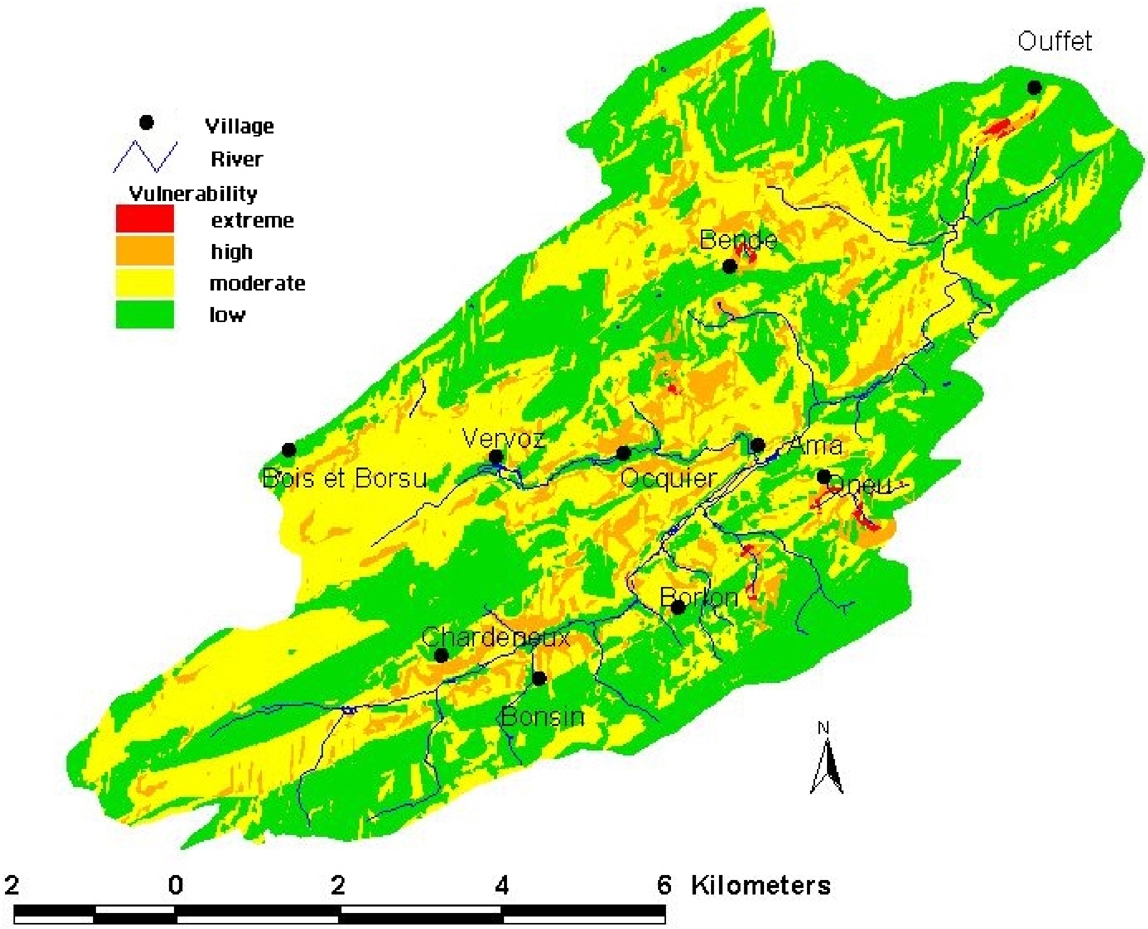

Fig. 97: Vulnerability map according to the PI method.

Five methods were applied on Néblon before applying the PI method: Gaule (1998) applied EPIK and Gogu (2000) applied GOD, ISIS, DRASTIC, EPIK and the German method and compared the results (Gogu & Dassargues, 2001).

For the limestone aquifer, results of Intrinsic Vulnerability as assessed by different methods can be summarized as follows:

- high or very high vulnerable according to GOD, ISIS and the German method;

- moderate to high vulnerable according to PI
- moderate vulnerable according to DRASTIC and EPIK;

As presented before, in the Néblon basin the limestone aquifer is in connexion with other aquifers (sandstone, silty-sandstone), there are also Tertiary and Quaternary deposits. As the PI method is applicable for all types of aquifers, but with special consideration of karst, despite of a real validation possibility, the obtained map seems to better reflect the conditions of the Néblon basin.

The results show that all those tested vulnerability methods, including PI method, remain highly subjective. They can be used as rough screening tools but unfortunately they cannot be validated, as the combination of their parameters is still empirical. In the future, a more physical point of view (see Brouyère, this report) must be adopted for obtaining "physically consistent" results.

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2.9 The Albiztur Karst Unit, Basque Country, Spain

– Comparative application of the European Approach (in this case the PI method) and the EPIK method of intrinsic vulnerability mapping –

2.9.1 General

The Albiztur Karst Unit (27 km^2) is formed by Urgonian limestones (Salubita System, 21 km^2) and by Jurassic carbonate rocks (location see Fig. 55). The Salubita spring is the main point of discharge and its mean yearly discharge is 670 l/s. The intrinsic resource vulnerability has been estimated using two methods: EPIK and the European Approach. Only the factors O (overlying layers) and C (concentration of flow) were taken into consideration. The European Approach is a conceptual model for vulnerability mapping but does not prescribe detailed assessment schemes. Thus, the tables from the PI method (Goldscheider et al., 2000) were used to determining the factors O and C (which are called P and I in the PI method).

2.9.2 Geology and geomorphology of the survey area

Most of the pervious rocks in the catchment are Urgonian, represented by sandy and massive limestones with a maximum depth of 700 m. In the southern part, sandy and clayey materials appear and they are in contact with an important fault (Azkoitia, Fig. 98). The Errezil Fault is the northern limit of the catchment. On the carbonate rocks, very important karstic depressions are present (Fig. 98), some of wich are filled with Quaternary detritic materials.

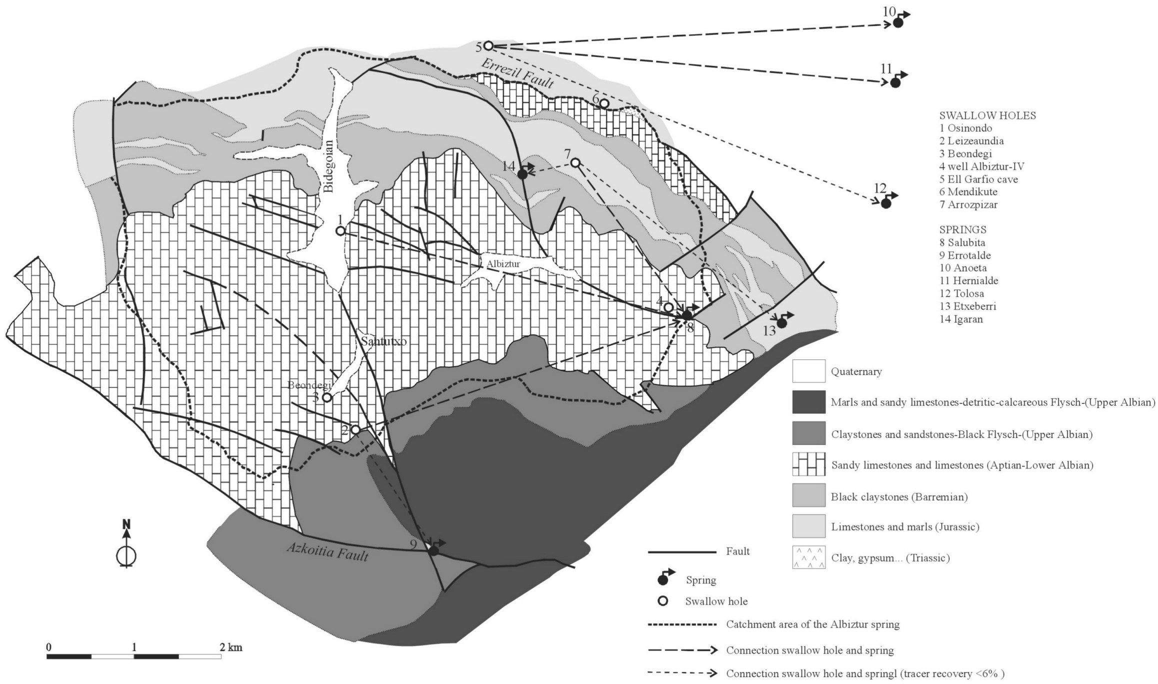

Fig. 98: Hydrogeological map of the Albiztur karst hydrogeological unit.

The most important depression is Bidegoian (1.4 km^2, maximal depth 70 m). The permanent stream of this closed depression disappears in a swallow hole (Osinondo). Tracing tests showed (Fig. 98) a very good connection with the Salubita spring; the epikarst here is linked to the karst network. In the closed depression of Santutxo (0.3 km^2) with only one small permanent stream, an old industry landfill is present. Tracing tests in this zone show a poor connection to the spring. Although this connection is proven, it seems to be limited because of the lack of an active drainage network; in this zone the epikarst is not well linked to the karst network. The small village of Albiztur is on a third depression; with maximum depth of 30 m.

The fast response of the Salubita spring and its high discharge, in comparison to the small recharge in the swallow hole, suggests the existence of a well developed epikarst (concentrating flows horizontally). Most of caves in the area have a vertical development. A well developed karrenfield of about 13 km^2 is present in the catchment. On the karrenfields, soils are very thin or absent. The separation of the Salubita spring hydrograph by using sulfate as a natural tracer (Antigüedad & Mugerza 2001) allowed us to estimate the active epikarst surface, which is of a similar size to the karrenfield surface.

About half of the area is covered by meadows and pastures and, in the western part, agriculture and farming are the main activities. At present, contaminated waters from houses and waste waters from cattle are going directly to the ground. The need of a tool for the protection of groundwater is obvious. For this purpose two methods have been applied by Mugerza (2001): EPIK (Doerfliger 1996) and a preliminary version of the European Approach. The final vulnerability map using the (preliminary) European Approach was obtained by combining the O- and the C-maps. The preparation of these two maps and their combination is explained in the following sections.

2.9.3 Intrinsic Vulnerability

2.9.3.1 O-map (overlying layers)

The O-factor indicates the effectiveness of the overlying layers. The map was constructed following the steps used to develop the P-map (protective cover) of the PI-Method suggested by Goldscheider et al., (2000) on the basis of the previously existing German GLA method (Hölting et al., 1995).

The O-map is based on the characteristics of the recharge (R), topsoil (T), subsoil (S), lithology (L), fracturing (F), thickness of each strata (M) and bedrock (B = L · F). The total protective function (P_{TS}) was calculated using the formula for the P factor of the PI method (see there). In the test site, the factors mentioned above were simplified as follows:

- Topsoil-T: According to the field capacity of soils (50 mm) in this area, it is assumed that the field capacity up to 1 m depth is less than 50 mm, so that the protective function of the topsoil (score T) acquires a value of 10.
- Recharge-R: As the mean yearly rainfall is close to 1300 mm and the potential evapotranspiration is nearly 30-50%, the recharge is higher than 400 mm/y. Because of it a value of 0.75 is used for the R factor.
- Subsoil-S: There are only two types of subsoil in the area - luvisol (S=120) and alluvial Quaternary deposits (S=75). The S values reflect the grain size distribution of these subsoils.
- Lithology-L: According to the rocks visible in the survey area, three types of lithology can be distinguished: claystone/marl (L=20), sandstone (L=15) and limestone (L=5).
- Fracturing-F: In places where the epikarst is visible, there was no difficulty in estimating a score for the F factor. There are other areas where the epikarst is not visible, but spring responses suggest that it has to be present. In this case, the existence (dimensions and location) of the epikarst has been estimated from hydrogeological and hydrochemical data, including tracer tests. For the total area, three degrees of fracturing were considered (Tab. 42) to create the F-map.

Tab. 42: Fracturing and karstification in the Albiztur test site.

Rock type	Fracturing	F
Cretacic limestones	Epikarst strongly developed and not sealed	0
Non-cretacic limestones	Strongly fractured or strongly karstified and not sealed	0.3
Rest of rocks	Moderately jointed	1

- Thickness of each stratum (m)-M: It is quite difficult to know exactly the thickness of each stratum, because of the complex geometry of the limestone deposits. Moreover, the absence of wells across the area makes estimation more difficult. Existing geophysical data give an idea of the thickness of the Quaternary deposits, but there is no information about the other materials. The thickness of soil was measured in 34 points of the area, which served to estimate the thickness of subsoil. The Tab. 43 shows the estimated thickness.

Tab. 43: Estimated thickness of the different units in the test site.

Bedrock	Estimated Thickness (m)
Claystone and sandston (Albian-Aptian). Cretacic	125
Urgonian limestone. Cretacic	90-400
Claystone and marl (Barremian). Cretacic	90-190
Marl (Albian-Aptian). Cretacic	50
Limestone (Malm-Neocomian). Jurassic-Cretacic	100
Marl and limestone (Dogger). Jurassic	90
Breccia, limestone, dolomitic rocks (Lias). Jurassic	40
Alluvial Quaternary Deposits	Estimated Thickness (m)
Bidania depression	35
Santutxo depression	15
Albiztur depression	10
Subsoil	Estimated Thickness (m)
On limestone	0.1
On claystone marl, sandstone and (locally) Urgonian limestone	0.6

As the aquifer being considered is unconfined, the factor *artesian pressure* (A) is not applicable in the test site. The combination of all factors, described schematically in this paper, gives the O-map, which shows three ranks for the O factor. The existence of only three ranks (2, 3, 4) means that for this area the total protective function (P_{TS}) is higher than 10 and less than 10000, so that the effectiveness of the protective cover is between low and high according to the classification of the PI method.

2.9.3.2 C-map (concentration of flow)

The C-factor expresses the degree to which the protective cover is bypassed by surface and lateral near-surface flow. So, it is possible to define areas where different flow processes predominate, which depend on vegetation and slope of the ground surface.

In the test site, a system based on the dominant flow process (depending on the hydraulic conductivity and thickness of the soil) was used, together with the factors vegetation and slope, to assess the proportion of surface and near-surface flow. The C factor was determined using the assessment scheme for the I factor (infiltration conditions) of the PI method (Goldscheider et al., 2000), which was, however, adapted to the characteristics of the survey area (Albiztur Unit) and the available data.

As a first step, the soil properties were determined. As seen in the S-factor of the O-map there are two main types of soil in this area. The saturated hydraulic conductivity is, in general, higher than 1.10^{-4} m/s (data collected on the field) for both the Quaternary deposits and luvisol, so that the Quaternary deposits are assumed as type A and the luvisol (depth 30-100 cm) as type C, as shown in Tab. 44.

Tab. 44: Soil properties and dominant flow processes.

soil properties	examples	dominant flow process	soil type
High permeable soil on permeable rocks	Rendzina on karst, deep sandy soil	Infiltration	A
Low permeable soil (or thin soil) on low permeable rocks	Clayey soil (luvisol)	Surface flow	C

The soil properties (type of soil) together with the land use (forest or meadow/pasture) and slopes allowed the C’ factor (C’-map) to be established, which reflects the extent of surface subsurface flow. The slope was classified using the divisions of the Basque soil mapping guidelines instead of the classification proposed in the PI method. Tab. 45 shows the combination of vegetation, type of soil and slopes, and the score assumed for each situation.

Tab. 45: Determination of the C’ factor dependent on the soil type, land use and slope gradient.

	Land use: FOREST		
Soil type	< 5 %	5 – 30 %	> 30 %
A	1	1	0.8
C	0.8	0.6	0.4
	Land use: MEADOW/PASTURE		
Soil type	< 5 %	5 – 30 %	> 30 %
A	1	0.8	0.6
C	0.6	0.4	0.2

The final C-map was obtained by combination of the C’-map with the map of the catchment areas of sinking streams. For the Surface Catchment Map, three zones were delineated, as shown in Tab. 46.

Tab. 46: The C-map is obtained by combining the information on the surface catchment map and the C’ factor.

Surface Catchment Map	**C' factor**				
	0.2	**0.4**	**0.6**	**0.8**	**1.0**
Swallow hole, sinking stream, 10 m buffer	0.0	0.0	0.0	0.0	0.0
100 m buffer	0.2	0.4	0.6	0.8	1.0
Catchment of sinking stream	0.4	0.6	0.8	1.0	1.0

2.9.3.3 Final vulnerability map (OC-map)

The OC vulnerability map shows the spatial distribution of the protection factor π, which was calculated analogously to the formula proposed in the PI method [$\pi = O \cdot C$]. As shown previously, the O factor is divided in three ranks (2, 3, 4) and the C factor in five ranks (0.0 - 1.0). So, π factor ranges between 0.0 and 4.0, where high values represent a high degree of natural protection and low vulnerability. In Tab. 47, the values obtained for the Albiztur Unit are shown. These results are shown in Fig. 99, which is the final vulnerability map (OC map).

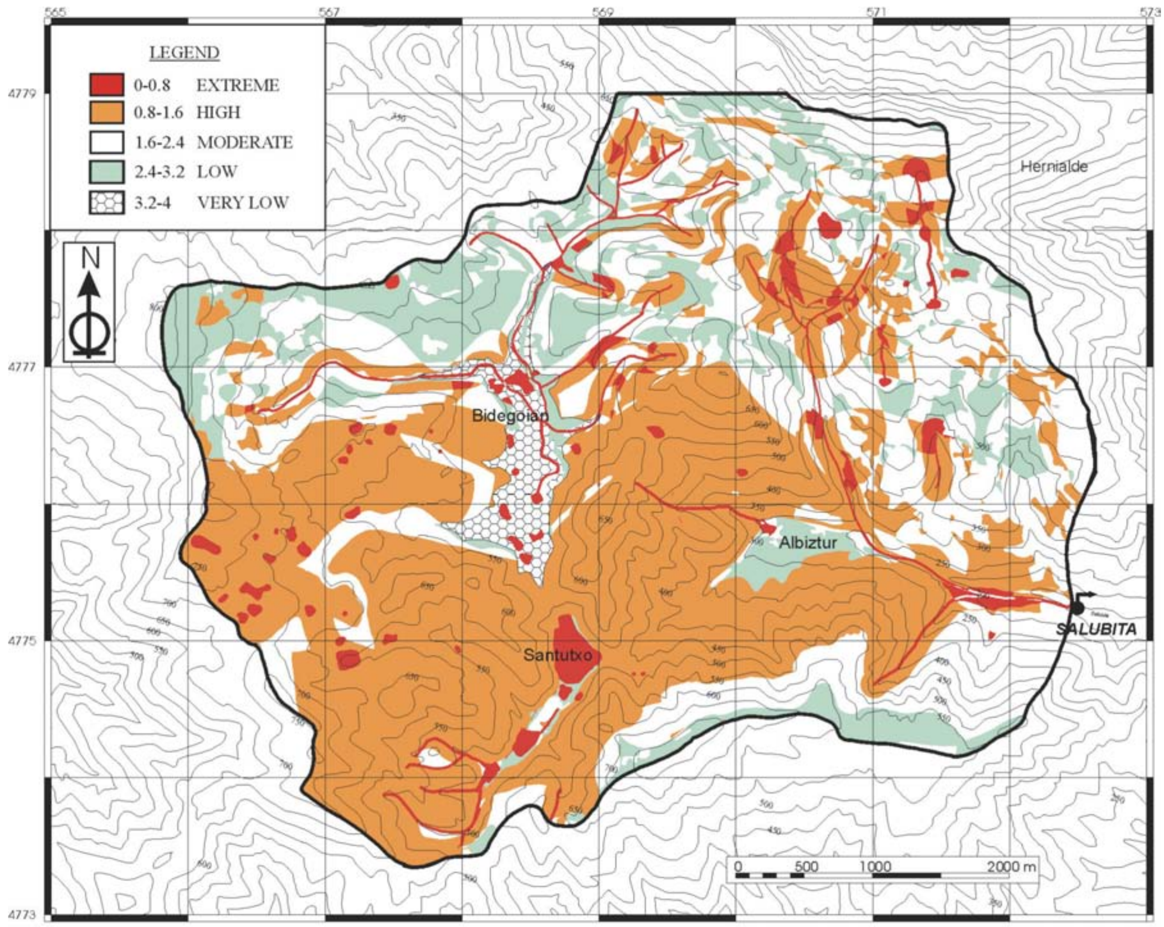

Fig. 99: Vulnerability map (European Approach) of the Albiztur test site.

Tab. 47: Legend for the vulnerability map, the O and the C map (modified after Goldscheider et al. 2000).

Vulnerability map Vulnerability of uppermost Aquifer		**O-map** Effectiveness of Protective Cover		**C-map** Degree of Bypassing	
Verbal description	π-factor	Verbal description	O-factor	Verbal description	I-factor
Extreme	0-0.8	Very low	-	Very high	0-0.2
High	>0.8-1.6	Low	2	High	0.4
Moderate	>1.6-2.4	Moderate	3	Moderate	0.6
Low	>2.4-3.2	High	4	Low	0.8
Very low	>3.2-4	Very high	-	Very low	1

2.9.4 Conclusions and comments

- In this area the EPIK method gives three vulnerability ranks (low, moderate and high). According to these results all the Urgonian calcareous rocks are high vulnerable and the low vulnerability coincides with the less pervious rocks.
- The European Approach (in this case the PI method) gives five vulnerability ranks (very low, low, moderate, high and extreme). This method makes differences within the Urgonian rocks, giving an extreme vulnerability for dolines and swallow holes and a high vulnerability for the rest of the Urgonian calcareous rocks. The European Approach provides more detailed information than EPIK.
- The European Approach takes into account the *bypass* effect via the C factor, which is very important in karstified areas. For the Bidegoian depression (filled by Quaternary de-

posits), the EPIK gives a moderate-high vulnerability because these sediments (maximal depth 70 m) are on very pervious Urgonian limestones. For the same materials, the PI provides a general low-very low vulnerability but an extreme vulnerability for the swallow holes present on the Quaternary sediments.

- The European Approach (in this case the PI method) is more objective than EPIK. The method of estimating the parameters of EPIK (for example the epikarst) is very subjective. The non-existence of evidences on the surface doesn't mean that there is no epikarst and the existence of evidence on the surface doesn't give information on whether the epikarst is active or not.

Acknowledgments: This work was funded by the CICYT HID99-0333 project.

2.9.5 References

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2.10 Lincolnshire and Schwyll, UK

– Application of a localised European Approach (LEA) in two test sites

2.10.1 Lincolnshire test site

2.10.1.1 General

The test site is located to the west and north-west of the village of Castle Bytham in the county of Lincolnshire which is in eastern England (Fig. 55). The field area is about 12 km^2 in size and is in the catchment of the Glen Brook. The area is south of the West Glen river system where the groundwater response to rainfall has been modelled by Rushton and Bradbury (1998).

Four vulnerability mapping methods were applied to this test site: DRASTIC, the Irish Method, EPIK and Localised European Approach (LEA).

2.10.1.2 Geological characteristics of the investigated area

The geological setting is one of Middle Jurassic to Upper Jurassic. A summary is found in the table below (Downing and Williams, 1969 cited in Rushton and Bradbury, 1998)::

Tab. 48: Geological characteristics of the test site.

Age	Geology	Description	Thickness
Upper Jurassic	Great Oolite Limestone	Limestone with thin marl and clay beds	2.5-8.0m
Middle Jurassic	Upper Estaurine Series	Sand, clay, limestone	5.4-14.0m
	Lincolnshire Limestone	Oolitic and argilaceous limestone with calcareous sandstones and cementstones	0-40m
	Northampton Sand	Sandstone, sands and limestone	0-10m

Within the Lincolnshire Limestone aquifer system there are three formations: the Lincolnshire Limestone, the Grantham Formation and the Northampton Sand Formation (Rushton & Tomlinson, 1999). The Great Oolite Limestone mentioned in the above table is referred to as the Blisworth Limestone by Rushton & Tomlinson (1999).

The landscape is a gently undulating one. There are some east-west trending dry valleys. These valleys are rather gentle and the topography is one of rolling hills. Other features of note include the swallow holes in the north of the area, which accept southerly flowing re-charge. About 30% of the area is covered in boulder clay, which is 14.5 m thick in the centre of the test site.

2.10.1.3 Intrinsic vulnerability mapping

Using the LEA (Localized European Approach) the majority of the area is classed as moderate vulnerability – boulder clay over limestone (Fig. 100). Some of the area consists of Upper Estaurine Series over limestone, which is classed as moderate vulnerability. Areas of soil over limestone were classed as an area of very high vulnerability, while the zones around the sinking streams and dolines yield an extreme vulnerability rating. Hence the sinking streams and dolines stand out from the background vulnerability ratings as being extremely vulnerable.

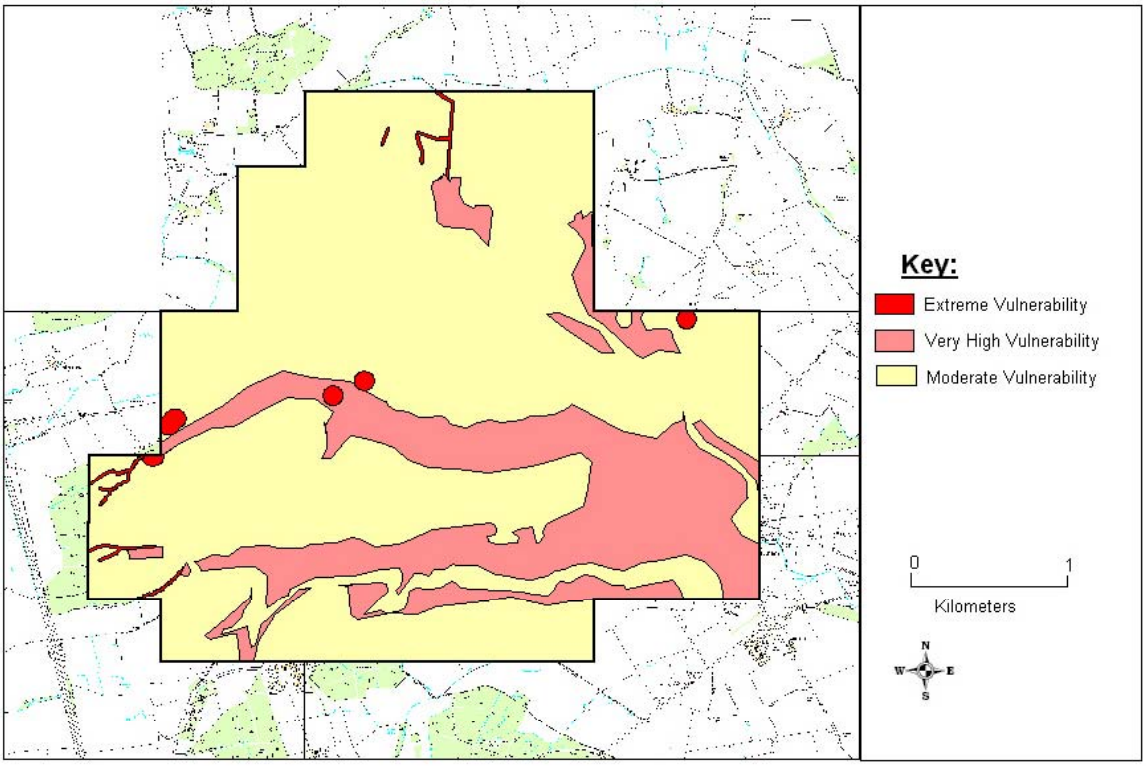

Fig. 100: LEA Vulnerability Map of Castle Bytham Test Site, Lincolnshire, England.

2.10.1.4 Validation by means of conductivity data

The conductivity data for the Lincolnshire Limestone setting is presented in Fig. 101.

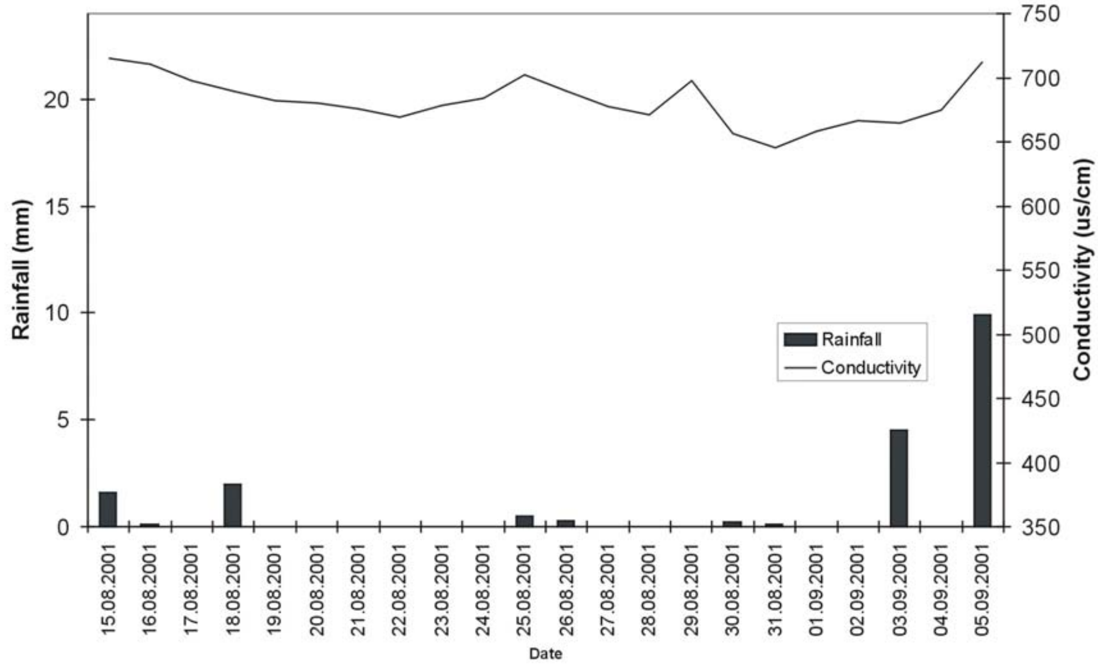

Fig. 101: Conductivity and rainfall for mid-August – September 2001 Castle Bytham, Lincolnshire.

The data illustrates a relatively high conductivity, which may imply that the groundwater has been in the karst system for a relatively long time. The changes in conductivity after rainfall events are gradual and gentle. Hence the small areas of extreme vulnerability and large areas of moderate vulnerability are probably a reasonably accurate description of the vulnerability of the test site.

2.10.2 Schwyll test site

2.10.2.1 General

Schwyll is the name given to a spring found on the southern rim of the South Wales Coalfield (Fig. 55). Carboniferous limestone outcrops around the coalfield and it is both folded and faulted. Schwyll is located south of the town of Bridgend. The test site is about 3 km^2 in size.

2.10.2.2 Geological characteristics of the investigated area

The formations within the Carboniferous Limestone vary in thickness from 8 m to 140 m. One formation consists of dolomitic limestone, while the others consist of shelly, oolitic, and crinoidal limestones.

The landscape of the field area consists of a plateau area dissected by a network of dry valleys. The field area stands proud above the flatter land to the north. Glacial outwash deposits of sands and gravels are found in the northern end of the test site along the River Ewenny. Head deposits are found on the lower slope of valleys (Wilson et al, 1990). Alluvium deposits – consolidated silts and soft laminated clays interbedded with non-cohesive loose sands and gravels and locally contains peat intercalations. Alluvium deposits are found on the Ogmore and Ewenny and Alun valley floors. A description of the hydrogeology of the Schwyll spring may be found in Hobbs (2000).

2.10.2.3 Vulnerability mapping

The protective cover map was constructed based on travel times calculated from the D'arcy flow equation. Hence the sand and gravel deposit and karst limestone yield a time of travel of hours or minutes and hence a very high vulnerability rating. The dune sands result in a time of travel of days, very high vulnerability. The thickness of the head deposits was 2 m, hence the head deposits were classed as very high vulnerability (Fig. 102).

On the flow concentration map a 10 m wide default contributing area was placed around the Alun and Ogmore, which loose surface water to the underlying aquifer (Hobbs, 2000). There was no need for one around the Ewenny river as it sinks outside and upstream of the border of the test site. A circular 50 m extreme vulnerability zone is placed around the doline.

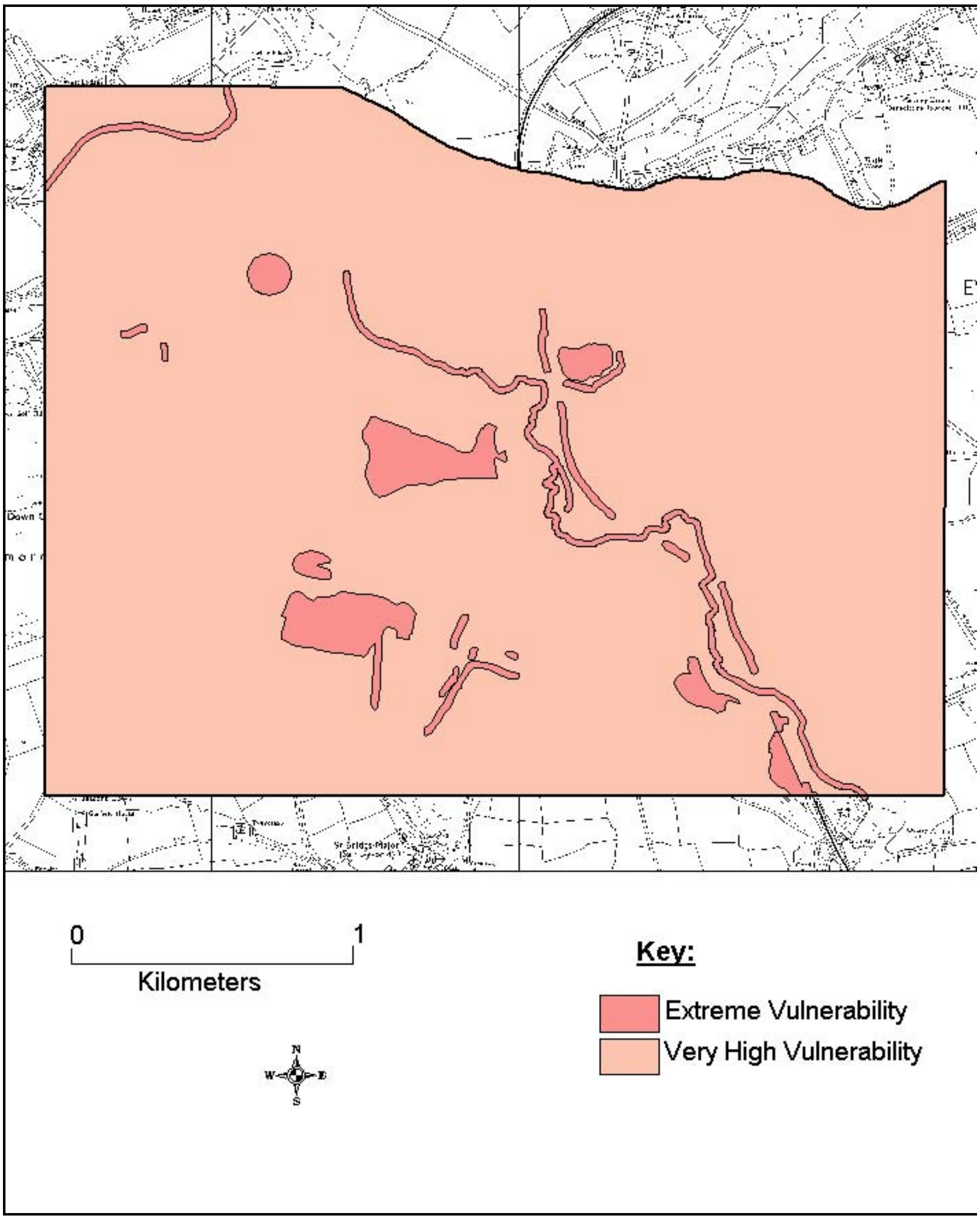

Fig. 102: LEA Vulnerability Map for Schwyll, Wales.

2.10.2.4 Validation by means of conductivity data

The conductivity is much lower for Schwyll that for Castle Bytham. This implies that the recharging water has not been in the karst system for a considerable time period. Hence it has not had the opportunity to dissolve constituents of the bedrock to its potential level.

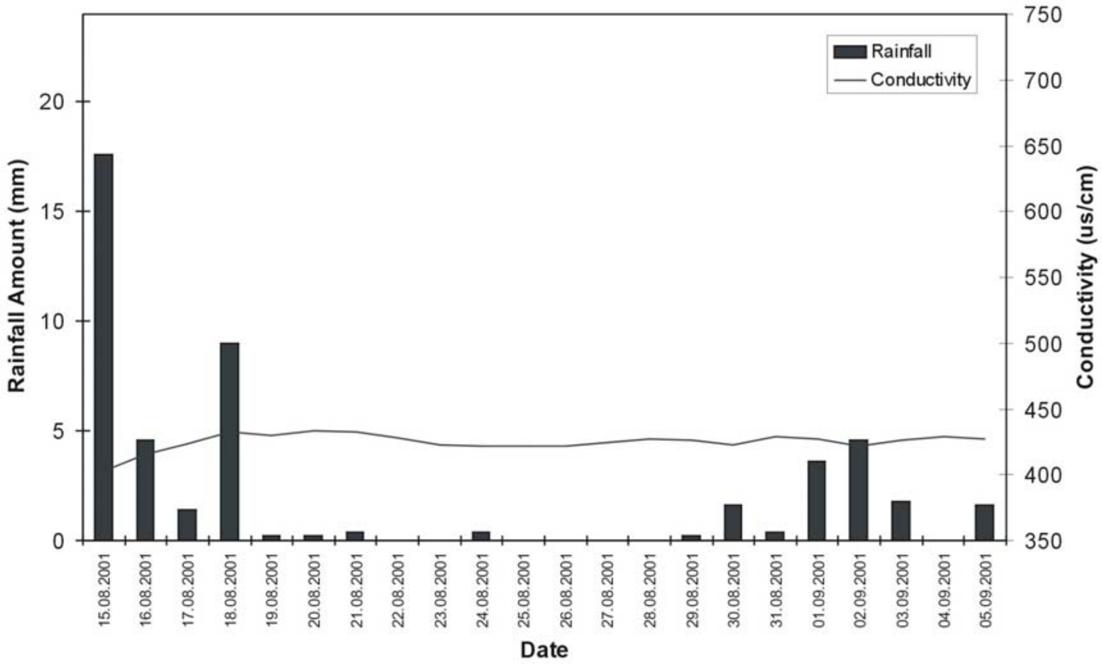

Fig. 103: Conductivity and rainfall mid-August to September 2001, Schwyll, Wales.

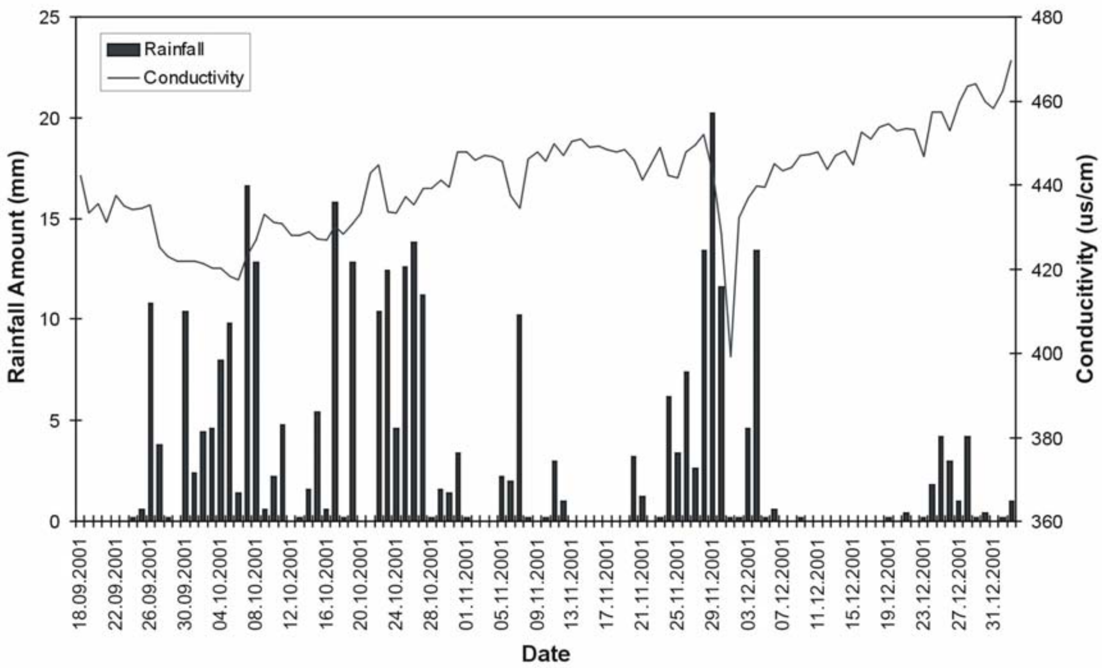

Fig. 104: Conductivity and Rainfall Data September – December 2001.

Considering the rainfall events and conductivity of Aug 15th-Sept 5th for Schwyll it is clear that the conductivity remained low, around 430 µs/cm (Fig. 103). This low conductivity implies that the recharge has become groundwater very rapidly and has not dissolved many constituents of the bedrock. There are also rapid decreases in the conductivity after some rainfall events, particularly after the 29.11.02 event (see Fig.5), which indicates a high degree of vulnerability and little effective buffering by the protective cover. Hence the conductivity agrees with the New Method classification of Schwyll test site, which is 96% very high vulnerability. Using Quinlan et al.'s 1991 classification system, the coefficient of variation of the conductivity (2.9%) indicates that Schwyll is moderately sensitive (see Fig. 104).

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2.11 Muranska Planina Plateau, Slovakia

– Intrinsic vulnerability mapping using a preliminary version of the European Approach

2.11.1 General and geological characteristics of the investigated area

The Muranska Planina Plateau is a karst system in Central Slovakia, covering an area of 126 km^2 (Fig. 55), mainly formed by Middle Triassic limestones and marginally by Middle Triassic dolomites. It is a morphologically shifted carbonate plate with edge slopes elevated 300 to 500 m above the neighbouring areas. No deeper boreholes exist within the region; however, some drillings were undertaken in the vicinity of springs. Water from the majority of the exploited springs is directly taken into pipelines.

2.11.2 Karst and Hydrogeology of the area

Numerous sinkholes, swallow holes, dry valleys and also several semi-poljes are present on the flat top surface of the Muranska Planina Plateau. Some parts of the karst conduit network were identified either by tracer experiments or by means of curve analyses of spring discharges. Explored caves are only 10 – 100 m long and surface karst features prevail. The Slovak Hydrometeorological Institute undertook extensive gauging of precipitation, surface stream discharge and spring discharge in the past. All water inputs and outputs were monitored by a very complex network of 28 spring gaugings, 34 surface stream gaugings and 5 rain gauging stations, during the complete hydrological decade of 1971-1980 (Kullman, 1990). A major part of this monitoring network is in use already today. Important springs are discharging on the Plateau edges, on the contact with granites, formed by an important fault. There are approximately 16 springs with a mean discharge > 10 l/s. The major spring outlet "Pod hradom" reaches 6130 l/s during the snow melt period, but falls down to 4 l/s in summer periods. The mean discharge is 245 l/s. Other major springs with mean discharge over 100 l/s are "Tisovec-dolny" (101 l/s), "Muran – v obci" (116 l/s) and "Pastevnik" (193 l/s). Based on the discharge curves, karstification processes are unequally spread through the structure. Major springs are exploited as drinking water sources for the whole region beneath. Reliable data on water consumption are available for a three-year period. Water quality monitoring was concentrated on several important springs with very high frequency of observations to reveal water-mixing relations. A series of basic qualitative data are available from all exploited springs.

Several qualitatively interpreted tracing experiments were carried out by local speleological groups on a limited number of swallow holes located on the Plateau top. In the framework of the EU PHARE project EC/90/WAT/11b, co-ordinated by the Slovak Ministry of Environment in the period of 1995-1997, a regional groundwater flow model was set and calibrated for the whole karstic structure (Fendek in Witkowski et al., 1997).

2.11.3 Intrinsic Vulnerability Mapping

In the year 2000, a task group of COST 620 proposed a "preliminary European Approach" reflecting an intermediate state of the discussion within working group 1 (Daly et al., 2000, unpublished, internal paper). The proposed method uses the two factors "O" and "C" and provides tables and formulae for their quantification. It was applied in the Muranska Planina Plateau for the first time and the results helped significantly the further development of the "European Approach" presented in this final report. Groundwater vulnerability map using the "O" and "C" factors was based on the evaluation of karstification from field investigation and using speleological and remote sensed data. Soil thickness and soil properties were obtained from 255 hand-dug holes (2 holes per 1 km^2). Mean estimated soil thickness was 40 cm, with standard deviation of 21 cm. A GIS and a digital terrain model were used to re-classify soil properties to cover the area. For unsaturated zone thickness determination, groundwater levels from the regional groundwater flow model were used. Land-use data were derived from digital topography maps and slope characteristics from a precise digital elevation model.

Following the concept of the "preliminary European Approach" (Daly et al., 2000), based on the PI method principles (Goldscheider et al., 2000), four layers have been taken into consideration during the construction of the "Overlaying Layers" – O map: **topsoil**, **subsoil**, **non-karst rock** and **unsaturated karst rock**. The protective functions of the topsoil and subsoil have been calculated using data obtained from field soil sampling. As a base for protective function evaluation for the latter two layers, the simplified geological map has been used. Thus every lithological type of rock has been assigned corresponding permeability and porosity values, as seen in Tab. 49.

Tab. 49: Permeability and porosity values of rock types in the Muranska Planina Plateau, key parameter and modifier values for bedrock.

Rock Type	Permeability [points/m]	Porosity [modifier]
Limestones, karstified	5	0.5
Limestones, not karstified	30	0.5
Dolomites	60	1.0
Granites, Jurassic limestones	30	0.2
Lower Triassic shales	60	0.1

By subtracting the modelled altitudes of groundwater table (Fendek in Witkowski et al., 1997) from surface elevations (Fig. 105), the thickness of the unsaturated zone was obtained. Protective function values of the bedrock have been summed with those of topsoil and subsoil and resulting values have been interpolated over the whole pilot area, resulting in the "O" - factor map.

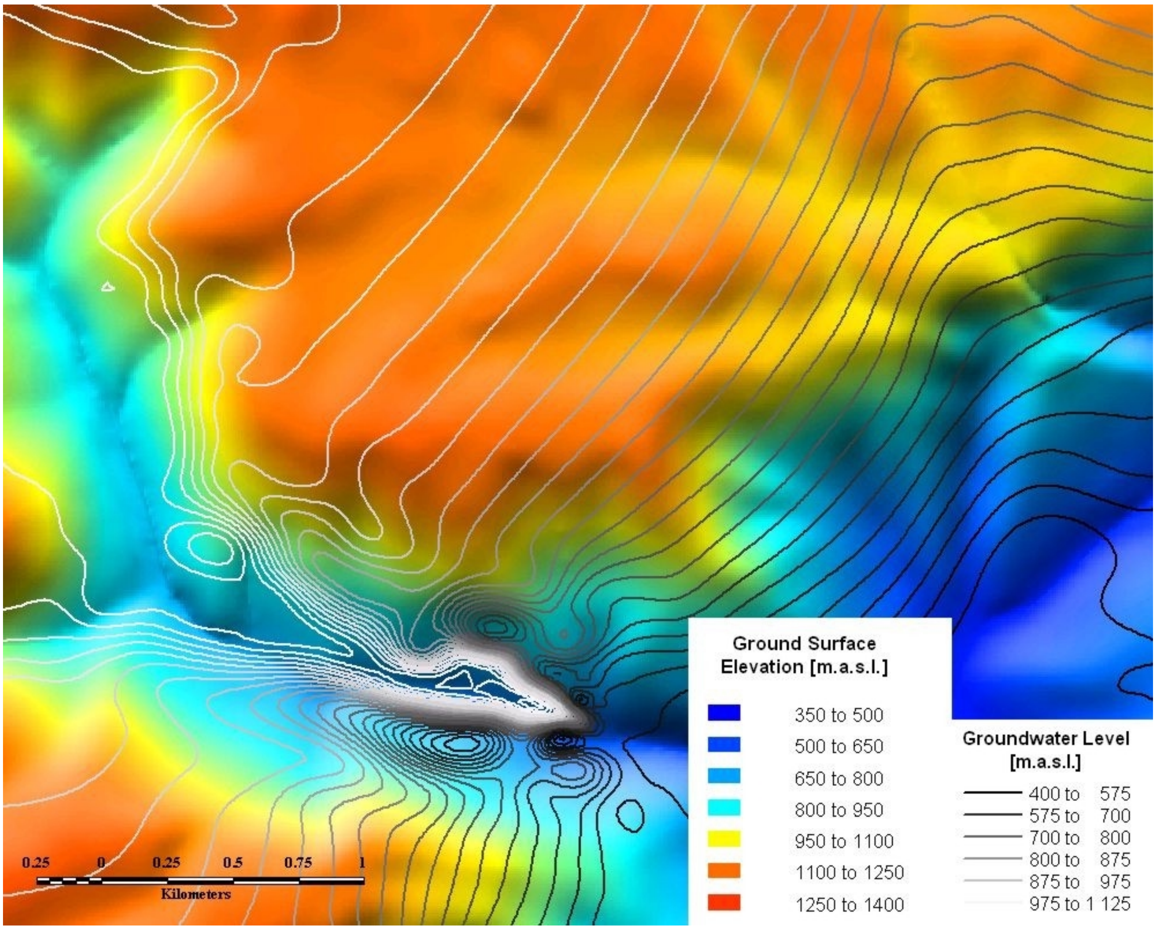

Fig. 105: Determination of the thickness of the unsaturated zone.

The first step in preparing the "Flow Concentration" – "**C**" map was to determine the dominant flow process dependent on soil properties. To achieve this, two partial maps have been prepared, depicting soil and bedrock permeability. For soil permeability, already calculated (O – map) protective function values of topsoil and subsoil were used. Depending on planar curvature and slope of the land surface, the whole area has been divided into nine geomorphological categories (Fig. 106). A value of the protective function of the soil was calculated for every category, based on the mean value of all sampling points within this category. The nine categories have been then reduced into two classes: low and high permeable soils. For the bedrock permeability map, a less complicated approach has been used. Carbonate rocks have been assumed to be highly permeable and the rest a low permeability.

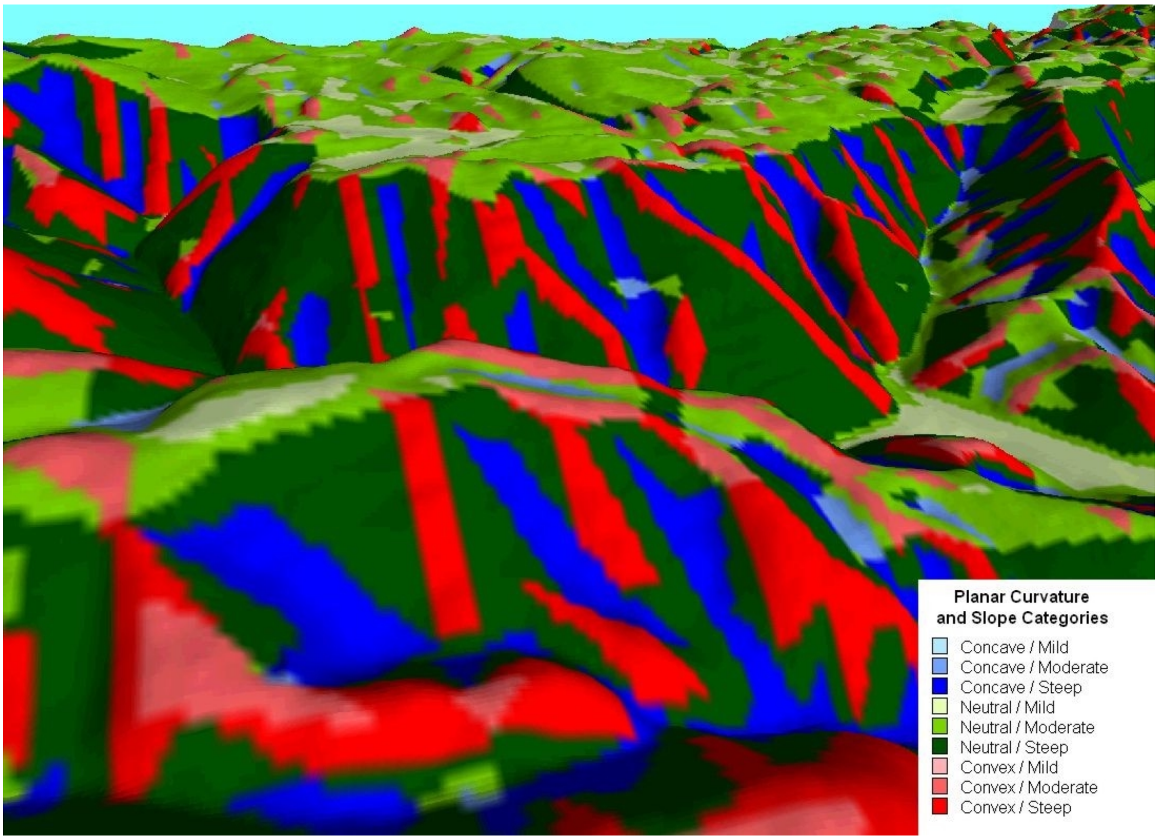

Fig. 106: A perspective view over an area depicting different morphological categories

By combining those two maps, three types of flow have been identified:

- high permeability soil, no low permeability layers (Type **A)**
- high permeability soil on low permeability layer (Type **B)**
- low permeability soil or shallow soil on low permeability layer (Type **C)**

Using these data, together with land cover and slope maps, the **C' - factor** values have been calculated. Later, the C' - factor map was combined with the map of swallow holes and sinking streams buffers, and the "**C"** - factor map was generated in this way.

The final vulnerability map has been created by overlaying the **O** map with the **C** map and multiplying the values. The resulting values ranging from 0 to 850 have been divided into 5 classes of vulnerability.

2.11.4 Discussion

- Groundwater vulnerability should be related to a certain *map scale* – it is not possible to apply it to all types of maps (state, regional, local), mainly because of buffer extension (10 or 100 m) of the swallow holes.

- Uncertainty connected to different data sources is very often unequal – e.g. in the case of Muranska Planina Plateau, the main role is played by the vadose zone thickness yet no borehole date were available. Groundwater table level was produced by mathematical modelling process, but does a groundwater table exist in all karst environments? In this way, the data availability plays the main role in the vulnerability estimations.

- Dealing with the "O" and "C" parameters of the "preliminary European approach", *one parameter* (soil & rock permeability) is *taken into account* ***twice***, in both O and C factors, but in opposite ways – giving contradictory results.

2.11.5 References

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2.12 Veszprém-Kádárta Plateau, Transdanubian Central Range, Hungary

– Intrinsic vulnerability mapping using the "preliminary European Approach", and hazard mapping –

2.12.1 Geographical and Geological Overview

The test site is the Veszprém-Kádárta karst plateau, which is about 20 km^2 in extent. It is situated in the Transdanubian Central Range near Veszprém, a town with 64,000 inhabitants (Fig. 55). The plateau serves as the catchment area for the Kádárta springs. The altitude of the plateau ranges between 300 m and 200 m. The mean annual temperature is 10°C. The Kádárta springs provide part of the drinking water supply of the town.

The Veszprém-Kádárta plateau is made up by NE-SW striking imbricate units. These consist of siliciclastic and carbonate sediments of Permian to Triassic age dipping 15° to 30° to the NW. The imbricate package is dissected by a set of NS striking normal and strike-slip faults. These relate to later stages of the Eoalpine evolution of the Transdanubian Central Range (DUDKO, 1991). Major valleys follow the strike of these younger faults showing that the actual topography is tectonically controlled. The Kádárta springs are located in one of these tectonically controlled valleys. At Kádárta the main aquifer has developed in a thick unit of greyish white Middle Triassic dolomites (the Budaörs Dolomite Formation), which is about 1000 m thick (BUDAI-CSILLAG, 1995). To the SE, this unit is underlain by a more calcareous but less permeable unit of likewise Middle Triassic age. To the NE, Late Triassic calcareous marls overly the Middle Triassic dolomites (the Veszprém Marl Formation). All over the N and NE part of the plateau, these formations are exposed on the surface, which is covered by a thin discontinuous layer of poor rendzina-type soils. The S-SW part of the area is covered by a 4 to 6 m thick blanket of Quarternary loess. This cover is absent only in the close vicinity of the karst springs, where only a thin soil layer covers the rock surface.

The borders of the catchment area are unequivocal and distinct both geographically and geologically. The infiltrating precipitation is drained by the Séd creek and by a number of karst springs. Spring discharge occurs in the NS striking fault controlled valleys at the contact of the aquifer and the less permeable marls. Hydraulic gradient is about 10 to 12 m/km towards the S-SW. The Kádárta group of springs has two discharge points. The Western and the Eastern Springs are located about 100 ms apart in the opposite sides of the Kádárta valley. The total yield of these springs is about 8,000 to 11,000 m^3/day.

The Kádárta catchment area, which is underlain by the Budaörs Dolomite Formation, is an example of a karst aquifer characterized by diffuse porosity rather than by karst conduits. The relevant karst forms are dry valleys and one big doline.

2.12.2 Intrinsic vulnerability mapping

In the Veszprém-Kádárta plateau, two methods (EPIK, sensu DOERFLIGER 1996 and DOERFLIGER & ZWAHLEN 1998; and DRASTIC, sensu ALLER et al.1987) had previously been tested and reported (MÁDL-SZŐNYI & NYÚL 2000, NYÚL & MÁDL-SZŐNYI 2000).

This section gives an overview of the results achieved when testing the "preliminary European Approach", which gave the best results for this test site.

Vulnerability assessment was carried out following the recommendations of the COST Action 620 Working Group 1 Task Group Meeting (Karlsruhe 1-3 June, 2000) (DALY et al. 2000, DALY et al. 2002). Determination of factors O and C was undertaken as required by the above recommendations. To determine P and K, the suggestions of an internal report of COST Action 620 were followed. Scale and the adequate precision of the application were 1:25.000.

2.12.2.1 Overlaying layers (O-factor)

Two kinds of topsoil, rendzina and brown forest soil, were distinguished in the test area (MÁDLNÉ-SZŐNYI & NYÚL, 2000). The thickness and hydraulic conductivity of the topsoil was examined. Archive soil maps (TEÖREÖK, 1941) were used. Additional information was provided by 120 shallow boreholes drilled within the framework of the present project. The predominant grain size of these soils is either „pebble" or „loam". Hydraulic conductivities were estimated on the basis of grain size. Rendzinas, being rich in the coarse fraction and having a high macroporosity, were estimated to have a hydraulic conductivity of $K=10^{-2}$ m/sec. Clay-rich, loamy brown forest soils were characterized by an estimated 10^{-7} m/sec hydraulic conductivity (Tab. 50).

Tab. 50. Assessment-scheme for the topsoil

Type of topsoil	Grain Size	Log K (m/s)	Scores/m from GS (S)	Macropores (multiplicator) (M)	Scores/m (S*M)
Brown forest soil	Clayey loam	-7	60	0.5	30
Rendzina	Gravel	-2	1	0.1	0.1

In the Kádárta test site the only type of subsoil is sandy loess. For the estimation of thickness of loess, archive boreholes and our own drillings were used. Since according to our field observations, no preferential flow paths have developed in the loess blanket, a hydraulic conductivity of 10^{-4} m/sec was estimated for the subsoil (Tab. 51).

Tab. 51. Assessment-scheme for the subsoil

Grain Size	Log K (m/s)	Scores/m from GS (S)	Preferential flow (multiplicator) (M)	Scores/m (S*M)
Sand	-4	5	1	5

Within the area the only substrate, which might be qualified as non karst rock, is the Veszprém Marl Formation (PEREGI & RAINCSÁK, 1980, 1983; KOVÁCS et al. 1998). It has considerable water retention capacity (KOVÁCS, 1998) so it was considered as a part of the karstic unsaturated zone.

The unsaturated karst bedrock consists of the unsaturated zone of the water-bearing karstified unit and of the epikarst if it is present. In Veszprém-Kádárta plateau, the water-bearing karstified unit is made up by dolomite, marl and limestone (PEREGI & RAINCSÁK, 1980, 1983, KOVÁCS et al., 1998), with dolomite as the predominant rock type. Specific epikarst has developed on the top of bare dolomite. It has a significant water retention capacity and a high protective capacity, but it was not considered within the framework of this assessment.

Tab. 52. Assessment-scheme for the non karst rock and the unsaturated karst rock

Type	Rock permeability			Primary porosity (multiplicator) (M)	Scores/m (S*M)
	Fissuring and karstification	Log K (m/s)	Scores/m (S)		
Veszprém Marl	Fissured, not karstified	-7	60	0.1	6
Sándorhegy Limestone	Sightly karstified	-6	30	0.2	6
Budaörs Dolomit Ederics Limestone	Fissured	-5	20	0.2	4

To estimate the thickness of the unsaturated karstic bedrock, we used the information provided by the standard topographic sheet, the contour map of the karst water table (KOVÁCS et al., 1998) and the isopach maps of the loess subsoil and the topsoil cover. With the help of the ArcView 3.1 program package, the elevation of the water table was subtracted from the standard topography sheet, and the resulted thickness was reduced by the combined thickness of topsoil and subsoil (where relevant). In this way, the isopach map of the unsaturated karstic bedrock was achieved.

Based on the estimated hydraulic conductivity of the rock formations of Veszprém-Kádárta plateau (KOVÁCS et al., 1998), the protective function of the karstified formations of the test site was calculated (Tab. 52).

According to the „origin-pathway-target" model as recommended by the Working Group of COST Action 620, the O-factor was assessed with the water table as a "target". Based on the scores representing the protective function of the individual O-layers per unit soil/rock thickness, using GIS, we have calculated the number of scores for every O-layer and after that the total number of scores for the O-factor. In this way the protective function of the overlying layers could be assessed (Fig. 107).

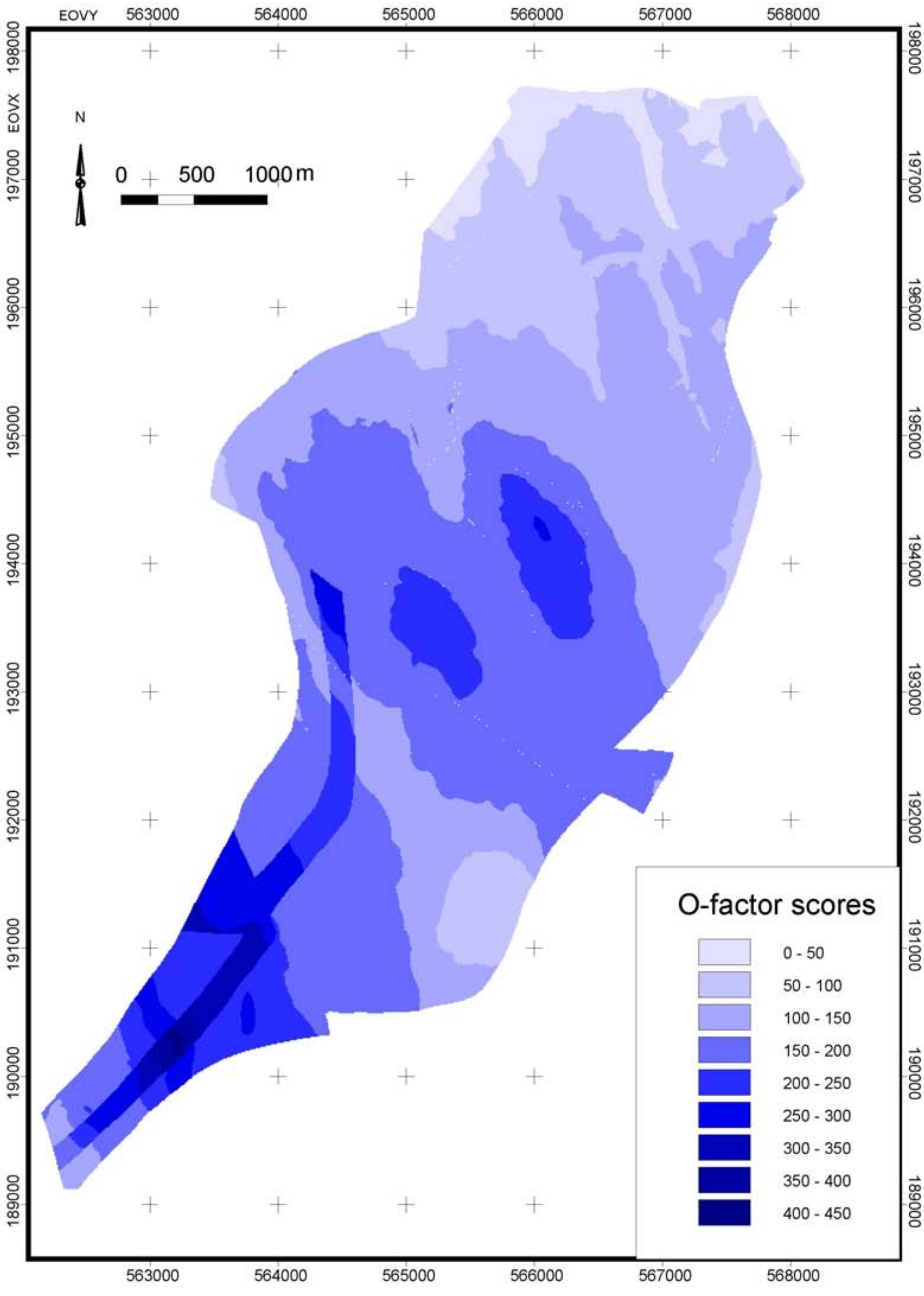

Fig. 107. Final map of O-factor

The O-factor was assessed also by using the travel-time concept. Calculations using the maximum hydraulic gradient and taking into account the thickness and hydraulic conductivity of each O-layers, we have estimated the residence time for each of those formations. By integrating these data, the total travel time to the karst water table could be calculated. Travel-time categories (0-3 days, 3-30 days, 30 days-1 year, 1-10 year) were established by taking into consideration the recommendations of COST Action 620 (Fig. 108). Macroporosity was neglected at this stage.

Fig. 108: Final resource vulnerability map

2.12.2.2 Flow concentration (C-factor)

Estimation of C-factor was undertaken as suggested by DALY et al., (2000).

The occurrence of the soil types identified (TEÖREÖK, 1941; MÁDL-SZŐNYI & NYÚL, 2000) in the test site is clearly lithologically controlled: rendzinas occur always on dolomite whereas brown forest soils are characteristic on loess substrate (PEREGI & RAINCSÁK, 1980; 1983, BUDAI et al., 1999). Rendzinas rich in rock fragments are classified as „highly permeable soils with no low-permeability layer" (type A). In this case, infiltration is the dominant flow process. Brown forest soils developed on loess are „low permeability soils or shallow soils over low permeability layer" (type C). (Tab. 53). Surface flow is the dominant flow process in this case.

Tab. 53: Dominant flow process

Type	Soil properties	Dominant flow process	Type
Rendzina on dolomit	High permeable soil, no permeable layers	Infiltration	Type A
Brown forest soils on loess	Low permeable soil or shallow soil on low permeable layer	Surface flow	Type C

Using the digitised version of the 1:10.000 scale topographic sheet, we constructed a slope-% map. Because of the essentially flat plateau-like morphology, the slopes are lower than 3.5%. Steeper slopes can be found in the valleys and sides of the hills. With the help of the landuse map (KOVÁCS et al., 1998, modified) „forests" and „other agriculturally cultivated or utilized" areas (arable land, meadow, pasture) were delineated. Finally, dominant flow processes, landuse and slope-categories were all combined to estimate flow concentration. The calculated scores from 1 to 0,4 represent decreasing flow concentration. (Tab. 54)

Tab. 54. Determination of the C'-factor

Soil properties	Forest		Medow/pasture	
	Slope		Slope	
	<3.5 %	3.5-27 %	<3.5 %	3.5-27 %
Type A	1	1	1	0.8
Type C	0.8	0.6	0.6	0.4

The assessment of flow concentration should be based on the delineation of those areas where protective cover is bypassed by runoff and infiltration (surface water-courses, swallow holes etc.). In the Kádárta test site such elements were not detected. We concluded that in this area no flow concentration has developed. Even though theoretically the potential of flow concentration cannot be completely discounted, it is not considered. The large doline in the SE part of the catchment area is not fed by any watercourse, so it can be excluded from flow-concentration. The same is true for the dry-valleys.

The C-factor was therefore given a uniform score of 1.0 for the whole area. The C-factor (multiplication by 1.0) therefore has no influence on the vulnerability assessed on the basis of O-factor. (Tab. 55)

Tab. 55. Determination of the C-factor

Surface catchment map	C'-factor			
	0.4	**0.6**	**0.8**	**1**
Area discharging inside karst area	0.6	0.8	1	1
Area discharging out of the karst area	1	1	1	1

2.12.2.3 Final OC vulnerability map

The possibility of flow concentration was excluded in the test area and therefore C-factor was assessed as 1.0. So the final resource vulnerability map is essentially identical to the O-map (Fig. 107.).

2.12.2.4 K-factor

To calculate vulnerability on the basis of K, we did not use the method proposed by ANTIGUEDAD et al., (2000). As for basic information, we have used results of tracer test (ESZTERHÁS et. al., 1998) and flow modelling (KOVÁCS et al., 2001). The dominant flow process in the dolomitic bedrock was supposed to be quasi-diffuse (intergranular) flow. By combining O, C and K factors, we derived sources vulnerability map for Kádárta spring.

2.12.2.5 P-factor

The climate is humid in the area with occasional extreme events. Precipitation events with more than 20 mm precipitation per day occur 4 to 10 times a year. Rainfall events exceeding 1mm/day occur 79 times a year. The rain to snow ratio is 5.9 on average (1991-2000). The average number of snowy days (for the same period) is 29 annually.

On conclusion, we can say that the favourable condition would promote flow concentration only 10 times a year. However, we know that flow concentration is definitely not favoured by the structure of the catchment. So in the Veszprém-Kádárta plateau, the precipitation reaches the aquifer by infiltration and not by concentrated flow. The O-factor provides an efficient protection (mostly loess areas) against contaminants potentially transported into the aquifer by infiltration. It has to be pointed out that at times of maximum precipitation events also the intensity of infiltration will be a maximum.

2.12.2.6 Discussion

Application of the "preliminary European Approach" (pEA) in the Kádárta test site resulted in final resource and source vulnerability maps. By comparing these two maps, it can be seen that the horizontal residence time is much longer than the vertical one. It means that the protective function of the O-factor is more and more important close to the Kádárta spring.

The resource vulnerability map is identical to the O-map, so its contour lines are remarkably similar to those of the isopach map of the unsaturated zone. We think that this must be the result partly of the method itself and partly of the total absence of flow concentration.

For the catchment of the Kádárta spring, the assessed intrinsic vulnerability obtained from the different methods (EPIK, DRASTIC, pEA) was also different. The results attained by using the EPIK method were not satisfactory. The standard DRASTIC method proved to be much more useful. The best approach was with the "preliminary European Approach". All the tested methods are highly subjective. The introduction of physically based validation methods will be inescapable in the future.

2.12.3 Hazard Mapping

2.12.3.1 Overview

The total area of the test site is covered by forest (11%), by grassland and bushes (22%), 59% are used for agriculture and 8% are settlements. The city of Veszprém is almost the only settlement. The test area includes approximately 30% of the total area of the city. Several villages with only a few hundred inhabitants are situated outside of the boundary of the area. The area is crossed by several roads: the national highway No. 8, main roads No. 73 and 82, several smaller roads and a network of cart tracks.

2.12.3.2 Description of hazards

The hazards within the test site were identified according to a field survey within the frame of the National Programme for the Protection of Vulnerable Aquifers and mapped on topographic maps at scale 1:10,000. According to their spatial extension, three types of hazards were distinguished – diffuse (polygons in the GIS), line and point hazards.

Diffuse (polygon) hazards result from urbanisation (leaking sewer pipes and houses detached from the public sewage system) (8.3 % of the total area), gravel pit (0.4 %), and intensive agriculture (58.5 %).

Line hazards include 65.6 km of roads and cart tracks outside the settlements. Of these, 7.8 km are highways with between 10,000 and 15,000 cars per day, 8,9 km main roads with 5,000 to 10,000 cars per day, 2.2 km minor roads with up to 2,500 cars per day and 46.7 km cart tracks. A wastewater pipe is also considered as a line hazard. This pipe is 3.3 km long, running from the treatment plant of a military airport towards the river Séd, which receives the partly treated wastewater.

The most common point hazards in the test site are illegal garbage dumps and manure heaps related to farmhouses. A noodle factory is situated at the boundary of the test site and is considered as hazard because of being detached from the sewage system. The graveyard of the Kádárta village is also situated at the boundary. The wastewater treatment plant of the already mentioned military airport is considered as a point hazard. Finally, a carrion pit (animal burial) is also found at the test site.

2.12.3.3 Determination of the Hazard Indices

The weighting factor H for each hazard was determined according to the values proposed by COST Action 620. This factor represents the harmfulness of hazards to the groundwater.

The ranking factor (Q_n) is used for making comparison between hazards of the same type according to the quantity of the harmful substances. This factor ranges from 0.8 to 1.2

Urbanisations with leaking sewage pipes are usually ranked according to the number of inhabitants of the settlement using a logarithmic scale from <1,000 to >1,000,000 inhabitants. In our case, 30% of the 61,000 inhabitants of the city of Veszprém are supposed to intersect with our test site, i.e. approximately 18,300 inhabitants which yields 0.97 for Q_n.

Roads were ranked considering the number of cars using these. A logarithmic classification ranging from <10 cars per day (Q_n = 0.8) to 15,000 cars per day (Q_n = 1.2) was used for this hazard.

Illegal garbage dumps were rated according to their volume: a high factor (1.2) was applied for several hundreds of m^3, while a medium value (1.0) was applied for tens of m^3 of garbage. For the gravel pit, a medium rate (1.0) was applied according to its moderate spatial extent.

Manure heaps are of medium size (5,000 – 15,000 m^3), therefore the same ranking factor (1.0) was applied for them.

All the agricultural area in the test site was classified as intensive agriculture. As the fertility of the soil is low in this area, a high amount of fertilizers is used in order to yield reasonable crops. Therefore a high factor (1.2) was applied for this kind of hazard.

The wastewater treatment plant and its pipeline were ranked by a medium factor (1.0).

The carrion pit (animal burial) is of small spatial extent (its diameter is about 2 m), so it was ranked by a low factor (0.8). The graveyard of the village Kádárta was also rated low (Q_n = 0.8) since it belongs to a small village.

The reduction factor (R_f) provides an assessment of the probability for a contamination event to occur. For the majority of hazards occurring in the Kádárta test site little information was available about the technical status, maintenance level, security measures, etc, i.e. the main factors for assessing the probability of a real contamination event. Therefore R_f was defined as 1 (no reduction) for these. For those, which are proven to be uncontrolled (e.g. illegal garbage dumps), obviously no reduction factor (R_f= 1) was applied. The noodle factory operates under control, so a reduction factor of 0.6 was estimated for it. The harmfulness of the cart tracks can be reduced because of the lack of the main pollution source, namely the spreading of salt in winter, therefore a value of 0.5 was applied.

Table 56 summarizes the hazards found in the Kádárta test site together with their Hazard Index and Hazard Index Class. The ranking and reduction factors have minor influence on the Hazard Index Class, weights determine the level of hazardousness. It is not surprising that most of the hazards appear to fall to the Hazard Index Class #2 (low hazard level). Three hazards fall to the "Very Low Hazard" category, the graveyard, the noodle factory and the cart tracks, while only one, the wastewater pipeline is categorised to the "Moderate" class.

Tab. 56. Hazard inventory of the Kádárta test site with the Hazard Index values and corresponding Hazard Index Classes.

Hazard	HI	HIC
Urbanisation with leaking sewer pipes and sewer systems (1.1.1)	34	2
Discharge from an inferior treatment plant (1.1.6)	35	2
Garbagedump, rubbish bin, litter bin (1.2.1)	40–48	2
Road, unsecured (1.4.1)	16–48	1–2
Graveyard (1.6.1)	20	1
Animal burial (1.6.2)	28	2
Gravel and sand pit (2.2.2)	30	2
Food industry (2.4.10)	22	1
Waste water piplines (2.7.1)	65	3
Manure heap (3.1.4)	45	2
Intensive agriculture (3.2.4)	36	2

2.12.3.4 Graphical interpretation

The graphical interpretation of the hazard map was done in a GIS, using standardised symbology (*shadesets, linesets, markersets*) (Fig. 109).

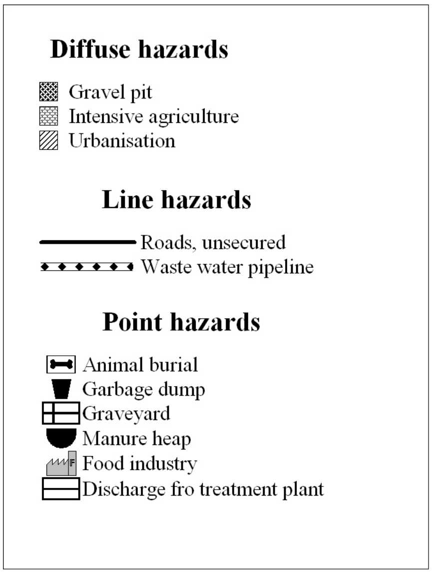

Fig. 109. Symbols representing the hazard in the test site

The unclassified map (Fig. 110) on a topographic base shows all hazards in the test site in red colour, independent of the Hazard Index. The portions of the test area without hazard were not indicated. The classified hazard map (Fig. 111) shows the mapped hazards in different colours according to the Hazard Index Class. Blue shading symbolises areas inside the test site without any diffuse hazards. Areas without shading are outside of the test site.

2.12.3.5 Usefulness of the hazard map for the test site

The hazard map of the Kádárta area was mapped at 1:10,000 scale and printed at 1:50,000 scale. This scale seems to provide sufficient resolution for representing the hazards investigated in the test site, because, for all we know, there are no areas with a high concentration of hazards. However, for decision-making purposes, more detailed maps, e.g. for the urbanised areas, would be necessary.

For a future combination of the Hazard Map with the Vulnerability Map, we suggest use of the Hazard Index values instead of the Hazard Index Class, because classification tends to eliminate the inherent differences existing in the original data. For the sake of combination, the point and line hazards must be transformed to polygons (e.g. making use of buffer zones). However, at first the theoretical basis of selecting a useful extent for the buffer zones or for the grid cells needs to be elaborated.

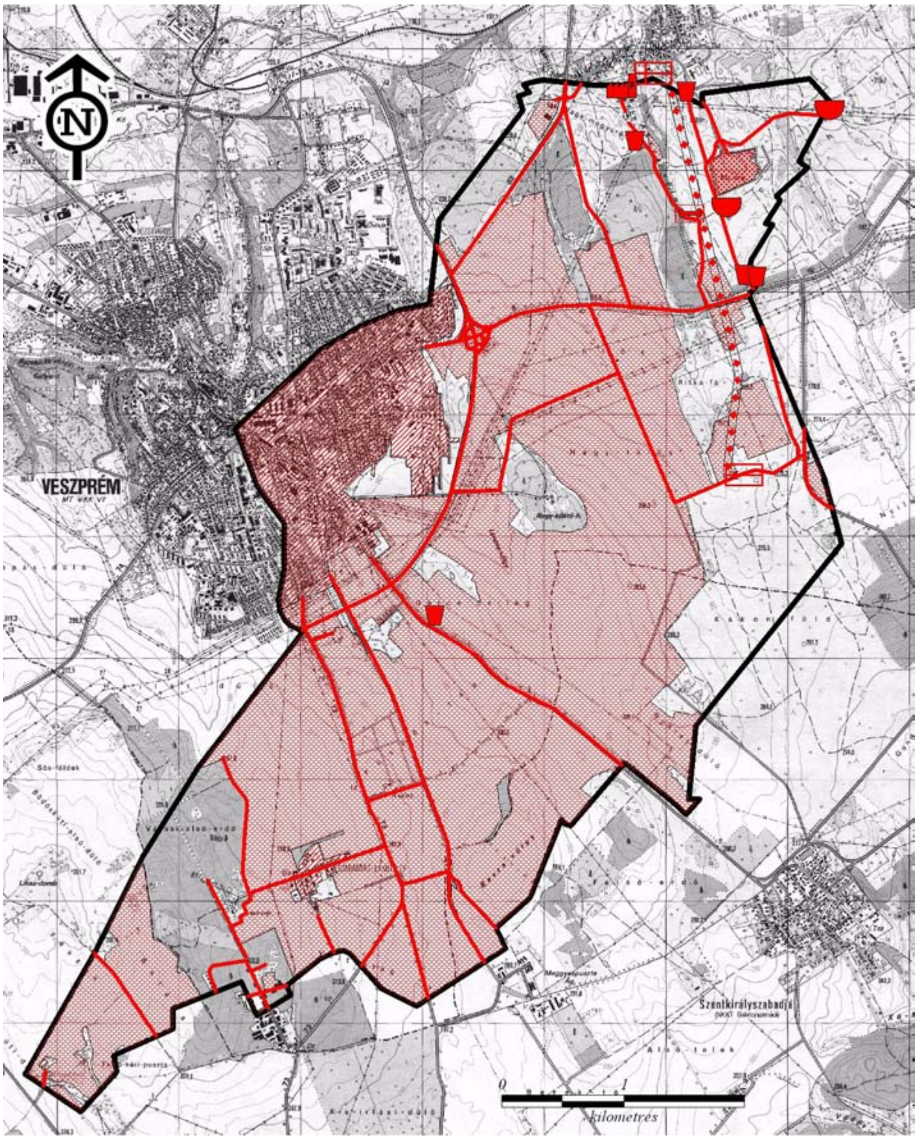

Fig. 110. Unclassified hazard map of the Kádárta test site

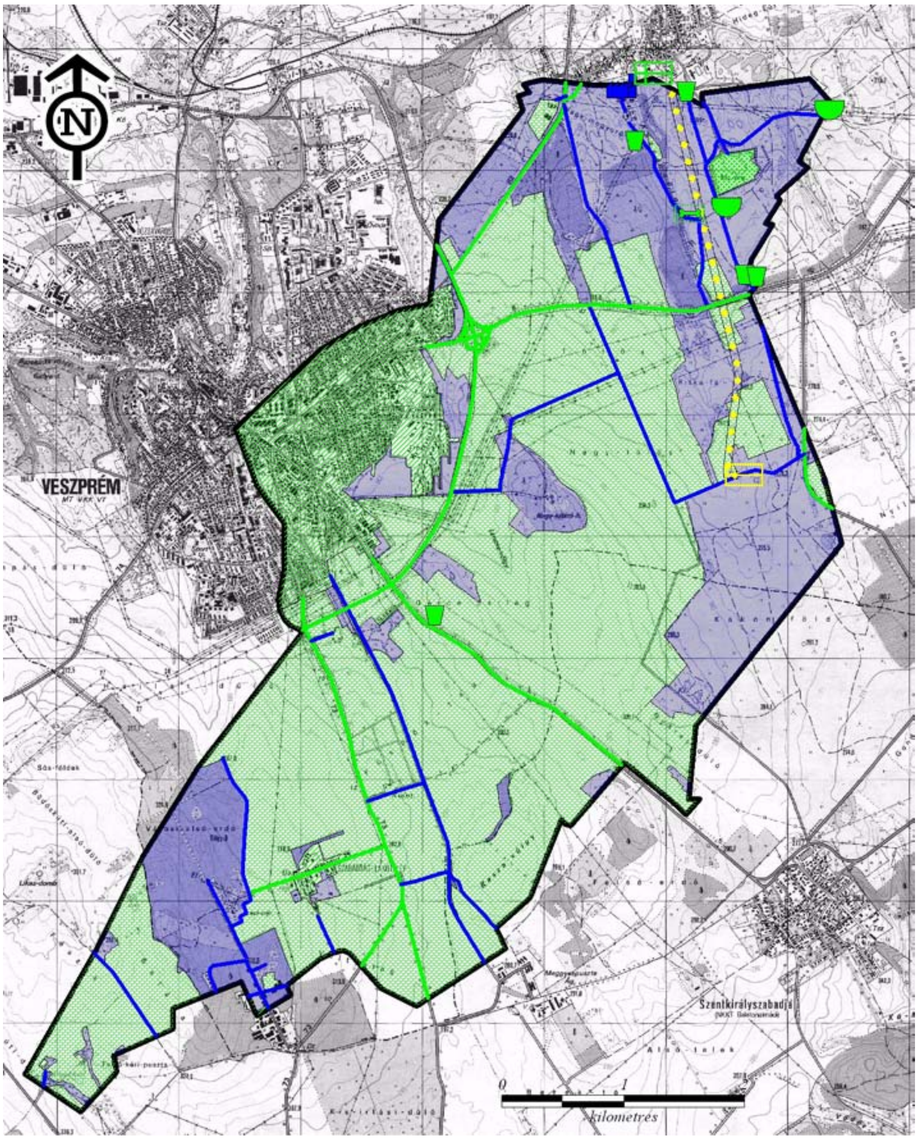

Fig. 111. Classified Hazard map of the Kádárta test site

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Conclusions

Over the five years of activity embraced by our Action we have remained dynamic, actively seeking inputs from other areas of related science. From the first Management Committee Meeting, the delegates of the 15 participating countries decided to involve various invited and regular experts in order to open and reinforce their thinking. As a result more than 20 regular and invited experts routinely participated in our meetings, bring with them their knowledge of many scientific disciplines related to vulnerability mapping, such as; GIS, risk assessment, fate and transport of chemical and biological compounds as well as numerical simulation and remote sensing, to name but few. Their rich contribution rapidly became essential and is greatly reflected in our final report.

Our numerous meetings, in 12 or more countries, have each involved field trips and scientific presentations by numerous national vulnerability specialists. These visits have been extremely useful in both helping delegates understand the different national approaches to vulnerability as well as acting as a conduit for our Action to communicate both potential areas of development and our findings. Our policy of dynamic scientific exchange between participating institutions has moreover; contribute to opening new perspectives for 20 of more young scientists who have benefited from COST's excellent Short Term Scientific Mission program. They were able to discover the different approaches and sensitivities to the subject of vulnerability throughout Europe and were able to learn a lot from their hosts. This will certainly be of great benefit to their careers as well as the future management of the groundwater resources within Europe.

We can consider the development and reinforcement of a European network of institutions actively working on vulnerability, as a significant success for this Action. This network will provide a sound foundation with which to champion the protection of European groundwater resources into the future. Our Action is clearly linked with a lot of new research that has already begun within various disciplines related to vulnerability. In almost all the participating countries, government departments or private partners are promoting; different, new and promising research. In some instances, changes in regulations governing groundwater protection have resulted more or less directly from our discussions and findings.

Regarding our scientific and practical results, we produced on the one hand a general basis and approach to Karst Vulnerability, which enjoys a large consensus; on the other hand we were able to provide promising reflections on the significance of groundwater vulnerability as a protection measure explicitly recommended within the European Water Directive. Our results are already relatively well known within Europe and have been presented in several symposium and conferences within the first semester of 2003. Moreover, they have already been used in countries outside of Europe such as in the Lebanon.

Some specific conclusions related to each Working Group or important chapters are summarised below:

The European Approach to **Intrinsic Vulnerability Assessment** of karst aquifers is certainly very different to the only pre-existing, karst specific approach – the EPIK method developed in Switzerland. Unlike EPIK the output from COST620 is an approach and not a methodology. This has the advantage that it is highly flexible and non-prescriptive, able to be adapted for use in a wide variety of karstic environments and to be used for either source or resource protection. The corresponding disadvantages are that actual data collection and processing methods are not defined and some of the factors (K and P in particular) are only described in general terms. However, the broad nature of the conceptual approach developed by COST620 means that the approach could, if required, be integrated into more comprehensive groundwater and even surface water protection schemes. The K factor is the only wholly karst specific feature in the approach.

In order to convert the European Approach into a robust methodology it will require extensive testing in various karst environments in order to develop appropriate criteria for measuring, quantifying and categorising the factors. The practical application of the ‘C’ and ‘P’ factors would benefit from further research. Finally the development of suitable validation and quality assessment methods are required to allow for objective assessment of vulnerability

The proposed approach to **Specific Vulnerability Assessment** considers both the properties of contaminants and protective layers. A specific attenuation process is becoming effective only if the diffuse flow of the layer meets the favourable conditions in which they occur (a sufficient residence time and an intimate contact with the bedrock or suspended matter, which is not transported out of the system).

The result of the specific European approach depends on:

- The type of contaminant or group of contaminants (the approach can be used for any type of contaminant);
- The type of layer (it can be used in karst areas as well as in any type of aquifer);
- The quality of intrinsic data (this assessment can be used with methodologies developed in the framework of the intrinsic European Approach, but also in combination with any kind of intrinsic vulnerability assessment, insofar as the specific approach is just further correcting the intrinsic assessment).

The outputs of this approach are a specific attenuation map, and, when it is combined with intrinsic data, a specific vulnerability map can be created.

For the moment, only two field applications have been performed: Vaulion (VD, Switzerland) and Sierra de Líbar (Andalusia, Spain), due to the late setting of the final method. This specific European approach or method requires a wider application, and the results must be calibrated and evaluated by independent hydrogeological methods (tracing tests and natural physical, chemical and isotopic tracing).

As the hazards are not regularly scattered in an area (there are places threatened by agricultural hazards, others by industry or infrastructure), the assessment must not be systematic: It is obvious to analyse microbiology, nitrate or atrazine in groundwater during specific vulnerability assessment in farming areas, and not in industrial and urban ones. So the choice of the different specific maps is dictated by the contaminants really expected and analysed in the groundwater of an investigated area.

Regarding **Hazard Mapping,** a significant achievement of COST620 concerns the development of a seamless methodology for the production of hazard maps. The methodology considers the potential sources of groundwater contamination resulting from human activities taking place mainly at the land surface. Distinguishing between infrastructure, agricultural and industrial development activities, the purpose of the proposed hazard inventory is to cover all the various hazards that are considered relevant and to allow, through a reasonable subdivision, the mapping, evaluation and assessment of the hazards in an economically feasible and practical manner. The easy-to-use hazard data collection software developed as a specific activity in COST620 will no doubt serve considerably in this regard. A first, ‘unclassified’ hazard map shows the potential sources of contamination through an appropriate mapping symbol and signature (point, line or polygon) for each type of hazard according to its spatial properties and the mapping scale. The second, ‘classified’ hazard map depicts the potential impact of these hazards on groundwater. Here, the mapping process is driven by a mathematical algorithm allowing for the calculation of the potential degree of harmfulness for each hazard. The algorithm considers both weighting and ranking coefficients as well as a probability term to represent the likelihood of a contamination event. The resulting Hazard Index values are then grouped according to just five possible levels of impact and shown on the map according to the colours of the rainbow. For the majority of the test sites that are documented in this report, hazards proved conspicuous by their absence, and hence these sites did not lend themselves to be the most appropriate case studies to demonstrate the hazard mapping methodology. Nonetheless it is hoped that these provide some further illustration, especially with regard to the calculation of the Hazard Index.

When considering **Risk Mapping**, an evaluation of available studies dealing with groundwater protection, showed that prior investigations and interpretations were mainly restricted to the protection capability and capacity or vice versa of the vulnerability of a catchment. This is especially the case in Central Europe, where by legislation all groundwater is regarded as an important natural resource and therefore requires protection and safety measures without considering the economical value of the individual resource. The increasing demand of land for urbanisation, industrialisation and intensified agricultural land use and the resulting competition on available land request a rethinking of our former concepts and a stronger inclusion of economical aspects in groundwater protection.

Though COST620 concentrates on the original project proposal of vulnerability mapping, it has become evident in the course of the Action that especially detailed vulnerability maps are still more usable by experts than decision makers. However, for practical purposes we need a cost orientated evaluation of the possible groundwater damage, therefore vulnerability estimation has to be supplemented by a full risk assessment including the evaluation of the damage to ecological and economical aspects. In fact the more theoretical approaches and legislative regalement seem to have already been bypassed by very pragmatic and cost-orientated practical decisions. A good example of this is the prioritisation procedure for contaminated groundwater, where remediation is undertaken or given priority to those resources, which have higher economical value.

In chapter 6, valuation of groundwater resources is still done in a simplified way by reference to existing or planned future use of the water, similar to the approaches taken in Ireland and the UK. Future risk assessments will need a more sophisticated approach, where the potential ecological and economical damage is considered with the cost for remediation or alternative water supply or even alternative ecosystems.

In the **Applications,** the proposed COST 620 approach to intrinsic and specific vulnerability mapping, hazard and risk mapping, and validation was applied in twelve test sites in eight European countries. The results show that the proposed approach provides a powerful and comprehensive tool for resource and source protection zoning, sustainable groundwater management and land use planning within the framework of the European Water Directive.

Finally, our collective work clearly demonstrates that a large group of competent people encouraged to work together can both obtain impressive results and mutual enrichment. It has been, for all of us, a great pleasure to work together on this rich adventure and the COST program has to be clearly thanked for providing us with such a wonderful opportunity.

Glossary

Allogenic recharge: Surface runoff that enters a karst system from adjacent non-karst rocks.

Attenuation: The breakdown or lessening of the concentration of a contaminant in groundwater due to a wide range of processes.

Autogenic recharge: water that enters a karst system directly from local precipitation.

Conduits: Solutionally enlarged passageways in karst systems in which most of the karst groundwater may flow. Conduits allow the rapid flow of water and are a major reason why karst groundwater is often highly vulnerable to contamination.

Contamination: Deterioration in the quality of groundwater due to human activities. Note that in general usage the terms contamination and pollution are often used with a range of meanings, which can easily lead to confusion. Because of the number of authors contributing to this publication the single broad term contamination only is used to describe a worsening in water quality resulting from human activities.

Contaminant: Any substance causing contamination.

Developed spring: A spring that is being used as a water supply source. Synonym: harnessed spring.

Diffuse recharge: Infiltration of water through many small entry points so that recharge to an aquifer takes place spatially in a relatively uniform manner.

DIRAC-type solicitation: Momentary/instantaneous pulse-type tracer injection; synonyms: DIRAC pulse, slug injection.

Doline: Small to medium sized surface depression formed by solution and typical of karst areas. Dolines range in size from a few metres to a few hundreds of metres in diameter. Synonym in USA is sinkhole.

Dry valley: A valley in which the waters of the stream/river that originally cut the valley now flow below the present land surface by means of an underground drainage network.

Epikarst: An upper weathered zone in a karst system with relatively high permeability. It can range in thickness from less than two to tens of metres. Not present everywhere in karst areas.

Groundwater resource: All the groundwater in an area that can be used.

Groundwater source: A point at which a water supply is abstracted from an aquifer.

Hazard: An event, process or activity, which has the potential to degrade the quality of the environment, in this case the quality of the groundwater.

Intrinsic vulnerability: The Intrinsic vulnerability of groundwater takes into account the geological and hydrogeological characteristics of an area, but is independent of the nature of the contaminants and the contaminant scenario.

Karst: An area of limestone or other soluble rocks, in which the landforms are mainly the result of dissolution by water and where most drainage is underground in enlarged fractures and conduits. Used as a noun and frequently as an adjective.

Karstic: Pertaining to karst. Less commonly used adjectival form of the word karst.

Karstification: The development of karst landscapes and karst drainage mainly through the process of dissolution by water.

Karst system: An interconnected area of karst rock, which forms a hydrogeological unit.

Non-karst rock: Rock the composition of which is such that chemical solution (dissolution) is not a significant process during weathering.

Origin: Assumed place of release of a contaminant in the context of the origin-pathway – target concept of contamination. Synonym: source (in sense of cause of contamination).

Paleokarst: Karst features formed in the past and no longer part of an active karst system.

Pathway: Everything between origin and target.

Phreatic zone: See saturated zone.

Point recharge: Infiltration of water to an aquifer through a limited number of entry sites so that the recharge is locally concentrated. Is typical of karst areas due to the presence of such features as swallow holes.

Pollution: see Contamination: The term pollution is not used in this report to describe groundwater degradation. This is because of the widespread variations in meaning of the words contamination and pollution as used by different people and thus the danger of misunderstanding.

Polje: A steep-sided, flat-bottomed, closed depression found in many karst districts and ranging in extent from 1 km2 to hundreds of km2. Poljes are usually partly covered by sediments and are liable to seasonal flooding.

Protection zone: Delineated area in which groundwater is protected by restrictions on human activities. Logically the most severe restrictions are applied in those zones close to groundwater sources and in areas of very high vulnerability.

Risk: Probability of occurrence of adverse events multiplied by the consequential damage.

Saturated zone: The zone below the water table in which all spaces are water filled. Synonym: phreatic zone.

Specific vulnerability: The Specific Vulnerability of groundwater takes into account the properties of a particular contaminant or group of contaminants in addition to the intrinsic vulnerability of an area.

Subsoil: The unlithified granular material between the topsoil and the bedrock. Synonyms: overburden, superficial deposits. The term “unconsolidated deposits/sediments” is misleading and should be avoided.

Swallow hole: point where a sinking stream goes underground.

Target: The water that has to be protected. Synonym receptor. For groundwater resource protection the target is the surface of the groundwater in the aquifer; for groundwater source protection the target is the water in the well or spring.

Travel time: The time taken by a contaminant to move from its point of entry to the ground until that contaminant reaches the target.

Topsoil: The biologically active zone of weathering of the uppermost layer of the earth’s crust.

Unsaturated zone: The zone between the land surface and the water table. Synonyms: zone of aeration, vadose zone.

Vadose zone: see unsaturated zone.

Validation: (In the context of vulnerability mapping) Procedures used to ensure the validity of the conceptual understanding of the prevailing hydrogeological conditions.

Verification: (In the context of vulnerability mapping) Procedures used to determine that a vulnerability assessment method produces the correct results.

Vulnerability: The sensitivity of a groundwater system to contamination.

Zone of aeration: see unsaturated zone.

Appendix - Existing Vulnerability Mapping Methods

Classification of Vulnerability Mapping Methods

A variety of approaches to vulnerability mapping exist and a classification of these approaches is given below:

Hydrogeological Complex and Setting Methods

Hydrogeological complex and setting methods take a qualitative approach. The underlying principle of hydrogeological complex and setting methods is that two different areas with similar hydrogeological properties have a similar vulnerability (Vrba & Zaporozec, 1994). They are useful at smaller scales e.g. 1:1,000,000 (Goldscheider, 2002). A description of a number of applications of this method may be found in Magiera (2000).

Index Methods and Analogical Relations

Index methods and analogical relations are based around mathematical descriptions of hydrological and hydrogeological processes that are believed to influence vulnerability (Goldscheider, 2002). The AVI method (Van Stemport et al. 1993) described later in this chapter is an index method; others are described by Magiera (2000). Index methods and analogical relations may consider specific contaminant migration through the groundwater system.

Parametric System Models

Parametric system models include matrix systems; rating systems and point count system models (Gogu & Dassargues, 2000). For each of these methods a variety of parameters are selected and these parameters represent the vulnerability of the study area. Matrix systems are those where the combination of parameters is completed by means of a matrix. One example is the Irish Method (DELG/EPA/GSI, 2001). Rating systems include GOD (Foster, 1987) and AVI method (Van Stemport et al. 1993). In a rating system each parameter is divided into a series of intervals. Associated with each interval is a value and these are either summed or multiplied to give a final vulnerability score.

Point count system models are also known as parametric rating and weighting methods (Gogu & Dassargues, 2000). Similarly to rating systems there is a numerical score associated with each parameter that is divided into intervals. The difference being however that weights are introduced to stress the influence of one parameter over the other(s) in the final vulnerability score. Examples include DRASTIC, EPIK, SINTACS which are briefly described later in this chapter.

Mathematical Models

Mathematical models are seldom used for vulnerability mapping though one has been designed to validate vulnerability map – VULK (Jeannin et al 2001, Goldscheider, 2002).

Statistical Methods

Statistical methods involve the selection of parameters that are deemed to influence vulnerability and the correlation of this parameters with contaminant migration (Goldscheider, 2002). Hence statistical methods are used for specific vulnerability assessments.

Key References:

GOGU R.C. & DASSARGUES A. (2000*)* Current trends and future challenges in groundwater assessment using overlay and index method. *Environmental Geology,* vol.39, no.6, p549-559

GOLDSCHEIDER N. (2002) Hydrogeology and Vulnerability of Karst Systems – Examples from the Northern Alps and Swabian Alb. Dissertation Dept. of Applied Geology of Karlsruhe 236pp; Karlsruhe.

VRBA & ZAPOROZEC (1994) Guidebook on Groundwater Vulnerability Mapping. *International Contributions to Hydrogeology (IAH)* vol.16, 131pp., Hannover.

Existing Vulnerability Mapping Methods

The following section represents a source of data on existing methods. It is not intended to provide detailed methodologies, rather a brief summary of each method with key references. Comment is made, where the method has provided input into the European Approach.

EPIK

EPIK was developed at CHYN in the University of Neuchatel in the early 1990's to address particular risks posed to groundwater quality in the mountainous Alpine Karst of Switzerland. The method uses four parameters that are empirically combined and have been "weighted" by the consensus of experts working in the region. EPIK can only be used on karst aquifers and is typically applied at a scale of 1:10,000.

EPIK utilises four parameters; epikarst (E), protective cover (P), infiltration conditions (I) and karst development. These parameters are combined using a weighting system which is the basis of the final intrinsic vulnerability map on which four (?) vulnerability classes are recognised.

EPIK has been applied in many karst areas of Europe, mostly as a result of work carried out by COST 620. EPIK is thought to have been the first method that included "flow concentration" as a parameter. The method is based on a clearly defined conceptual model of a karst aquifer.

DOERFLIGER, N. & ZWAHLEN, F. (1998): Practical Guide, Groundwater Vulnerability Mapping in Karstic Regions (EPIK). - *Swiss Agency for theEnvironment, Forests and Landscape (SAEFL): 56 p.; Bern*.

DOERFLIGER N., JEANNIN P.-Y. & ZWAHLEN F. (1999): Water vulnerability assessment in karst environments: a new method of defining protection areasusing a multi-attribute approach and GIS tools (EPIK method). - *Environmental Geology 39(2)*, pp. 165-176.

Irish Method

The Geological Survey of Ireland developed the Irish method with input from the wider hydrogeological community. Vulnerability mapping is an essential component of the national methodology for the production of groundwater protection schemes along with aquifer maps and source protection areas. It is used to influence land-use planning decisions for developments such as landfills, on-site wastewater treatment systems and land spreading of organic wastes. About 50% of the country (~30, 000 km^2) has been or is currently being mapped.

The Irish method includes the evaluation of the hydrogeological properties of protective layers overlying groundwater as well as potential bypassing of these layers by karst features. The method is used for all hydrogeological settings in Ireland. For bedrock aquifers, the relevant geological layer is taken to be the subsoil (topsoil is not directly included, although it is included in the groundwater protection responses for particular developments and in permeability mapping). Unsaturated bedrock is not taken into account, as it is not considered to provide significant protection in Ireland owing to its fissured nature. The relevant hydrogeological characteristics of the protecting subsoil layer are the permeability and thickness. The range of permeabilities seen in Irish sub-soils is compartmentalised into three classes – high, moderate and low. These are combined with the thickness of subsoil and the presence of karst features to give four categories of vulnerability. While the method started as a largely qualitative method, subsequent data collection and modelling work has lead to its evolution into a largely quantitative method.

DALY, D. AND DREW, D.P., 1999. Irish methodologies for karst aquifer protection. In: Beck, B.F., Pettit, A.J. and Herring, J.G. (Editors), *Proceedings of the Seventh Multidisciplinary Conference on Sinkholes and the Engineering and Environmental Impacts of Karst*, Harrisburg-Hershey, Pennsylvania, p267-272.

DoELG/EPA/GSI, 1999a. *Groundwater protection schemes.* Department of Environment and Local Government, Environmental Protection Agency and Geological Survey of Ireland. 24pp.

DRASTIC

DRASTIC is a weighting and rating system developed by the United States Geological Survey in the mid 1980's. The method utilises seven parameters: depth to groundwater table (D), net recharge (R), aquifer media (A), soil media (S), topography (T), impact of the vadose zone media (I), hydraulic conductivity of the aquifer (C). The depth to water table and vadose zone media are given the highest weighting. DRASTIC would seem to be best suited to regional assessments (1:50 000 and 1:100 000 scales) and has been applied in a large number of countries worldwide.

ALLER J.R., BENNET T., FEHEER J.H., PETTY R.J., HACKETT G. (1987) DRASTIC: a standardised system for evaluating groundwater pollution potential using hydrogeological settings. S EPA, Ada, OK, EPA/600/2-87-035.

German Method

The German State Geological Surveys (GLA) and the Federal Institute of Geosciences and Natural Resources (BGR) established a method for assessing the protective function of the layers above the groundwater surface. This method does not address the saturated zone within an aquifer.

The method puts considerable emphasis on travel time as a measure of the effectiveness of overlying layers to protect underlying groundwater. Thus, the protective function is dependent on the main factors controlling the travel time: the thickness of each stratum and the properties of the material. The protective cover includes all strata between the ground surface and the piezometric surface, this includes; topsoil, subsoil and unsaturated bedrock, both karstic and non-karstic.

The protective function of the topsoil is assessed according to its effective field capacity (eFC), that of the subsoil by considering grain-size distribution (GSD), and that of the unsaturated bedrock is assigned according to lithology (as well as the degree of fracturing and karstification where relevant).

The method does not consider the special properties of karst, such as flow concentration, in any depth. The basic assumption being that infiltration occurs diffusely and that infiltrating water slowly percolates vertically through the unsaturated zone towards the groundwater table. The method was translated into English by VON HOYER & SÖFNER (1998).

HÖLTING, B., HAERTLE, T., HOHBERGER, K.-H., NACHTIGALL, K. H., VILLINGER, E., WEINZIERL, W. & WROBEL, J.-P. (1995): Konzept zur Ermittlung der Schutzfunktion der Grundwasserüberdeckung. – Geol. Jb., C63: 5-24; Hannover.

VON HOYER, M. & SÖFNER, B. (1998): Groundwater vulnerability mapping in carbonate (karst) areas of Germany. – BGR Hannover, Archive Nr. 117 854, unpubl. report for EC-project COST Action 620; 38 p.

GOD

This method considers three parameters;

- The nature of groundwater occurrence (ranging from none to unconfined).
- Overall Aquifer Class which considers the degree of consolidation and lithological character.
- Depth to groundwater which takes into consideration the confined or unconfined nature of the aquifer.

A numerical value is attached to each parameter division and the three values are then multiplied together to form an aquifer vulnerability index (Gogu & Dassargues, 2000). This method includes the need to assess potential risks to the aquifer and as a result may be said to approach the rigour of a general risk assessment.

FOSTER S.S.D. (1987) Fundamental Concepts in Aquifer Vulnerability, Pollution Risk and Protection Strategy. Vulnerability of soil and groundwater to pollutants ed. *Proceedings and information committee for hydrological research,* TNO, p.69-86.

VRBA J. & ZAPOROZEC A. (1994) Guidebook on Mapping Groundwater Vulnerability. International Contributions to Hydrology, IAH. vol.16, 131pp., Hannover.

AVI

The AVI method uses two variables to formulate a vulnerability index, these being; the thickness of each sedimentary layer above the uppermost saturated aquifer (d) and the estimated hydraulic conductivity of each of these layers (k). The "hydraulic resistance" (c) of each layer is then calculated as the quotient of thickness and conductivity (c = d/h).

The total hydraulic resistance for several sedimentary layers is established by summing the values for each layer. The hydraulic resistance describes the resistance of the layers to vertical flow (lateral spread of contaminants is taken to be insignificant). The hydraulic resistance has an aspect of time, as it indicates the approximate time of travel for water to move by advection downward through the sedimentary layers under a hydraulic gradient of one. The AVI index does not however take into account other factors, e.g. climate, hydraulic gradient, porosity, diffusion etc. It is perhaps most suitable at a large regional scale.

VAN STEMPORT D., EWERT L., WASSENAAR L. (1993) Aquifer vulnerability index. A GIS-compatible method for groundwater vulnerability mapping. *Canadian Water Resources Jrnl*, Vol.18, no.1, p.25-37.

ISIS Method

ISIS is a weighting and rating method that assesses eight parameters; annual mean net recharge, topography, soil type, soil thickness, lithology of the unsaturated zone, aquifer medium and aquifer thickness. This system consists of three weights based around landuse; normal conditions, strong contaminated agricultural area and strong superficial drained area. The final index is divided into six classes of vulnerability.

CIVITA M & DE REGIBUS C (1995) Sperimentazione di alcune metodologie per la valutazione della vulnerabilità degli aquiferi, *Quaderni di Geologia Applicata, Pitagora* Ed. Bologna, 3: 63 – 71

SINTACS

The SINTACS method is similar to DRASTIC. The method utilises the same parameters, but has four different weighting systems depending on the hydrogeological setting. The weighting system has been designed to illustrate the relative importance of the parameters within different settings, which are known as Normal, Severe, Seepage, Karst and Fissured. Normal and Severe reflect the density of human settlement and the intensity of landuse. Other factors are also considered e.g. irrigation. Seepage is used to describe areas that are frequently flooded or swampy i.e. where there is seepage from the surface network to the groundwater.

Karst setting is used for areas that are deeply karstified and fissured or areas of limited surface karst features.

CIVITA M. & DE MAIO M. (1998) Mapping Groundwater Vulnerability by the Point Count System. Chapter 11 in *Managing Hydro-Geological Disasters in a Vulnerable Environment*, ed. K. Andah Publ. Grifo Publishers.

M. CIVITA & M. DE MAIO (2000) SINTACS R5, a new parametric system for the assessment and automating mapping of groundwater vulnerability to contamination - Pitagora Editor (Bologna)

REKS Method

REKS is a method developed in Slovakia and represents a derivation of EPIK. Direct application of EPIK failed in Slovakia due to the need to account for extensive outcrops of non-karstic rock present. The K and I factors of EPIK were modified and amalgamated into a K factor that is specific to this method. The R factor distinguishes between karstic and non-karstic rock environments.

REKS is a rating system using four parameters: rocks (R), epikarst (E), karstification (K), soil cover (S). It is in the early stages of development and there are no published pre-defined ranges of vulnerability classes as yet.

MALIK, P., & SVASTA, J., (1999): *REKS* – An alternative method of karst groundwater vulnerability estimation. Hydrogeology and Land Use Management. *Proceedings of the XXIX Congress of the International Association of Hydrogeologists*, Bratislava, September 1999, pp. 79–85.

Austrian Approach

The Austrian Approach is a weighting and rating system, which was developed for high Alpine karst regions. It includes three factors; input (P), infiltration (I), and exfiltration (K). The approach is partly based on the PI-method for the characterisation of the unsaturated zone, which is included in the factor I. Beside the steady state conditions of an area the system dynamics are described by hydrogeochemistry and isotope-hydrology. A validation factor K is established for quality assurance and it includes the calculation of the storage capacity of the whole system and the mean residence time of water.

CICHOCKI, G., ZOJER, H., ZOJER, HT. 2001: Karstwasserschutz und Vulnerabilität. Entwicklung eines Modells in den Karnischen Alpen - *Mitteilungen IAG BOKU*. Institut für Angewandte Geologie. Universität für Bodenkultur Wien, 2001.