Groundwater and Ecosystems

Edited by

Alper Baba, Ken W.F. Howard
and Orhan Gunduz

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Groundwater and Ecosystems

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PREFACE

Groundwater's global role as a vital source of fresh drinking water is well documented, and efforts are underway in many parts of the world to manage groundwater reserves responsibly and sustainably. Less well understood and frequently neglected, however, are natural ecohydrological systems that are supported by groundwater as it emerges from the subsurface to enter wetlands, streams, lakes and coastal estuaries. These systems – Groundwater Dependent Ecosystems (GDEs) frequently exhibit rich biological diversity and can provide enormous economic wealth. A study published in Nature (1997) valued the global value of wetland ecosystems alone at US$ 14.9 trillion.

In recent years, GDEs in many industrialized countries have shown signs of serious degradation, primarily the result of groundwater abstraction and pollution. Many such systems, including a number of well documented cases in Eastern Europe, are no longer sustainable. As a consequence, the conservation and sustainable management of GDEs has emerged as one of the most urgent environmental research priorities of our time. In 2003, the International Association of Hydrogeologists (IAH) established a Commission focusing on issues related to GDEs; in 2005, Council of IAH approved a proposal from the Portuguese Chapter of IAH to host the XXXV IAH Congress with 'Groundwater and Ecosystems: Interdependencies' as the primary theme. This will be held in Lisbon in late 2007.

Much can be achieved prior to the 2007 Congress, and the NATO ARW recently held at the Canakkale Onsekiz Mart University in Turkey under the auspices of the NATO Security Through Science Programme provided a valuable opportunity for specialists in key related fields to make some important progress and establish strategies and priorities for much needed interdisciplinary work. GDEs lie at the interface of biology with geology, hydrogeology and geochemistry and the challenge is to bring these fields together in a synergistic and productive way.

A large percentage of the world's population lives in cities and either depends on, or is affected in some way, by groundwater. Moreover, groundwater has become a very important and complex issue that attracts the interest of many diverse stakeholders. Many problems related to groundwater and ecosystems are shared by countries throughout the world and there is growing recognition that much can be gained by co-operation on an international scale. This is no time to be complacent and it is critical that key problems be identified, that the potential consequences of these problems be understood, and that the development of solutions begins urgently. Important data gaps must be recognized and filled without delay.

NATO Advanced Research Workshops (ARWs) are advanced-level meetings, focusing on special subjects of current interest. They provide ideal fora for pursuing multi-disciplinary issues of urgent strategic concern. The ARW on Groundwater and Ecosystems held in Canakkale, Turkey comprised 20 papers which addressed a broad variety of issues. Subject matter ranged from reviews and case studies to specialized scientific papers. Major groundwater and ecosystem issues identified and examined at the workshop included:

- Role of groundwater in wetlands
- Other ecosystems dependent on groundwater
- Interactions between groundwater and surface water
- Coastal areas, including saline water intrusion
- Problems related to water and agriculture, urbanization, etc.
- Groundwater and ecosystems in the context of climate variability and climate change
- Groundwater in the hyporheic zone
- Aquifer vulnerability
- Groundwater fluxes, quality and exploitation
- Groundwater protection
- Groundwater management
- System modelling

In particular, the workshop provided participants who work as aquatic scientists in various parts of the world to share findings and discuss their experiences related to the interaction between groundwater and ecosystems. Broad questions posed and debated during the workshop included:

- What is the role and importance of groundwater on the ecosystems?
- How can the economic value of groundwater in the protection and management of ecosystems be assessed?
- How can the effects of groundwater pumping and pollution on ecosystems be evaluated from an economic perspective?
- How and to what extent can ecosystems be protected, managed and, where appropriate, restored?

Groundwater models that focus on ecosystems which are sensitive to changes in the groundwater flow system from both a qualitative and quantitative standpoint were recognized as key tools in developing an understanding of the issues surrounding GDEs. While potential solutions were identified to many of the problems, inadequate baseline data, lack of funding and the absence or low profile of hydrogeologists, environmental geologists and ecologists in institutions were recognized as major impediments to future progress. The workshop, with its focus on both countries in transition and the traditional NATO countries, stimulated considerable discussion and proved effective in initiating exchange of information and strengthening of co-operation amongst experts from NATO, Partner and Mediterranean Dialogue countries.

In conclusion, good progress was made and the proceedings contained here provide an excellent foundation for future work. Much remains to be done, but the commitment shown by workshop participants sends a very positive and encouraging message. Ultimately, the organizers hope that the workshop will have contributed to improving groundwater and ecosystems in the regions addressed and thereby lead to increased security and quality of life.

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REMOTE SENSING TECHNIQUES TO MONITORING COASTAL PLAIN AREAS SUFFERING FROM SALT WATER INTRUSION AND DETECTION OF FRESH WATER DISCHARGE IN COASTAL, KARSTIC AREAS: CASE STUDIES FROM GREECE

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Abstract: This study aims to present remote sensing applications in monitoring coastal areas. It briefly describes the use of up-to-day remote sensing technology, applied to geosciences for multitemporal and multispectral monitoring of the environment, from geological point of view. It also gives emphasis: i) to detection and delineation of areas in coastal depositional landforms (plains) suffering from salt water encroanchment (intrusion), usually resulted from water overpumping, by the use of multitemporal satellite images where the geobotanical anomalies are shown (salt-tolerant plants, or halophytes, as plant indicators), and b) to delineation of water discharge zones along the coastal erosional rocky, mainly karstic, areas by the use of satellite thermal infrared images. The study also focuses on salt water intrusion in coastal areas where it becomes a hydrogeological issue which causes problems to cultivation. One other hydrogeological issue, is the discharge of significant unused quantities of ground water, in coastal areas. In our Laboratory, efforts have taken place to detect and delineate these coastal areas, with the help of LANDSAT-5/TM satellite images (30 m. resolution). With certain digital processing techniques of the above images, areas which are "suffering" from salinity are located in the coastal areas of Pieria prefecture (Macedonia Province, Greece). Furthermore, with the help of the TM6 band (thermal

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infrared), of LANDSAT-5 satellite, fresh water springs were also detected in the coastal areas, of the Gulf of Itea (Central Greece). The above findings can help hydrogeologists to locate areas suffering from salt water intrusion and coastal areas where ground water discharge takes place.

Keywords: remote sensing; Landsat Satellites; coastal salt water intrusion; submarine fresh-water springs

1. Introduction

Remote sensing is the science of deriving information about an object without actually coming into contact with it. Remote Sensing has practically come to imply data acquisition of electromagnetic radiation (commonly between the 0.4 μm and 30 cm wavelength range) from sensors flying on aerial or space platforms, and its interpretation for deciphering ground object characteristics.

The remote sensing technology for observing, measuring and monitoring the Earth resources (Earth Resources Satellites), started systematically in the early 70's. During this period, the US government (NASA) initiated and implemented the so-called LANDSAT program, which is still in operation today. It has been operating using a series of Earth Observation (EO) satellite systems. For the first time, these automatic-operating satellites provided a constant and complex information flow from space.

The success of the LANDSAT program has spawned many similar earth resources satellites by several other nations as well as private industries. Presently, more than 25 earth observation systems are providing data on a routine basis for operational applications in various fields, e.g. cartography (map updating, topographic, geological and thematic base mapping), land cover/use assessment, and monitoring environmental conditions on land and at sea. Different orbit configurations are used, and satellite sensors can view the Earth in vertical, side or stereo modes.

Compared to ground observations, remote sensed satellite data show important advantages. Satellite images provide a synoptic and repetitive overview of the Earth's surface. In addition, the near global, repetitive collection of the data using satellite sensors is cheaper than collecting the same type and quantity of information using conventional methods, e.g. ground survey, aerial photography.

The information content of the space borne imagery is limited by the data characteristics in terms of spectral, temporal and spatial resolution. Spectral resolution stands for the data recorded simultaneously and separately in several portions of the electromagnetic spectrum utilizing atmospheric windows. Temporal resolution stands for the repetition rate. Spatial resolution describes the smallest unit to be identifiable on an image. The spatial resolution is described as Picture Element (Pixel). It may range from 0.6 m (very high resolution data) as in the QuickBird satellite to 1 km or several kilometres per pixel (very low resolution) as in the meteorological satellites (Buchroithner, 1999).

LANDSAT program (USA/NASA) started out with 80 m medium resolution systems (multispectral/MSS mode) of LANDSAT 1-3 satellites. The program continues with 30 m medium resolution system (multispectral/TM mode) of LANDSAT 4-5 satellites. In 1999, LANDSAT-7 was launched, carrying the ETM + scanner (system), with multispectral mode of seven bands with 30 m resolution (excluding thermal band with 60 m resolution) and one band in Panchromatic/PAN mode, providing high resolution data of 15 m (Figure 1). Also, in 1999, the TERRA satellite was launched by NASA, carrying various multispectral scanners. These scanners provide data in 15 m, 30 m and 90 m resolution (medium to high resolution satellite data), in the visible, short-wave infrared and thermal infrared spectrum, respectively.

The French SPOT program (SPOT 1-4) started out in 1986 with 20 m resolution systems in multispectral mode and 10 m in PAN mode (high resolution data). In 2002, SPOT-5 was launched, carrying multispectral mode of 20 m and 10 m resolution and PAN mode with 5 m resolution (high resolution data).

The Indian Remote Sensing (IRS) system started out in 1988 with 36.5 m resolution multispectral data of IRS-1A and IRS-1B satellites. In 1995, IRS-1C was launched and provided the highest (5, 8 m) spatial resolution data, commercially available until 1999, when the US IKONOS satellite was launched, providing users with very high resolution data of less than 5.8 m.

The last 3-4 years, in addition to IKONOS systems which gives multispectral images of 4 m resolution and PANS images of 1 m resolution, the QuickBird systems (2001) gives multispectral images of 2.5 m resolution and PAN images of 0.6 m resolution. Also, other satellite systems from Japan, Russia and other countries were launched, providing users with multispectral and PAN data of various resolutions. In Table 1, the satellite data resolution and mapping scales are shown (Buchroithner, 1999). For more details about main current operational satellite systems and satellite data-mapping scales, see the book edited by Herbert Kramer, 2002.

2. Purpose of This Study

The main purpose of this paper is to show two examples of reconnaissance surveys which have taken place in Greece, by the use of remote sensing techniques, in order to:

a) detect and delineate areas in coastal depositional landforms (plains) suffering from salt water encroanchment (intrusion), usually resulted from water overpumping, by the use of multitemporal satellite images (see first study area).

b) delineate water discharge zones along the coastal erosional rocky, mainly karstic, areas by the use of satellite thermal infrared images (see second study area).

The salt water intrusion in coastal areas is a hydrogeological issue which causes problems to cultivation. Another hydrogeological issue is the discharge of significant unused quantities of ground water, in coastal areas. In our Laboratory, efforts have taken place to detect and delineate these coastal areas, with the help of available LANDSAT-5/TM satellite images (30 m. resolution).

With certain digital processing techniques of the above images, areas which are "suffering" from salinity, are located in the coastal areas of Pieria prefecture (Macedonia Province, Greece). Furthermore, with the help of the TM6 band (thermal infrared) of the LANDSAT-5 satellite, fresh water springs were also detected in the coastal areas of the Gulf of Itea (Central Greece). The above findings can help hydrogeologists to locate areas suffering from salt water intrusion and coastal areas where ground water discharge takes place.

3. First Study Area: Pieria Salt-Affected Areas

3.1. ENVIRONMENT OF THE STUDY AREA

The study area is a coastal, almost flat region, consisting mainly of clays, clay-sands and sandy loams. The color of these disposals is black, due to the organic material contained in these sediments, derived from marshy plants (Figure 1).

3.2. MATERIALS

The following data were used for this study:

- Topographic map of the Hellenic Army Geographical Service (HAGS), of 1:50.000 scale.

- Geological map of the Institute of Geological and Mineral Exploration (IGME), of 1:50.000 scale, and map of IGME showing salt water intrusion zones in Korinos coastal area (Tzimourtas, 1999).
- Thematic Mapper (TM) images of the LANDSAT-5 satellite, recorded on 10-04-1986 and 21-08-1999.

All TM bands (images) of LANDSAT-5 were used (resolution of 30 m), except the thermal infrared band (band 6, resolution 120 m). The digital processing of the satellite images was carried out using the EASI/PACE digital image processing software.

3.3. METHODOLOGY

As it is shown in Figure 2 (IGME, 2000), in Pieria coastal area, the salinity occurrence to the coastal water table aquifers is intense, mostly due to water over-pumping for field-irrigation reasons.

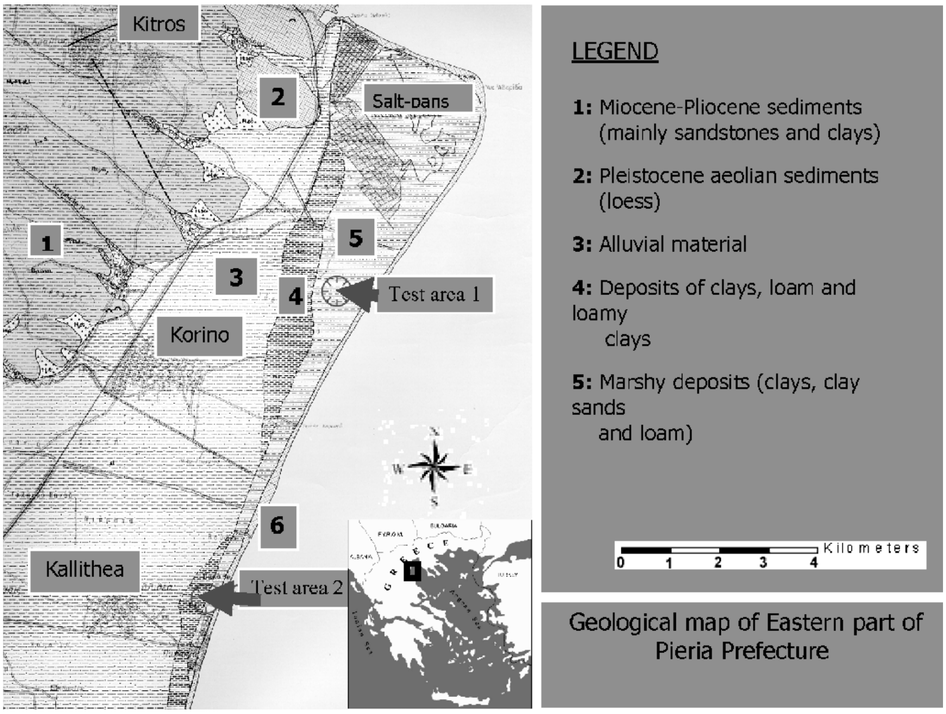

Figure 1. Geological map of eastern part of Pieria Prefecture.

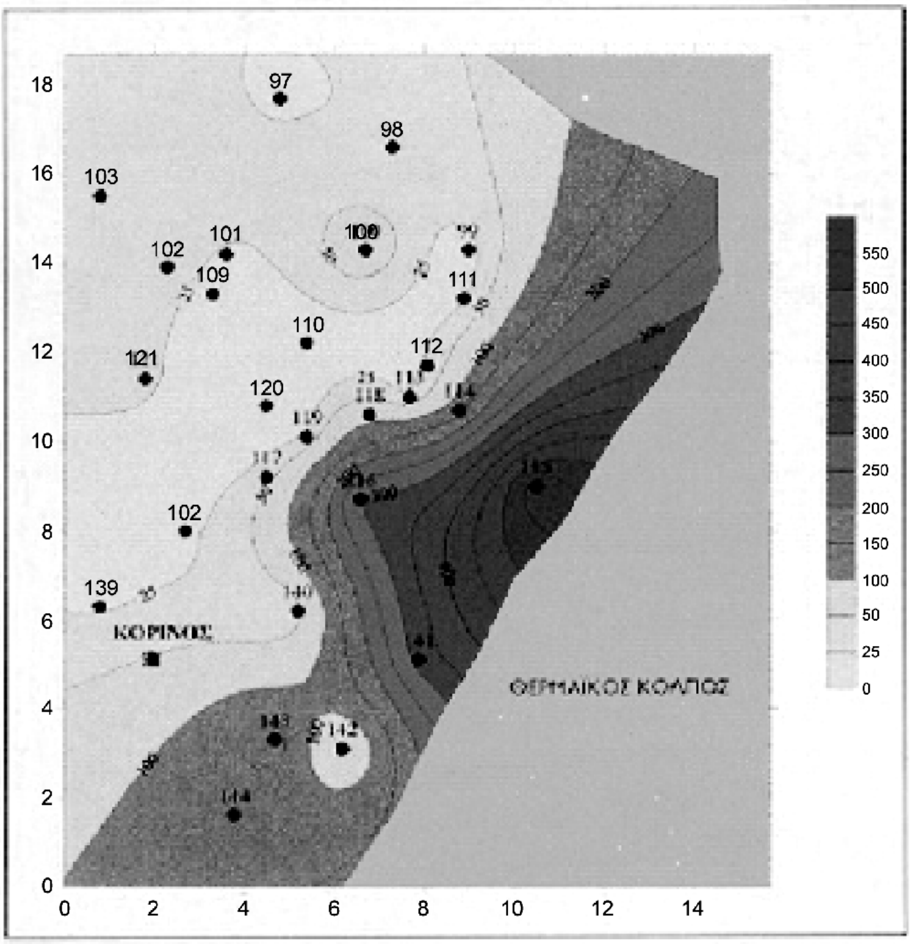

Figure 2. Salt water intrusion zones in Korinos coastal area (Tzimourtas, 2000).

The spectral signature (reflectance) of the stressed and salt–tolerant vegetation is different from the spectral signature of the healthy, cultivated vegetation in the adjacent, not affected by salinity, areas. This happens because the stressed vegetation is weak, or even inexistent, due to the salinity content of the ground.

For the location of the above areas, affected by salinity, various image enhancement techniques of the TM imagery were used. The best results for visual analysis and interpretation were produced by the Principal Component Analysis (PCA) methodology. With the help of PCA methodology, the "volume" of multispectral data of TM is reduced, without loosing any of the initial information of the image. The principal component transformation was applied to the spectral zones of visual (TM1, TM2 and TM3) and reflected infrared (TM4, TM5 and TM7). The first three PC images (PC1, PC2 και PC3), contain more than 99% of the information included in the initial six spectral bands. The spectral differences among various surface materials are better distinguished in the PC images than in the initial TM images. The first principal component, PC1, stresses the topographic features which are strongly correlated in the initial six bands (Sabins, 1997). The second principal component, PC2 (Figures 3 and 4), stresses the differences between visible and infrared spectral bands, and serves to enhance any spectral differences between those parts of the spectrum (Canas and Barnett, 1985; Astaras and Soulakelis, 1992; Sabins, 1997).

In our study, the PC2 image distinguishes the salt-affected areas (test areas 1 and 2) from the adjacent healthy-vegetated areas. That is, the grey-tone values, texture and pattern are significantly different between the above two areas (Figures 4, 5, 6 and 7). Before the visual interpretation, the two PC2 images (Figures 3 and 4), were enhanced via "square root" algorithm of EASI/PACE software, which causes higher contrast in the lower pixel values of the images.

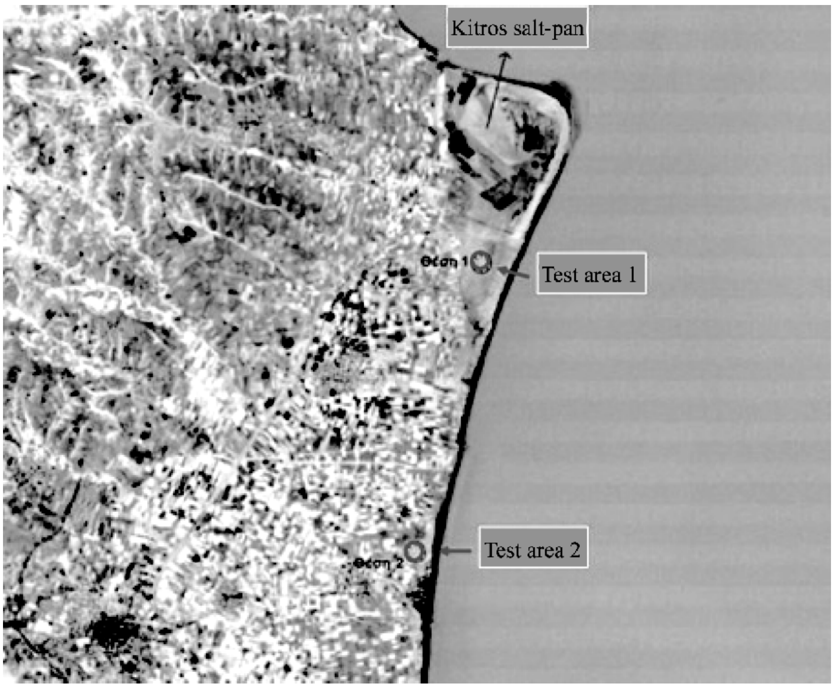

Figure 3. Digitally processed PC2 image of the LANDSAT-5 satellite (acquisition date 10/04/1986) covering the Pieria study area. In sample areas 1 and 2 (O), certain grey spectral signatures (tone and texture) of the salt-sensitive vegetation are shown.

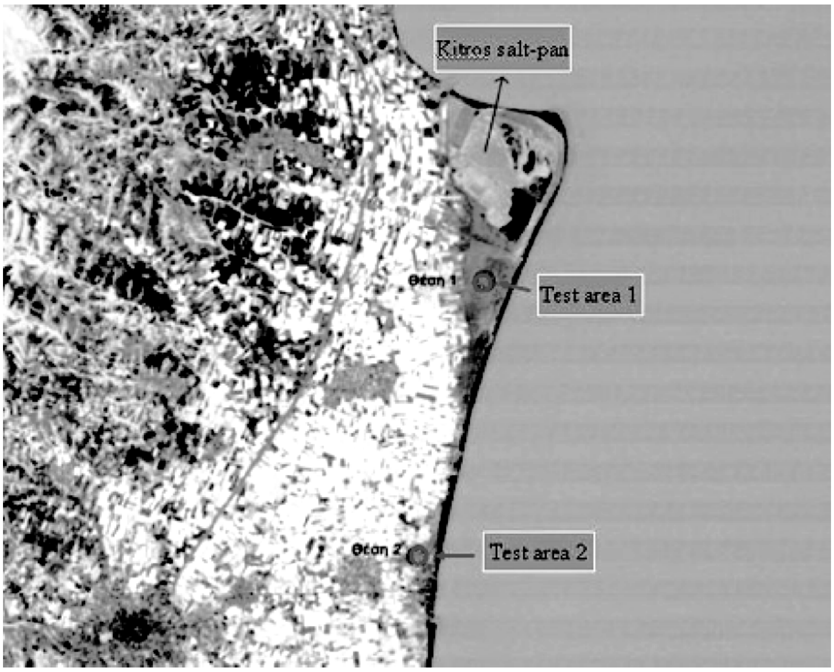

Figure 4. Processed PC2 image of the LANDSAT-5 satellite (acquisition date 21/08/1999) covering the Pieria study area. In sample areas 1 and 2 (O) certain grey spectral signatures (tone and texture) of the salt-sensitive vegetation are shown.

Figure 5. Abandoned-uncultivated field, occupied by salt-tolerant vegetation, in test area no. 1 (see Figures 3 and 4), due to underground salt water intrusion.

4. Second Study Area: Itea Submarine Fresh-Water Springs

4.1. ENVIRONMENT OF THE STUDY AREA

The study area is the Gulf of Itea, surrounded by hills, underlain by limestones and conglomerates of limestone origin (Figure 6).

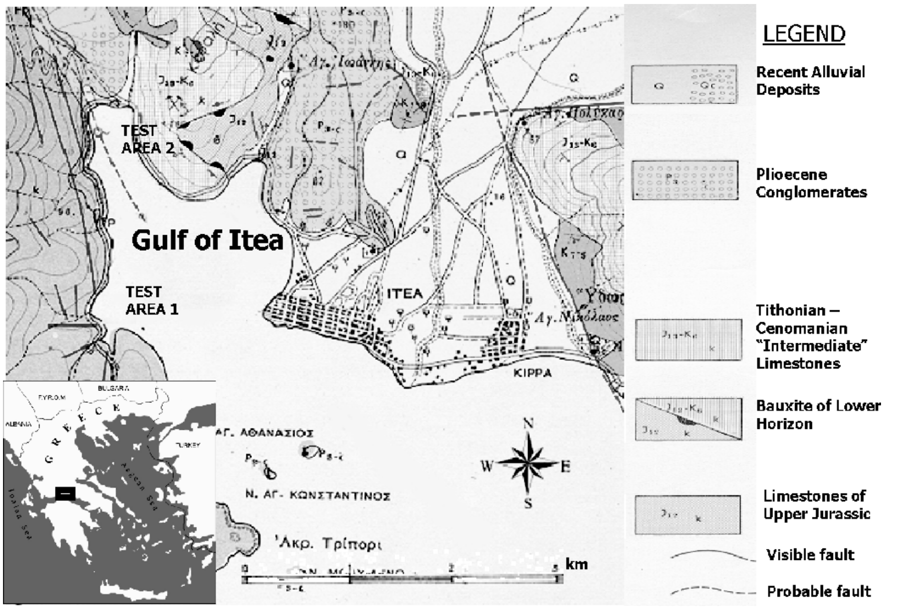

Figure 6. Geological map of the Gulf of Itea and surroundings, on which, test areas 1 and 2 are shown (O~).

4.2. MATERIALS

The following data were used in this study:

- Topographic map of the Hellenic Army Geographical Service (HAGS), of 1:50.000 scale.
- Geological map of Institute of Geological and Mineral Exploration (IGME), of 1:50.000 scale.
- Thematic Mapper (TM) image from the LANDSAT-5 satellite, recorded on 22-05-1986.

During the digital image processing, the TM 6 thermal infrared band was processed more, eventhough its' lower spatial resolution (120 m) than the rest

TM bands (30 m), because the thermal radiation in the spectral range of 10.4-12.5 μm is recorded only by TM6 sensor of LANDSAT-5 satellite. The digital processing of the satellite images was carried out using the EASI/PACE digital image processing software.

4.3. METHODOLOGY

The coastal area of the Gulf of Itea is mainly covered by limestones (Figure 6) with secondary porosity, created by tectonics and dissolve of limestones along tectonic discontinuities. Therefore, the sub-surface water on the land, which is freshwater, moves down the gradient and eventually becomes lost in the sea in the form of submarine springs (Figure 7).

Sea water has a relatively higher density owing to higher total dissolved solids. Due to the density contrast, the freshwater rises and spreads on the sea surface, forming a plume, as the mixing process of the freshwater with the sea goes on concurrently (Astaras, 2001).

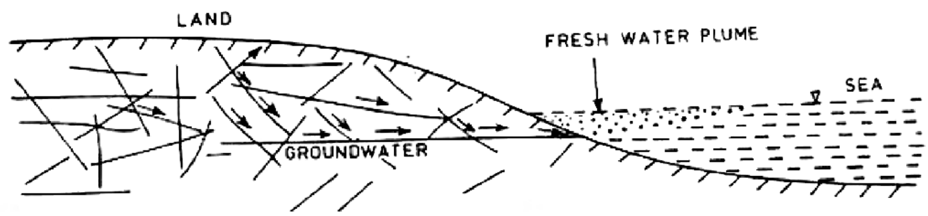

Figure 7. Representation of ground water discharge in coastal areas (Gupta, 1991).

For the detection of these sea-surface freshwater plumes, which show different (lower) temperature than the sea-water, the thermal band of TM6 was used, enhanced by the use of "equal" algorithm of EASI/PACE software. In the resulting image (Figure 8), the grey-tone values of the sea-surface temperature, are distributed between 0 (black/cool) and 255 (white/warm).

From the digitally processed TM6 image, the zones of thermal anomalies on the sea-surface were located, since the fresh-water plumes show lower temperatures than their adjacent sea-surface. On the TM6 image, the fresh-water plumes show dark grey-tone values, as they are shown in test area 1 (Figures 8, 9) and 2 (Figures 8, 10).

The above fresh-water plumes coincide with the extension of the known tectonic faults and lineaments, developed on the limestone rocks, mainly karstic, as it is shown on the geological map (Figure 6).

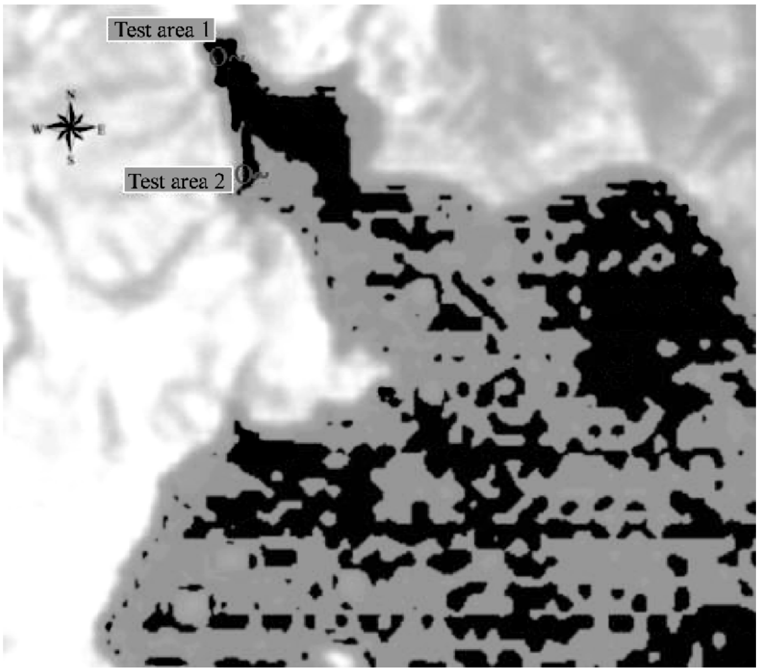

Figure 8. Digitally processed TM6 band (thermal infrared image) of the LANDSAT-5 satellite (acquisition date 22-05-1986), covering the Itea study area. Freshwater plumes (dark tones), resulted from submarine springs (O~) are shown in test areas 1 and 2.

Figure 9. Close view of ground water-coastal discharge (spring), in test area no. 1, inside the red dashed line.

Figure 10. Ground water-coastal discharge (spring), in test area no. 2, inside the red dashed line.

5. Discussions and Results

From the aforementioned two case studies, the following conclusions were drawn:

The use of the available high resolution TM data of LANDSAT-5 satellite seems to be an excellent tool for geoscientists studying hydrogeological phenomena, such as salt-water intrusion and groundwater discharge (submarine springs) in submarine coastal areas. In particular:

A) In the Gulf of Itea, with the help of digitally processed TM6 thermal band, two test areas were located, showing submarine groundwater discharge. This thermal band contributed significantly to the detection of temperature changes of the sea-surface, which occur due to submarine freshwater discharges. TM6 band, despite its relatively low spatial resolution (120 m), detected and located the sea-surface fresh-water plumes because these plumes were relatively large in size.

In future studies, more accurate delineations of plumes can be achieved, if higher resolution satellite thermal images are used, such as the Enhanced Thematic Mapper (ETM+) data of LANDSAT-7 satellite with 60-m spatial resolution thermal band and/or ASTER of TERRA satellite, with 90-m spatial resolution thermal bands.

B) In the Pieria coastal area, the boundaries between salt-affected and salt-unaffected vegetation (crops) could be delineated with the help of multi-temporal TM images. These boundaries can be seen more clearly on the PC2 image of August 1999 than on the image of April 1986. This occurs because salt-affected plants are more stressed during summer, when water resources are scarce.

C) The use of very high resolution satellite data, such as those of SPOT, IKONOS and QuickBird satellites, may not be useful for detection of groundwater discharges (submarine springs), since these satellites lack the ability to record thermal bands. Of course, very high resolution data provided by these satellites could prove useful in detection and delineation of coastal areas suffering from salt water intrusion.

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EFFECTS OF FLY ASH FROM COAL-BURNING ELECTRICAL UTILITIES ON ECOSYSTEM AND UTILIZATION OF FLY ASH

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Abstract: Electric power plants that burn fossil fuels emit several pollutants linked to the environmental problems of acid rain, urban ozone, and the possibility of global climate change. Not only are gaseous and particulate emissions from coal-fired power plants of environmental concern, but also their byproduct, fly ash may lead to contamination because of the possible release of both major and trace elements. Although there are serious efforts to use fly ashes as construction materials or soil amendments, the amount of the stocked waste ash keeps increasing because production exceeds the amount that can be used in the construction industry. Waste ash that cannot be used in above-mentioned industries needs to be disposed of. The safe disposal of waste ashes requires adequate identification and classification of their heavy metals as well as their toxicity levels. This paper summarizes the effects of the fly ash from coal-burning electrical utilities on ecosystems and provides information about how to beneficially use this kind of ash.

Keywords: contamination, fly ash, thermal power plant, waste

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1. Introduction

Mining activities have been a current source of pollution of trace elements introduced into the atmospheric, terrestrial and aquatic ecosystems. Many trace elements such as arsenic (As), lead (Pb) and mercury (Hg), are known as environmental pollutants, since they are toxic to ecosystems. Chemical contamination has been reported in areas where mining and smelting have been carried out since the 1900s, and where significant amounts of various elements were mobilized by weathering and leaching from mining wastes. Mining operations and their mining waste disposal methods are considered one of the main sources of environmental degradation. Social awareness of this problem is of a global nature and government actions to stem the damage to the natural environment have led to numerous international agreements and laws directed toward the prevention of activities and events that may adversely affect the environment. The impact on ecosystems due to wastes from mining and processing activities may appear in groundwater, surface water and soil. Migration of contaminants from waste disposal sites to surrounding ecosystems is a complex process. Soil and water contamination around ash disposal sites (Deborah and Ernest, 1981; Suresh et al., 1998; Gulec et al., 2001; Baba and Kaya, 2004) has recently been the subject of considerable research world over.

Use of lignite in power generation has led to increasing environmental problems associated not only with gaseous emissions, but also with the disposal of ash residues. In particular, use of low quality coals with high ash content results in huge quantities of fly ash to be disposed of. A main problem related to fly disposal is the heavy metal content of the residue. When low quality lignite is burned, its fly ash contains several toxic elements, such as lead (Pb), zinc (Zn), cadmium (Cd), nickel (Ni) and cobalt (Co), which can leach out and contaminate soils as well as surface water and groundwater. The extent of the heavy metals in fly ash depends on both the mineralogy and particle size distribution of the raw material being burnt and the combustion temperature. Although the extent of the heavy metals can be optimized by controlling the particle size and burning temperature, such procedures could be costly. Furthermore, since the coal mineralogy is generally constant for a given coal deposit, not much can be done to control the heavy metal content in fly ash. However, leaching of heavy metals from fly ash can be prevented by adequate waste disposal techniques. The leached heavy metals from fly ash may become a hazard to the environment because of their contribution to the formation of toxic compounds. This can lead to health, environmental and land-use problems (Davison et al., 1974; Kaakinen et al., 1975; Klein et al., 1975; Campbell et al., 1978; Wangen et al., 1978; Gehrs et al., 1979; Hansen and Fisher, 1980; Hulett et al., 1980; Laumakis et al., 1996; Inyang, 1992; Georgakopoulos et al., 1994;

Fernandez-Turiel et al., 1994; McMurphy et al., 1996; Kamon et al., 2000; Baba and Turkman, 2001; Georgakopoulos et al., 2002a; Georgakopoulos et al., 2002b; Mandal and Sengupta, 2002; Baba, 2003; Baba et al., 2003; Baba and Kaya, 2004).

Coal-burning electrical utilities annually produce millions of tons fly as a waste by-product world-wide, for example in 1996 approximately 800 million tons of coal was burned in the US to produce electricity. This lead to the generation of over 90 million tons of ash (Butalia, 1996), and the environmentally acceptable disposal of this material has become of increasing concern. Disposal of coal fly ash is a major environmental concern in India for over 100 million tonnes of ash produced yearly (Praharaj et.al., 2002). It is estimated that 125 million tonnes of fly ash were produced annually in China (Ma et al., 1999). Ash produced during 2002 approximated to 12.5 million tones for Australasia (Australia and New Zealand) (Heidrich, 2003). Also, Turkey has eleven coal-burning electrical utilities, which produce about 15 million tonnes of ash per year. Except for some countries such as U.S., India and some European countries, fly ashes have not been utilized in many part of the world. Also these countries have not used much of these wastes. For example, US have used just 33 percent of their ash in 1998 (ACAA, 1999).

Past and recent research has established the potential of fly ash for use in a variety of construction applications, such as fills, concrete, pavement, grounds and others; however, there is still a need to find new uses, and increase utilization, so less ash will need to be disposed. Also, environmental improvements usually result from utilization because of the engineering controls involved (Usmen et al., 1992).

2. Physical and Chemical Properties of Fly Ash

2.1. PHYSICAL PROPERTIES

The physical properties of ash depend upon a number of factors, including the type of coal burned, the boiler condition, the type and efficiency of the emission controls, and the disposal method (Adriano et al., 1980). Certain characteristics tend to be similar in most ashes. Fly ash is mainly composed of silt-sized material having a diameter from 0.01- 100 µm (Chang et al., 1977). Fly ash is a very fine, powdery material composed mostly of silica. Fly ash is composed of essentially spherical particles, which have a smooth texture. When compared with mineral soils, fly ash has lower values for bulk density, high surface area, hydraulic conductivity, and specific gravity (Stewart and Tyson, 1996). Both crystalline (mullite) and amorphous (glass) phases have been identified by X-ray diffraction in fly ash (Mattigod et al., 1990).

The pH of fly ash can vary from 4.5 to 12.0 depending on the sulfur content of the parent coal (Adriano et al., 1980). Fresh unweathered fly ashes can have pH a value higher than 9 but it is rare to find pH values higher than 8.5 for weathered fly ash. Ash values of up to 12.5 have been reported for ashes produced in the western US (Chang et al., 1977). Many fly ash material have a near neutral pH when originally placed in the pile, the pH of ash usually drops rapidly as pyritic materials weather.

An environmental concern has been the possibility of leaching of heavy metals from the fly ash into the underlying groundwater. The process of stabilizing metals in the fly ash and thereby preventing them from leaching can lead to a material, which has a low permeability, resulting in reduced flow through the fly ash. For this reasons permeability of fly ash is important. For example; permeability for unstabilized Yatagan (Turkey) was found to be 9.82 x 10^{-6} cm/sec, and for Soma (Turkey), the permeability was 5 x 10^{-8} cm/sec. Ashes, in general, showed a trend of decreasing permeability with the addition of stabilizer (Usmen et al., 1992). Limited data are available for permeability of U.S ashes. The permeability of U.S. fly ash ashes gave values ranging from approximately 2 x 10^{-4} to 5 x 10^{-5} cm/sec. Coefficient of permeability data cited for British ashes range from 5 x 10^{-7} to 8 x 10^{-5} cm/sec at maximum dry density (Seals, 1996). Generally, fly ash has low bulk density (1.01-1.43 g/cm^3), hydraulic conductivity, and specific gravity (1.6-3.1 g/cm^3) (Roy et al., 1981; Tolle et al., 1982; Mattigod et al., 1990).

2.2. CHEMICAL PROPERTIES

Chemical composition of fly ash varies depending on the quality of coal burned and the operating conditions of the thermal power station. Approximately on an average 95 to 99% of fly ash consists of oxides of silica, Al, Fe and Ca and about 0.5 to 3.5% consists of Na, P, K and S. The chemical properties of fly ash will largely be determined by the metal oxides that were surface adsorbed during particle formation. In the U.S., fly ash from eastern coals, which usually have higher sulfur content, tend to be higher in Fe, Al and S and lower in Ca and Mg when compared to those derived from western coals. Ash from eastern coals also tends to be higher in the trace elements As, Cd, Cr, Pb, V and Zn. Most of these elements can substitute into the iron pyrite structure, and coals higher in pyrite therefore tend to produce fly ashes, which contain higher levels of these elements (Stewart and Tyson, 1996).

Fly ash is classified based on the nature of constituents present. Class C fly ashes contain less than 70% but greater than 50% of silica, alumina, and iron oxide which are typical for western U.S ash. When the ash concentrations of these three constituents exceed 70%, fly as is classified as F, which is representative of those produced from eastern U.S. coal (Horn, 1995). Class C fly ash usually is referred to as 'high lime ash'. Many power plants in the west and mid-western states fueled with low sulfur coals from Wyoming and Montana that yield high-lime fly ash (Adriano et al., 2002). The chemical properties of fly ash for use in concrete are defined by ASTM Standards C 618-84 Standard Specification for fly ash and raw or calcined natural pozzolan for use as a Minerals into three classes, two of which are fly ash (Fonsdorff and Clifton, 1981). Both type of ash are pozzolanic (to varying degrees) meaning that although non-cementitious in raw form, they will react with lime to produce cementitious products in the presence of moisture and favorable temperature conditions (Usmen et al., 1992). The technology for use of fly ash in cement concrete and stabilized road bases is fairly well developed and has been practiced many years. Some research has identified many benefits of the addition of fly ash in concrete mixes, which include; improve workability; reduced heat of hydration; increased ultimate strength; increased resistance against alkali aggregates; resistance to sulfate attack; reduced permeability; and economy (Boles, 1986; Ahmed and Lovell, 1992).

3. Effects of Fly Ash on Ecosystems

Primary environmental concern of ash utilization is the release of certain environmentally deleterious and toxic constituents of ash into the air, soil and groundwater. Toxic metals and certain toxic and carcinogenic organic compounds are potentially the most dangerous of the ash constituents. The U.S Environmental Protection Agency (EPA) has strict guidelines on the allowable levels of certain constituents of waste materials released into the environment. The EP Toxicity test and Toxicity Characteristic Leachate Procedure (TCLP) are two of the methods employed by U.S. EPA to determine whether or not a material exhibits toxic characteristics.

The distribution of toxic trace elements in fly ash particles and their leachabilities were found to be primarily dependent on the amount of unburnt carbon and iron in fly ash. The leachability of metals from fly ash depends on the nature of the leaching medium, solid liquid ratio, temperature, and pH of the medium. The effect of leaching time greater than four hours was found to be negligible on the leaching of fly ash under reflux boiling conditions. High temperature and low pH favoured the leaching of iron from fly ashes and concentrations of major and trace elements (except Ca) in the leachate followed the similar profile as that of iron under otherwise identical operating conditions (Khanra et al., 1998).

It was noticed that the addition of ash both to calcareous and acid soils, at rates ranging up to 8% by weight, caused an increase in the yield of several crops, while higher levels often produced a decrease (Scotti et al., 1999). The negative effect was put down to reduction in bioavailability of some nutrients due to high pH (generally from 8 to 12), high salinity and high content of phytotoxic elements, especially boron (Aitken and Bell, 1985; Adriano et al., 1980). The addition of ash to the soils causes a variation in the primary composition of both soil (Chang et al., 1977; Elseewi et al., 1980; Petruzzelli et al., 1986) and plants (Adriano et al., 1978; Elseewi et al., 1980; Francis et al., 1985). The change in plant composition results both from high contents of nutrients and toxic elements in the fly ash and from their solubilities that are mostly due to the pH of the ash-soil mixture (Adriano et al., 1982; Elseewi and Page, 1984; Scotti et al., 1999).

3.1. EFFECT OF CHEMICAL PROPERTIES ON ECOSYSTEMS

Fly ash is enriched in many trace elements, particularly metals. These metals are part of the pyrite structure in the coal and become concentrated in the fly ash during the combustion process. These elements may be surface adsorbed on the glassy spherical fly ash particles. Elements that are surface adsorbed can be quite mobile. Many of these trace elements could be quite leachable under low pH conditions.

Trace amounts of antimony, arsenic, barium, beryllium, chromium, cobalt, copper, lead, manganese, mercury, nickel, selenium, thallium, vanadium and zinc are present in coal. When electric utilities burn coal, these elements are released. Most of these elements are carried by particles of ash, mainly on their surface. Coal-burning power plants are equipped with devices to capture ash particles before they reach the air. Particle control devices typically capture more than 99% of the ash, so very little ash enters the air (EPRI, 1998). Heavy metal-carrying ash captured by these devices is usually sent to ash ponds or land disposal sites. Most of these elements such as arsenic, barium, chromium, lead and zinc dissolve in water are carried to the groundwater and soil by rain and snow. For example, U.S. Environmental Protection Agency (EPA) estimates that each year U.S. power plants release about 60 tons of arsenic into air-56 tonnes from burning coal. Also, U.S. power plants release about 63 tons of lead into air in 1995-57 tonnes from burning coal. In Kentucky, approximately 3 million tonnes of coal ashes are produced annually. Disposal of fly ash is a major issue because of the ash's potential to contaminate groundwater with arsenic, boron, and heavy metals (Evangelou and Neathely, 2005).

Also, one study was done in India, to monitor the ground water quality in order to determine the potential impact of ash ponds. It was found that ground water quality was deteriorated due to the presence of fly ash ions (macro and micro such as Fe, Ca, Mg etc.) which were leached out from the ash to some extent. Contamination is likely to increase with toxic and other ions with the passage of time (Suresh et al., 1998).

The groundwater chemistry and the nature of the suspended colloids (size, composition) strongly suggest that fine fly ash particles were suspended and therefore moving with the groundwater flow. At wells exhibiting large amounts of suspended colloids ($\approx$10–100 mg L^{-1}), the water was enriched in CO_2 and depleted in O_2. The colloids were typically between 0.1 and 2 μm in size and were primarily silicates. These results show that, where infiltrating water is percolating through a site that has been mixed with fly ash, the secondary carbonate mineral in the soils are being dissolved; removal of this cementing carbonate phase may consequently release soil silicate colloids to be carried in the flowing water (Gschwend et al., 1990).

Fly ashes are placed in an underground coal mine in US to control subsidence. Some ash materials were characterized to determine potential groundwater impacts. No problems were found with respect to heavy metals. Fly ash leachates are high in dissolved solids and sulfates. Chloride and boron from fly ash may also be leached initially in high concentration (Singh and Paul, 2001).

Many forest ecosystems in Germany are strongly influenced by emissions of pollutants like SO_2 and alkaline dusts. A study was conducted in pine stands in the Dubener Heide in Northeastern Germany. The results showed that this forest area soil was influenced mainly by emissions from coal-fired power plan (Klose et al., 2001).

Fly-ash also affects the physicochemical characteristics of soil because it is generally very basic, rich in various essential and non-essential elements, but poor in both nitrogen and available phosphorus. The massive fly-ash materials have been a potential resource for agricultural activities as well as other industrial purposes. Practical value of fly-ash in agriculture as an ‘effective and safe’ fertiliser or soil amendment can be established after repeated field experiments. What remains to be disclosed here is the biological processes and interactions due to 'lack and excess' of the fly-ash exposures along with abiotic and biotic factors. These may involve the symbiotic fixation of nitrogen and the biological extraction of metals following immobilisation of toxic heavy metal ions, as well as other neutralisation and equilibration processes during weathering (Gupta et al., 2002).

The most severely acidic conditions, which come from the waste by-products of the coal-fired thermal power plant, are found in the eastern United States. Environmental Protection Agency (U.S. EPA) believes that acid rain has been the primary cause of the acidification of hundreds of streams in the mid-Atlantic highlands and the New Jersey Pine Barrens and of many lakes in the Adirondack Mountains of New York (U.S EPA, 1994). The National Acid

Precipitation Assessment Program (NAPAP) identified acid rain as one of several possible causes of increased nitrate leaching and acidification of surface waters in several northeastern watersheds. Acidification is believed to harm populations of fish and invertebrates in small streams and lakes (NAPAP, 1992; Carlin, 2002).

3.2. EFFECT OF RADIOACTIVE ELEMENTS ON ECOSYSTEMS

Some trace elements in coal are naturally radioactive. These radioactive elements include uranium (U), thorium (Th), and their numerous decay products, including radium (Ra) and radon (Rn). During coal combustion most of uranium, thorium, and their decay products are released from the coal matrix and are distributed between the gas phase and fly ash. The partitioning between gas and solid is controlled by volatility and chemistry of the individual elements. Virtually 100 percent of the radon gas present in feed coal is transferred to the gas phase and is lost in stack emissions. In contrast, less volatile elements such as thorium, uranium and the majority of their decay products are almost entirely retained in the solid combustion wastes (USGS, 1997). The average ash yield of coal burned in the United States is approximately 10 percent by weight. Therefore, the concentration of most radioactive elements in solid combustion wastes will be approximately 10 times more in the original coal. The concentration of uranium in fly ash is changed from 10 to 30 ppm in U.S. (USGS, 1997). Leachability of radioactive elements is critically influenced by pH that results from the reaction of water with fly ash. Extremes of either acidity ($pH < 4$) or alkalinity ($pH > 8$) can enhance solubility of radioactive elements (Tadmore, 1986).

Fly ash, the major ingredient in Autoclaved Cellular Concrete (ACC), contains elevated concentration of uranium-238, thorium-232, and their radioactive decay products. Radon-222 is a chemically inert, radioactive gas formed several species down the uranium decay chain. The large macropores and interconnected micropores of ACC facilitate the outward diffusion of the produced radon (Laton et al., 1996).

A coal-fired thermal power plant (750 MW_{el}) has been in operation since 1972 in Velenje (Slovenia) and currently produces almost a million tonnes of fly ash per year. Fly ash with uranium content of at least 25 mg kg^{-1} is transported as a slurry and was disposed at first into a lake and later into wet ponds on a depository of an area of 0.50 km^2. The deposited ash has direct contact with the lake water. Leaching of radionuclide from fly ash into lake and rain water and pile seepage water are the main sources of radioactive contamination of the lake and its outflowing waters according to Mljac and Krizman, 1996.

One of the studies was done radionuclide concentration in fly ash Yatagan (Mugla) in Turkey. This study shows that uranium concentration in fly ash varies from 25.03 to 36.40 ppm. Thorium concentration ranges between 20.0 and 30.13 ppm in fly ash. The average level of uranium and thorium in fly ash collected from the thermal power plant is 28.72 and 16.13 ppm (Baba, 2002).

4. Utilization of Fly Ash in Engineering Applications

The major issues related to the utilization of fly ash depend on the type of collection system, source and type of coal, plant operating conditions and the temperature of combustion. The variability of the physical and chemical properties of fly ash can be monitored by through comprehensive testing.

Over the past few decades, use of various waste products in highway construction has gained considerable attention in view of the shortages and high costs of suitable conventional aggregates, increasing costs of waste disposal, and environmental constraints. Use of waste by-products as economical replacements for conventional materials such as natural soils and aggregates can alleviate disposal costs and environmental pollution and conserve high-type highway materials for higher-priority uses (Usmen et al., 1983).

Most of the fly ash presently produced by electric utilities and industry is landfilled or stored in disposal ponds, although approximately 33% was beneficially utilized for various purposes in 1998 in US (ACAA, 1999). Landfilling is not an optimal solution for disposal because of landfill space limitations and tipping costs. Many industries are also facing rising regulatory and internal "green" corporate demands to reduce their waste disposal streams. As a result, the use of fly ash as a soil amendment in the reclamation of disturbed areas became a research topic of growing interest in the early 1990's (Daniels et al., 2002).

Fly ash has long been utilized in some European countries, particularly in Great Britain, since 1930 (Usmen and Chou, 1990). When treated and applied correctly, fly ash can be put to multiple productive uses in civil engineering, mine reclamation and agricultural application. Potential uses of fly ash include:

- raw material in portland cement manufacture
- replacement for cement in concrete and grout
- cement replacement in precast concrete products
- aggregate for the stabilization of highway subgrades
- aggregate for road base material

- material for structural fill
- raw material for metal reclamation
- filler material in plastics
- sanitary landfill cover or liner
- backfill for controlling subsidence in abandoned mines
- backfill for fighting mine fires
- amelioration of soils
- raw material in brick manufacture
- mine subsidence and acidic drainage control
- material for absorbing oil spills, and
- absorbent for dewatering sewage sludge etc.

As the use of ammonia-related technologies by coal-burning electric utilities becomes more widespread in North America, utilities and marketers must address options for the management of fly ash containing varying amounts of ammonia (NH_3). Potential users of fly ash treated with NH_3 must understand that there are acceptable levels of ammonia for use in cement and concrete, as well as in other beneficial applications (Theodore, 2000).

Fly ash can be used in the manufacture of aggregate, horticultural applications and autoclaved cellular concrete. Generally, class F fly ash (ASTM C-618) is considered suitable for autoclaved cellular concrete (Pytlik and Saxena, 1996), which is a lightweight building material with unique properties for application in interior and exterior construction. The concentration of heavy metals in leachates of crushed autoclaved cellular concrete (ACC) were below 100 times their applicable drinking water standards, which is the regulatory hazard threshold. The possible microencapsulation in the ACC concrete-structure, and the moderate to high pH of ACC leachates, contributed to the low concentration of heavy metals released to and solubilized in the aqueous extractants (Laton et al., 1996).

Controlled compaction of fly ash permits up to 19 percent by weight more storage in disposal areas. With drainage, the fly ash can be effectively and economically utilized as a fill material to construct stable embankments for land reclamation on which structures can be safely founded (Brendel and DiGioia, 2000).

The use of fly ash at coal mining facilities has increased significantly in recent years due to pressure from utilities and industrial customers, regulatory agencies and environmental groups. Few power companies partially filled a small underground mine on its property with a ground composed of fly ash in US. In this case, the company had an incentive to stabilize the deep mine workings so that it could expand its existing surface ash/sludge disposal area over the mine. Although the mine voids contained acidic water, this was no detectable surface discharge and no evidence of groundwater contamination. To date, there have been no documented cases where coal combustion by-products have been injected into underground mines for the primary purpose of reducing acidic mine drainage (Aljoe, 1996).

The potential use of fly ash in agriculture has been explored by various research agencies, scientists and institutes. Several reports are available on the impact of its use in agriculture. In agriculture, gypsum provides valuable macro and micro nutrients and helps to maintain soil moisture. Fly ash, when tilled into the soil, encourages better root growth (ACAA, 1996).

Soil modification, which is the changing of soil behavior principally through the reduction of excess moisture to expedite construction, is an effective and economical construction expediting technique with generally modest engineering requirements. In most instances, soil modification of construction activity. A wide range of soil problem soils can be modified with class C fly ash to improve behavior. Also, class C fly ash stabilization is the value-engineering route to increasing pavement life cycle. For example; soil stabilization and modification has been very successful in the Rocky Mountain Region, USA (Roof, 1996).

Dairymen are interested in using the fly ash byproduct from electrical co-generation plants in their corrals and bedding. The ash reportedly provides a good base material in the corrals when used at interfaces between concrete surfaces and dirt. Fly ash has a high pH compared to manure. Fly ash will reduce coliform bacterial growth when mixed with various forms of manure (Kirk et al., 1998). The reduction seems to be in proportion to the increased pH of mixture.

Fly ash can potentially serve as an alternative liming material without negatively affecting corn production in areas where use of conventional liming materials can be uneconomical due to transportation costs (Tarkalson et al., 2005).

The application of an amendment composed of fly ash and sewage sludge mixtures on sandy soil could increase the enzyme activity and reduce the availability of heavy metals. The decreased soil pH indicated that sandy soil amended with 10% of ash-sludge mixture had a higher mineralization rate. The dehydrogenase activity in sludge-amended soil was suppressed by the addition of fly ash. The addition of fly ash has a beneficial effect on the nutrient cycles of N and P. In terms of nutrient cycling, addition of sludge and fly ash in some cases will benefit the soil (Lai et al., 2000).

Stabilization/solidification technology is the most widely used technique for the treatment and ultimate disposal of both radioactive and chemical hazardous wastes. Cement-based products, commonly referred to as grouts, are the predominant materials of choice because of their associated low processing costs, compatibility with a wide variety of disposal scenarios, and ability to meet stringent processing and performance requirements. Class F ash has been the material of choice, primarily because the specifications in ASTM C 618 have been sufficient for the quality control required in meeting the physical performance requirements of the ground product (Gilliam, 1996). Because of this characteristic, Class F fly ashes have been used in radioactive waste disposal in USA.

Laboratory column studies were used to predict the effect on the leachability of lead when using fly ash or a fly ash/sludge mixture as a cover for a lead tailings site by Clevenger and Dave (1998). A high pH fly ash cover produced a leachate with a pH 12. This was sufficiently high to allow for the formation of lead hydroxide complexes, which are slightly soluble. Therefore, the leachate had an average lead concentration of about 5 mg L^{-1}, while the pH in the leachate from the column with only tailings was 7.80 and lead concentration was below the detection limit ($\leq$0.1 mg L^{-1}). The fly ash cover changed the amount of the remaining lead, making it less available. Clevenger and Dave (1998) also mention that rainfall rate did not affect the fly ash cover.

5. Summary and conclusions

Effects of fly ash on the ecosystem and utilization schemes of fly ash have been reviewed in this paper. It is noted that coal-fired power plants all over the world can be major sources that generate huge quantities solid wastes. Most part of this waste contains fly ash, which can environmentally affect soil, water and air. Especially, fly ash is rich in many trace elements, particularly heavy metals. Many of these heavy metals can be leachable under low pH conditions.

In 1976, US Congress passed the Resource Conservation and Recovery Act (RCRA) attempting to encourage the reuse of potential resources and minimize disposal problems. In 1983, the US EPA issued a guideline for the procurement of cement and concrete containing fly ash. This guideline has helped encourage the use of fly ash and the elimination of many specifications, which prohibited its use even though it was often technically feasible and economical.

Several areas of fly ash utilization involving technology demonstration projects have been completed or are underway. These include mine filling, construction of roads, embankments, hydraulic structures, raising of dykes, manufacture of several building components like bricks, blocks, tiles and its use in agriculture. The future poses challenges to the scientists, technologists and engineers towards sound management of fly ash.

To preserve the ecosystem and develop a successful marketing program, fly ash producers need to address the following:

- Use of fly ash in the manufacture of concrete, grouts, flowable fill, stabilized road base, cement manufacturing and permeable backfill for retaining walls is feasible.
- Successful ash marketing programs require support of top management. Ash marketing groups need to develop communication tools, which adequately measure performance and the net benefits the organization achieves from ash marketing.
- The best way to avoid the development of new disposal facilities is to establish new utilization schemes with fly ash.

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GROUNDWATER AGE: A VITAL INFORMATION IN PROTECTING THE GROUNDWATER DEPENDENT ECOSYSTEM

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Abstract: Economic gains of use have led to a global explosion of groundwater development in the last several decades. Consequently, groundwater reserves have been depleted extensively. Continuing use of groundwater, which is initially supplied from the storage, causes increasing derivation of additional water from groundwater dependent ecosystems such as, streams, lakes and wetlands. A systematic groundwater age dating in the vicinity of a surface water body may help to quantify the spatio-temporal dynamics of interaction between these resources. Though, numerical flow and transport models may be used to infer the age distribution of groundwater feeding a surface water body, their efficient use requires extensive data that properly characterize the flow domain. In cases, such data is not available or requires to be supplemented by an independent approach, spatio-temporal age dating of groundwater by various tracers can be helpful in understanding the dynamics of flow in the aquifer. This paper provides brief information on how the groundwater age data can be used in surface water ecological problems. Examples from several field sites in Turkey are also presented.

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Keywords: groundwater age dating; aquatic ecosystem; Turkey

1. Introduction

An ecosystem consists of a dynamic set of living organisms that benefit from each other's participation via symbiotic relationships. These organisms interact among themselves and with their environment in which they live. An ecosystem is called "groundwater dependent ecosystem" (GDE) when its sustenance depends on groundwater input.

Under natural conditions, the amount of water stored in aquifer is in dynamic equilibrium between recharge and discharge. Because of the dampening function of aquifer, short-term "natural" changes in recharge are not directly reflected on discharge to GDE. Regardless of its scale, exploitation of groundwater disturbs the dynamic water balance. At the initial stage of groundwater development, the head decline in the aquifer reduces the amount of recharge to GDE. However, if exploitation continues and the groundwater head declines below the lower limit of GDE (e.g. a swamp, lake, estuary), an induced recharge, stealing a part of surface water input to GDE, occurs. Severe groundwater head declines eventually lead to complete dry-up of groundwater dependent ecosystem. Unfortunately, groundwater has been depleted globally partly because of climatic change but due mostly to excessive use of this renewable resource. It is estimated that 80% of reduction of global groundwater reserves is due to man-made use (e.g. Konikow and Kendy, 2005).

Because of the dampening function of aquifer, the effect of over exploitation on the GDE is felt after a "response time". The response time depends on the aquifer diffusivity (i.e. ratio of transmissivity to storativity; e.g. Balleau, 1988) and the distance between zone of groundwater abstraction and the GDE. If the site of groundwater abstraction is far from GDE, the response time may reach to several decades, particularly in large aquifer systems. In many cases, it becomes too late to take counter measures to sustain the groundwater recharge to GDE in order for sustaining the life. Even if the abstraction is fully stopped, it may take many years for natural recharge to recover the "natural" dynamic state to stop induced recharge from the GDE. Therefore, the effect of groundwater use on the GDE must be carefully studied before any development scheme is implemented.

One practical way to anticipate the effect of groundwater development on GDE sustenance is to determine the age of groundwater that feeds them. The response time to over exploitation is much shorter for a GDE that is fed by "young" groundwater (Figure 1). Accordingly, any decline in recharge to eco-system can be recovered quickly once the groundwater abstraction is reduced or stopped. However, in the case of GDEs fed by "old" groundwater, both the response and recovery times are usually much longer (see Figure 1).

Groundwater age can be determined by using numerical flow/transport models (e.g. Modflow and Modpath; Harbaugh, A.W. et al., 2000; Pollock, 1994) and environmental tracers (e.g. tritium and carbon-14). However, both techniques have uncertainties originating from inadequacy of the quality of input data required. About 20 to 30 per cent of flow/transport modeling exercises are estimated to fail in simulating the natural system due to misconceptualization (Bredehoeft, 2005) and the reliability of groundwater age-dating by environmental tracers depends on the information on physical and chemical processes that control the transport of tracer in the aquifer. Both techniques should be used together to cross validate their results.

In the following, first we briefly explain what the groundwater age is and how it is determined by use of numerical models and of environmental isotopic tracers. Some field examples of isotopic age dating from Turkey are also given to demonstrate how groundwater age data is related to sustenance of GDEs (Figure 2).

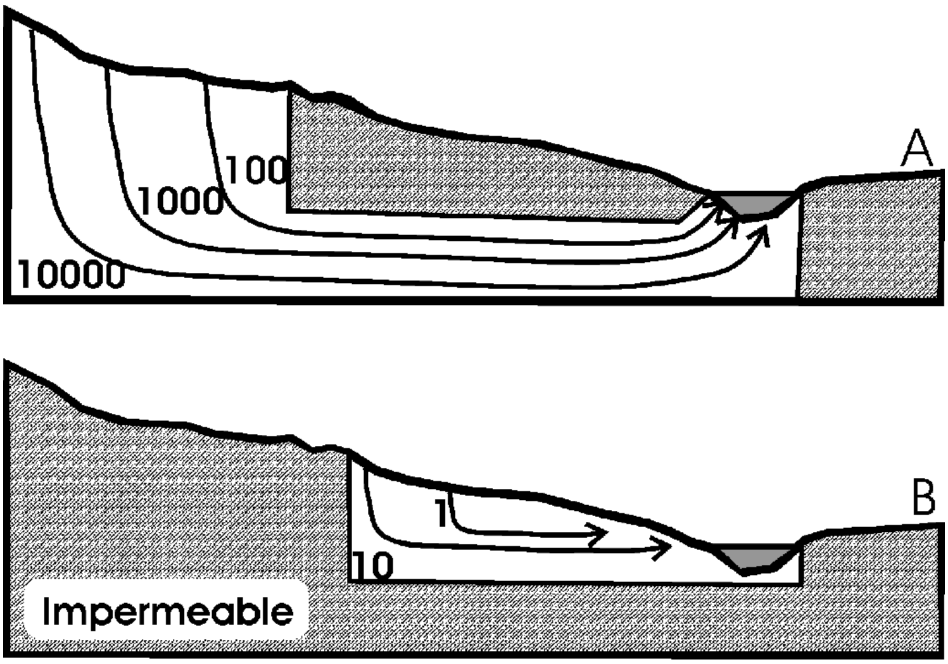


Figure 1. Age of groundwater feeding a GDE (top: GDE fed by old, regional groundwater, bottom: GDE fed by young, local groundwater, numbers denote magnitude of residence time in years).

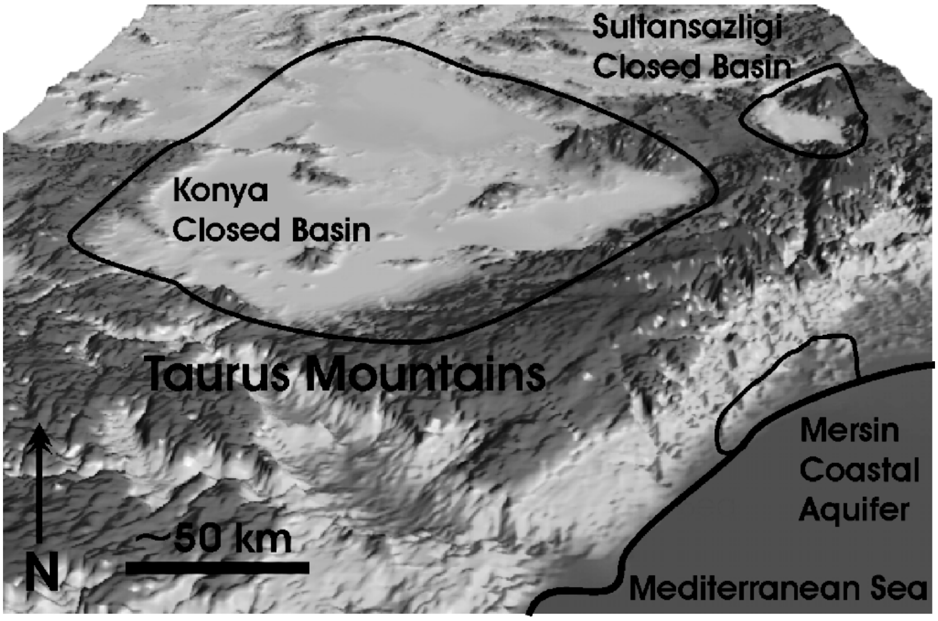


Figure 2. Location of sites mentioned in text.

2. Groundwater Age

2.1. DEFINITION

Groundwater age (also called residence time or time of travel) is the length of time that groundwater spends between recharge and discharge (or sampling). This age is different for every water molecule because they follow different flow paths and are subject to different hydraulic gradients during their journey in the aquifer. It is common practice to use the term "mean age of groundwater" (or mean residence time, MRT). The MRT of groundwater is like the mean age of a population in which there are individuals with ages ranging between, for example, 1 and 100 years. Different populations with the same age span may have different mean ages (MRTs) because of the differing weight of some ages in the mean. In the aquifers, the age span may vary from recent to tens of millions of years. The oldest water molecules are those belonging to formation water in sedimentary rocks or their equivalent in metamorphic-magmatic units.

2.2. NUMERICAL METHODS OF GROUNDWATER AGE-DATING

If, for any given time, the average velocity of a water molecule (v, LT-1) and the length of its flow path (d, L) are known, the time of travel (t, T) can be determined by the equation, T = d/v (T). If the average velocity and the length of flow path is determined from a point where the molecule is entered in the system, this time of travel is equal to the residence time or the "kinematic age" of the associated molecule. In a Darcian flow system, the velocity of water molecule (v, LT-1) is linearly related to hydraulic conductivity (K, LT-1) and the governing hydraulic gradient (I, LL-1). Because both K and I are spatio-temporally variable, the velocity of molecule (v = K I) varies during its journey in the aquifer. Such complications can be overcome by using numerical flow models (e.g. Modflow and Modpath) that determine the velocity of a water molecule and its trajectory in the aquifer at any given time and position. The error associated with kinematic age determined by numerical models is determined by the error of input data used. The uncertainty in the boundary conditions, geometry, geohydrologic properties (i.e. porosity, hydraulic conductivity etc.) and the stress conditions of the aquifer are the major sources of error. Elaborate information on "kinematic age" dating of groundwater can be found in Goode (1996).

2.3. TRACER METHODS OF GROUNDWATER AGE-DATING

A "tracer" is a substance that traces the water molecule from its entry to aquifer until its exit via natural discharge or sampling. Various chemical and isotopic species are used as tracers in groundwater age dating. However, all tracers have drawbacks to accomplish this task. Some of them are degraded, sorbed or lost during their journey and may have additional sources in the aquifer. These sinks and sources disturb the tracer mass balance and complicates the interpretation of observations. Probably the best tracer is tritium (3H) because it is a component of water molecule. Even the tritium can have sources (e.g. 6Li decay) and sinks (e.g. retardation by clay membranes). For the sake of simplicity, we will be dealing only with 3H and 14C as the tracers of groundwater age dating in the following examples. A brief outline of the use of these isotopes in age dating is given below.

Tritium is the only radioactive isotope of hydrogen element and it decays to stable helium-3 isotope with a half-life of 12.3 years. Two important sources of tritium in the hydrologic cycle are cosmogenic and anthropogenic production. Regardless of its genesis, once formed, tritium becomes a part of atmospheric moisture, joins the hydrologic cycle and is transmitted to groundwater as a part of recharge. Cosmogenically produced tritium causes the global precipitation to have a tritium content of 5 to 10 TU (Tritium Unit, is a ratio. 1TU = 1 3H/1018H). Anthropogenic tritium has been produced mainly by thermo-nuclear bomb tests prior to 1963 when tests open to atmosphere are banned. These tests elevated the atmospheric tritium background to several thousand TU as of 1963. Since then, atmospheric tritium content declined exponentially due to radioactive decay and now it is very close to its natural level in many places in the world. Groundwater's tritium content can be qualitatively or quantitatively evaluated to determine the age. Quantitative age dating is accomplished by means of transport models, which require long term observations of tritium input to aquifer and at the sampling point. Qualitative approach simply considers a water sample is more than 50 years old if it has no tritium while a sample with tritium has an age of younger than 50 years. Either by quantitative or by qualitative approach, ages older than ca. 50 years cannot be determined practically by tritium.

Carbon-14 is another isotope that is used to determine groundwater ages between several hundred and 50,000 years (under favorable conditions up to 75 years). Carbon-14 is cosmogenically produced in the stratosphere, converted to CO2 and becomes homogenized (we neglect anthropogenic influences on atmospheric carbon-14 content). By convention, the atmospheric 14CO2 content has 100 per cent modern carbon (pmc) as of 1950. Entry of carbon-14 to groundwater system is principally realized by the respiration of plants' roots (we neglect diffusive/advective bidirectional gas flux between atmosphere and groundwater table). Both the plant metabolism and the respired 14CO2 are in equilibrium with atmospheric 14CO2. Infiltrating recharge water equilibrates with soil 14CO2 gas before reaching at the water table where it is isolated from the 14CO2 supply. During the groundwater's movement in the aquifer, initial carbon-14 starts to decline due to radioactive decay with a half-life of 5730 years. Therefore, lowering carbon-14 content indicates increasing carbon-14 age of groundwater. Because of geochemical and isotopic reactions that add carbon-14 free carbon to groundwater, carbon-14 age dating of groundwater involves a complicated geochemical modeling stage. Otherwise, ages based on simple radioactive decay law may significantly overestimate the age of groundwater. Netpath (Plummer et al., 1994) and Phreeqc (Parkhurst and Appelo, 1999) are among the leading geochemical computer programs that can be used to calculate carbon-14 ages of groundwater.

3. Groundwater Age and GDEs: Examples from Turkey

3.1. KONYA CLOSED BASIN, DRIED UP LAKES, SALT-LAKE

Konya Closed Basin (KCB), located in the central part of Turkey, covers an area of ca. 40,000 km^2 that is surrounded by hills and mountain ranges with peak elevations varying around 1500 to 2500 m (Figure 3). KCB is divided into Tuz Golu (Salt Lake) and Konya basins located at the northern and southern halves, respectively by an elevated peneplain surface that extends in ca.

east-west direction. Konya and Tuz Golu sub-basins have relatively flat surfaces extending at 1100 m and 900 m elevations, respectively. These sub-basins were occupied by two large paleolakes during the cooler climates of Plio-pleistocene (see Figure 3). Maximum depths of Tuz Golu and Konya paleolakes are inferred to be 100 m and 20 to 40 m, respectively (Erol, 1991).

KCB is dominated by semi-arid climate with mean annual precipitation and potential evapotranspiration of 350 mm/year and 1100 mm/year, respectively. Most of the precipitation occurs between late autumn to early spring. The geology of the basin comprises mostly of Plio-quaternary alluvial and lacustrine sediments extending mainly in the paleolake bottoms. Mesozoic aged marine carbonates crops out along the Taurus Mountain Ridge located at the southern boundary. Rest of the water divide is made up of various lithologic units of Mesozoic to Tertiary age. Tertiary volcanics lay over the southwestern and mid-eastern parts.

Abundant groundwater can be accessed everywhere in the KCB via drillings. Karstified carbonates of Neogene and Mesozoic make up the main aquifer. Hydraulic head distribution is relatively smooth and reduces from 1100 m at the flank of Taurus Mountains at the south to 900 m near the Tuz Golu at the north. Because of the smooth topography with fertile soil cover, the KCB has vast agricultural lands. Accordingly, irrigation water has been supplied in increasing amounts since late 1960s. Furrow and sprinkler irrigation methods are commonly used. The number of registered groundwater production wells is around 15,000 while the number of unregistered is estimated to be 7500.

Because of extensive groundwater use, basin-wide hydraulic head has been declining with an increasing speed every year. Mean decline rate of groundwater during the last 4 decades is ca. 1m/year. Apart from the existing Tuz Lake (Golu), the KCB were hosting several shallow lakes, swamps, and salt marshes located mainly in the Konya sub-basin at south. Many of these were remnants of Konya paleolake and were providing invaluable nesting sites for migrating birds. Apparently, these GDEs had been fed by groundwater before severe head decline started in the last several decades during which some wetlands in the Tuz Golu sub-basin has also shrink or disappeared.

A groundwater age-dating study (Bayari et al., 2005) has been carried out to along a south-north transect extending along the whole KCB (see Figure 3). This study showed that a) tritium bearing groundwater exist only in the southernmost part of the KCB where recent groundwater recharge from Taurus mountains occurs, b) carbon-14 age of groundwater increases from sub-recent (i.e. 2 ka) at the flank of Taurus mountains to ca. 40 ka near Tuz Golu, c) absence of tritium in most of the regional flow path indicates absence of recent recharge, d) the velocity of groundwater as inferred from carbon-14 ages is 1 km per 300 years, e) carbon-14 ages are in good agreement with kinematic ages. In addition to these results, this study pointed out that thrifty water use policies have to be employed immediately in order for sustaining the present GDEs. Otherwise, existing GDEs will also dry up in the near future.

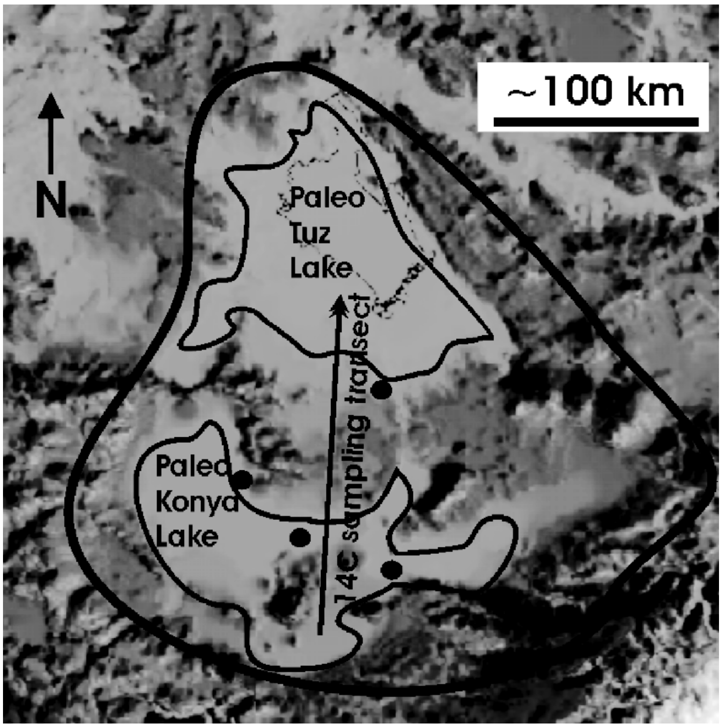
This study also shows the value of groundwater age dating in anticipating the fate of GDEs. If this age-dating study had been carried out well before the present situation existed, the threat directed to disappear GDEs would have been safely forecasted and remedial actions could have been taken. In the next case study, we present a similar case in which groundwater age serves as an early warning tool.

3.2. SULTANSAZLIGI CLOSED BASIN LAKES, BIRD PARADISE

Sultansazligi Closed Basin (SCB) with a drainage area of 3200 km^2 is located at the southwest of Erciyes dormant volcano rising over 3900 m (see Figure 2). The basin hosts a 100 km^2 large wetland that includes Yay and Col lakes and surrounding Sultansazligi and Akar reed fields. The wetland is one of the major nesting sites for migrating birds in Turkey and is under protection according to Ramsar convention. The elevation in the wetland and surroundings ranges between 1070 m and 1150 m while the mountains making up water divide extends over 3000 m. Like many other closed basins in Turkey, larger lakes also occupied the SCB during the Plio-quaternary. Today, semi-arid climate with long-term mean annual precipitation and evapotranspiration values around 400 mm/year and 1000 mm/year, respectively dominates over the wetlands. On the mountains that surround the basin, mean annual precipitation rises up to 600 mm/year.

Almost flat-lying central part of the basin is covered by Plio-quaternary alluvial and lacustrine material with some intercalations of volcano-detritics. The heights on the west, north and east are comprised of tuff and other volcanic rocks while on the south, Paleozoic to Mesozoic karstic carbonates of Taurus Mountains crop out. It is estimated that fractured volcanic rocks of Erciyes and karstified carbonate rocks of Taurus Mountains provide subsurface recharge to the low-lying Plio-quaternary units.

Because the organic rich soils of the basin provide fertile agricultural lands and the groundwater is accessible by drill holes almost all over the plain, the wetlands have long been under pressure of local farmers. Parts of the plain have been drained via ditches that serve to lower the water table and to supply additional water for irrigation. Furrow and sprinkler are the common irrigation methods in the plain where cotton, apple, corn and sunflower seed production is dominant.



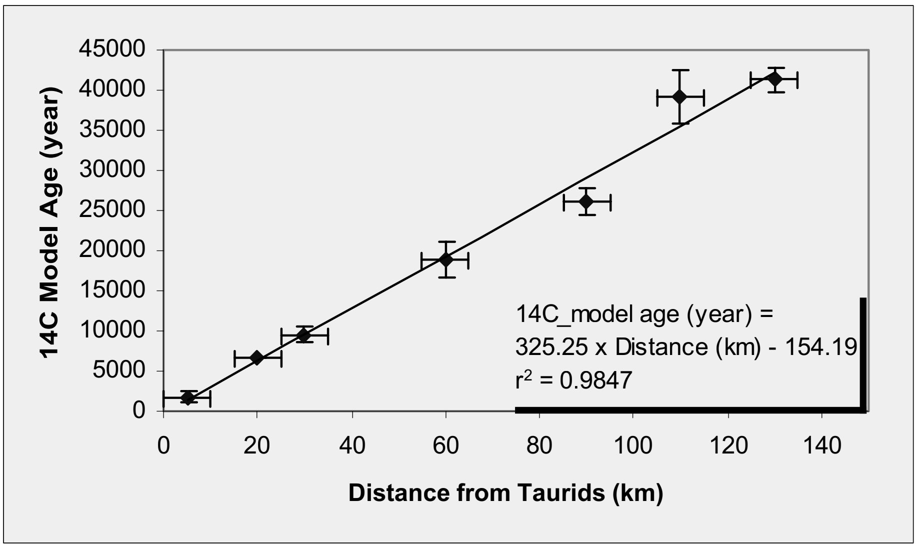


Figure 3. Konya closed basin (top: carbon-14 sampling transect and extent of paleo-lakes, bottom: carbon-14 age of groundwater along regional flow path, full circles show locations of disappeared wetlands).

There is a long lasting debate between farmers and NGOs that whether continuing groundwater use will eventually interfere with the water balance in wetlands of SCB. While one side argues that recharge of wetlands is only by precipitation and associated overland flow, others claim that upward recharge by groundwater can also be important. A groundwater age dating study has been initiated in order to provide answers to these questions. Preliminary results of an isotope hydrology study by Yildiz et al. (2005) indicates that, a) stable isotopic composition of groundwater around wetlands imply a high altitude recharge and, b) groundwater near wetlands have low or no tritium. In addition, these groundwater samples also have low carbon-14 activity (between 14.2 and 24.7 pmc, E. Yildiz, personal communication) that indicates up to 20 ka of tentative ages (Figure 4). Absence of tritium in wetland waters, which are isotopically enriched due to evaporation, provides a strong evidence of groundwater recharge to these GDEs.

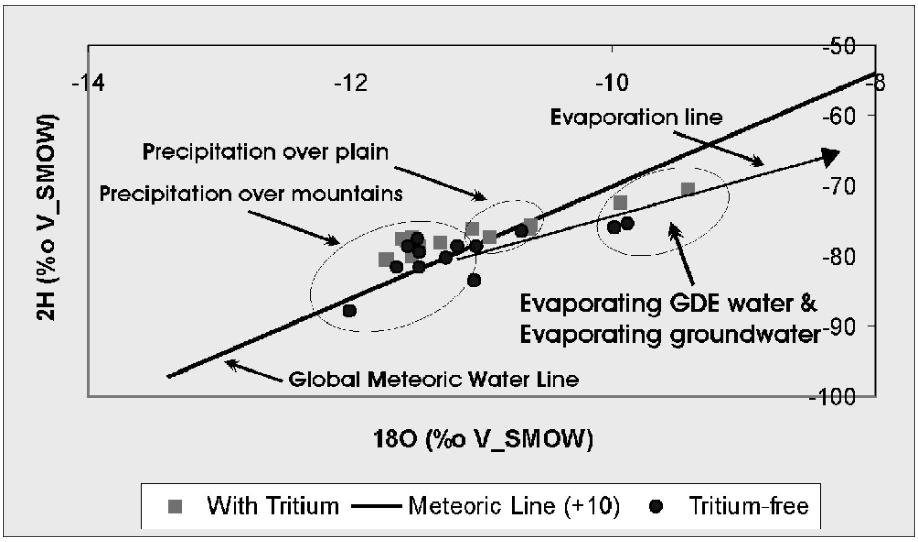


Figure 4. Groundwater's 3H and stable isotope composition in Sultansazligi closed basin.

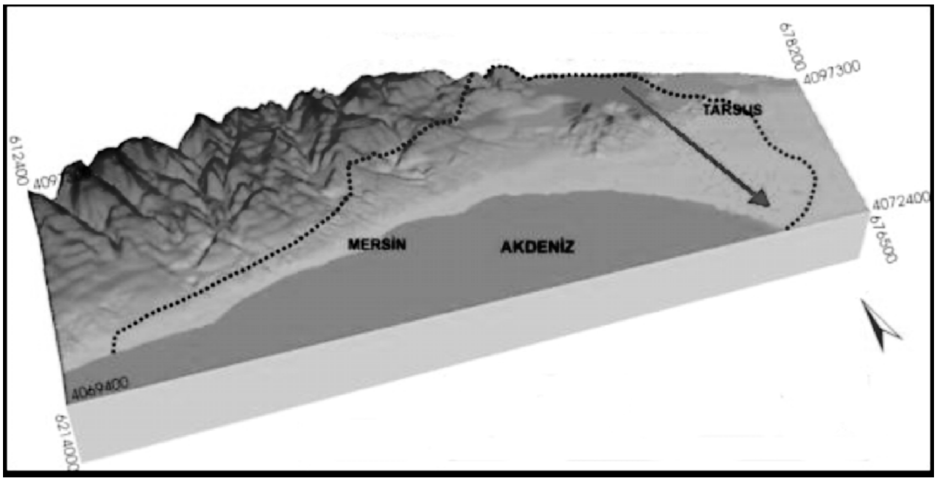
Though, these results need to be verified and extended with additional field data, it appears that groundwater recharge to these wetlands are far from being recent. Therefore, it seems highly probable that continuing over exploitation would continue to decrease groundwater recharge to these wetlands. Even the present groundwater use practice is limited, total recovery of GDEs' water balance seems to take decadal timescales.

3.3. MERSIN COASTAL AQUIFER, BIRD PARADISE AND MARINE AQUATIC SPECIES

Mersin Coastal Aquifer lies along the Mediterranean Sea coast of Turkey along which Taurus Mountains reaching over 3000 m forms a topographic barrier between sea and inland (Figure 5). The area between the mountain front and sea is a low-lying, 800 km^2 large coastal plain comprising of Plio-quaternary sediments of braided river to flood plain type. They lay on Tertiary carbonate and detrital units that are underlain by Mesozoic carbonates of Taurus Mountains. Mediterranean type climate with mild, humid winters and hot, dry summers dominate over the plain. Mean annual precipitation increases from 800 mm/year at the coast to more than 1000 mm/year in the mountains. Mean annual potential evapotranspiration is around 1000 mm/year.

Abundant groundwater are accessed everywhere in the plain via drill wells. Because of the complex sedimentation processes forming the plain, the aquifer is extremely heterogeneous in view of porosity and permeability distribution. Although, impermeable lithologies are common on the plain surface, presence of coarse material at subsurface enables a strong groundwater recharge from mountain side to coastal aquifer. Despite extensive and widespread use of groundwater, seawater intrusion is spatially limited and if occurs, can be recovered in a few years. As a consequence of favorable climate, abundance of groundwater and arable land, the plain is abound with cultivated lands and greenhouse fields. Several wetlands, mostly in the form of estuaries and lagoons exist in the southeastern part of the plain. These wetlands are fed by fresh groundwater and serve as nesting grounds for migrating birds and endemic aquatic species.

Because of the complex distribution of lithologic units in the plain, it is very difficult to construct a conceptual model of hydrogeologic system. Absence of this information prevents application of numerical flow models that could be used to understand the recharge-discharge mechanism of the aquifer. A preliminary survey of environmental isotopic signal in groundwater has been carried out to develop a conceptual hydrogeologic model (Hatipoglu, 2004). Tentative evaluation of 18O, 2H and 3H isotopes revealed that groundwater in the plain a) has a wide elevation range of recharge area extending from coast to mountains, b) the moisture in precipitation recharging the aquifer is of Eastern Mediterranean origin with deuterium excess value of +20. Spatial distribution of tritium in the plain indicates a complex groundwater flow mechanism. In parts of the plain extending along coastline where wetlands exist, groundwater has no measurable tritium. This suggests that, relatively deep circulating groundwater flow that is fed by the heights of Taurus Mountains recharge this zone. Tritium free waters with low recharge area elevations also exist in other parts of the plain that extend towards mountain front.



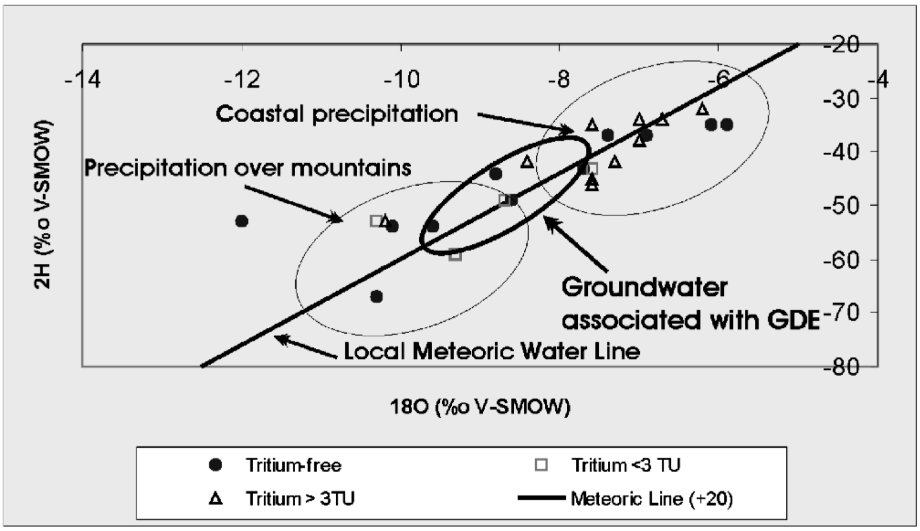


Figure 5. Mersin coastal aquifer (top: digital elevation model with arrow indicating isotopic sampling transect, bottom: Groundwater's tritium and stable isotope composition).

This suggests the presence of low hydraulic conductivity units. However, groundwater with 3H > 3TU, implying a local, young recharge, also exists next to tritium free zones. It appears that, high and low hydraulic conductivity units with fast and slow groundwater velocities exist side by side in the aquifer. This picture is in agreement with the inferred sedimentary depositional model of the coastal plain that comprises of a mixture of braided river and flood plain type facies. The results of this preliminary survey suggest that the groundwater feeding coastal wetlands are older than 50 years and this groundwater is most probably in hydraulic connection with that exist in other parts of the plain. Therefore, over exploitation of groundwater in places far from wetlands may provoke induced recharge in the near future. Additional, spatially high-resolution isotopic data is required for a better understanding of the hydrogeologic system.

4. Conclusions

GDEs fed by young groundwater are more quickly affected by the short-term changes in groundwater balance but any induced recharge caused by over exploitation may also be recovered quickly. GDEs fed by old groundwater are more prone to changes in groundwater balance and once induced recharge starts, it may take decadal timescales for GDE to recover original state. Groundwater age information may provide a fast and detailed picture of the hydrogeologic relationship between the groundwater dependent ecosystem and the groundwater. When combined with other types of hydrogeologic information, groundwater age data could help to better understand the hydraulic relationship between these water bodies.

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GROUNDWATER IN SEMI-ARID MEDITERRANEAN AREAS: DESERTIFICATION, SOIL SALINIZATION AND ECOSYSTEMS

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Abstract: An investigation concerning groundwater quality and its relation with the soil salinization, desertification and associated ecosystems is being developed in Alentejo, a region of South Portugal affected by a semi-arid climate. The precipitation occurs mainly during the cold season, between October and March/April, and the hot season has a heavy deficit of water. The geology is based on sedimentary and volcano-sedimentary rocks affected by the Hercynian orogeny. The plants and trees are adapted to this environment, where specific species resist to these conditions, as the *Quercus suber*, the cork oak. Some saline waters are present in certain areas, mainly on the flat ones, and the natural vegetation is controlled by the groundwater composition. Ecosystems with plants like *Juncus acutus, Juncus subulatus, Hordeum geniculatum* and *Parapholis incurva* are present and the human settlements tend to avoid those areas, where also the agriculture is difficult to implement. The presence of this type of waters is justified by deep faults that can transmit highly mineralized waters from deepness and by the concentration of salts at surface caused by the high values of evapotranspiration.

Keywords: Ecosystems; semi-arid area; mineralized waters.

A. Baba et al. (eds.), Groundwater and Ecosystems, 47–58.

1. Introduction

The climate in South Portugal is semi-arid, with high evapotranspiration levels in summer. In the area of Mértola, Alentejo region, in South Portugal, some groundwater with high levels of mineralization occur. The investigation of these kind of waters and the particular ecosystems that are present, similar to the ones that form some near shore areas, are the beginning of a study that will be continued in the future, in order to identify all the plants that are present on these particular areas.

2. Climate, Geomorphology and Geology

Alentejo region (Figure 1), in South Portugal, is affected by a semi-arid climate, where the precipitation goes from 400 to 800 mm per year (with an exception in a 1,200 m high mountain in the northern part of Alentejo, where it can reach more than 1,000 mm). In this area the potential evapotranspiration values that can go to more than 1,000 mm per year. The precipitation occurs mainly during the cold season, between October and March/April, letting the hot season with a heavy deficit of water, when the temperatures can reach more than 45ºC, during some days.

The geomorphology of Alentejo region is characterised by an extensive flat area with some residual relief. An exception is the S. Mamede mountain (1,200 m), which lies in NE of Alentejo. The W and NW zone consists of the littoral and river depressions, which are influenced by two main rivers, the Tejo and Sado sedimentary basins. In this area the sediments cover the Iberian Shield.

The South of Alentejo, where these high EC waters occur, is a flat area, with exception of the vicinity of the major rivers, which, due to the variations of sea level, cut deeply the landscape. This is the case of Guadiana River and its tributaries near the working area.

The geology of Alentejo is composed by three main geostructural domains of the Iberian Peninsula Precambrian and Palaeozoic Shield (Chacón et al., 1983): the Central Iberian Zone (CIZ), the Ossa Morena Zone (OMZ) and the South Portuguese Zone (SPZ) and by some sedimentary rocks on the NW and west parts. These kind of waters occur in the last one of the crystalline domain (Figure 2), the South Portuguese Zone, where the geology is represented by metamorphic rocks like shales, schists, phylits, greywackes, quartzites, acid and basic metavolcanic rocks, between others.

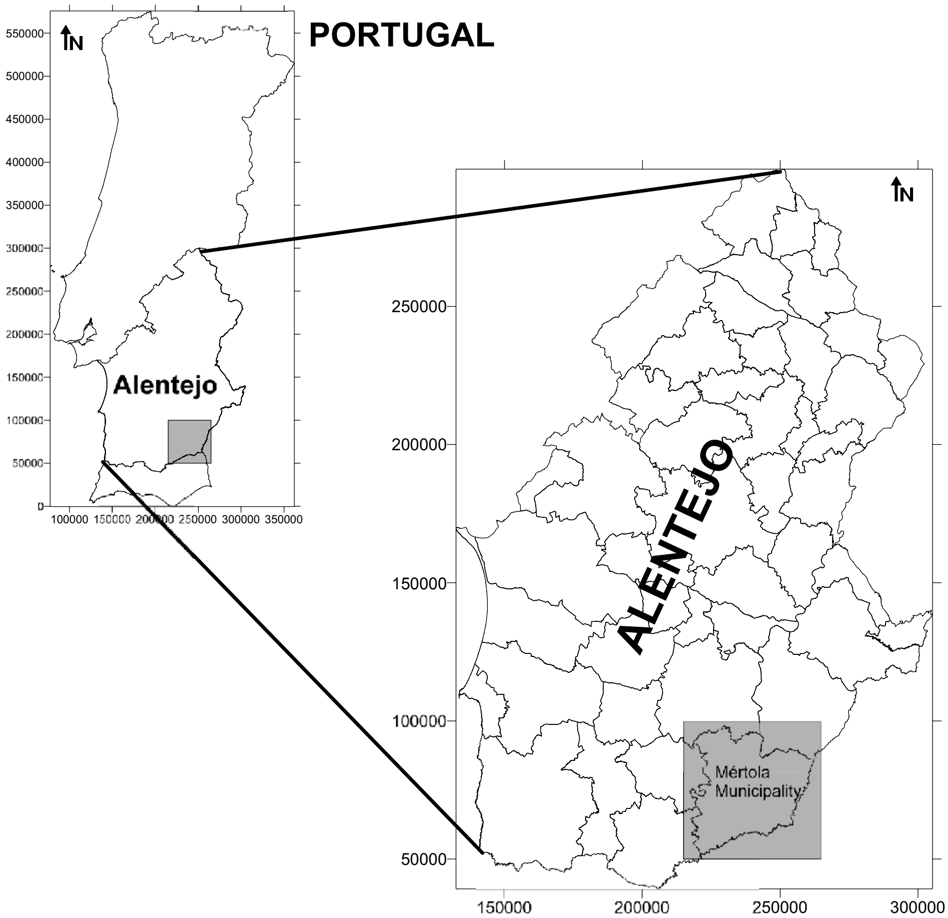

Figure 1. Position of Mértola municipality and Alentejo region in Portugal.

The SPZ geology is the result of a collision between two continents, with the closure of a palaeo-ocean during the hercynian orogeny. The SPZ seems to be an accrecionary prism that has evolved to an imbricate overthrust complex, with fault-strips of the oceanic sediments through the Precambrian substrate (Silva, 1989). The associated submarine volcanism gives rise to sulphide concentrations, represented by the Pyrite Belt, an area that correspond to the main pyrite mines both in Portugal and in Spain (Chambel et al., 1998). As seen in Figure 2, the SPZ is divided in three main domains:

- the Pulo do Lobo Sub-Zone, in the northern part
- the Volcano-Sedimentary Complex
- the Baixo Alentejo Flysch Group, in the southern part

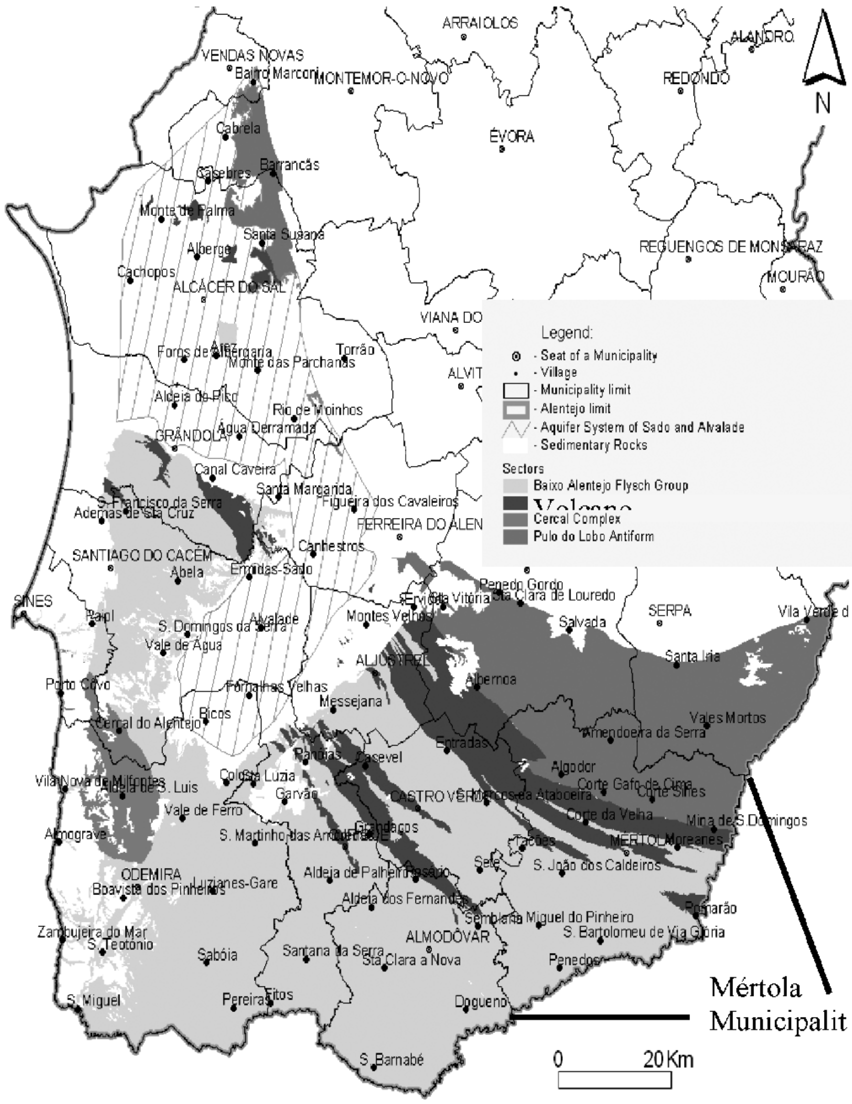

Figure 2. Geological setting of the South Portuguese Zone (SZP), in South Portugal. The legend is only representative of the area east and south of the line linking Vendas Novas to Vila Verde (de Ficalho).

The northernmost domain is Pulo do Lobo Sub-Zone, an anticlinorium that is the oldest of all the units, consisting of phylites, quartzites and some layers of acid and basic volcanic rocks.

The Volcano-Sedimentary Complex consists of original sedimentary and volcanic rocks containing massive sulphides. It is composed of alternating decimetric to metric layers of metamorphised acid and basic volcanic rocks, shales, graywackes, quartzowackes, siltstones, pellites, quartzites, sandstones, rare conglomerates and limestones and other types of rocks (Chambel and Almeida, 2000).

South of the Volcano-Sedimentary Complex are the most recent sediments of the SPZ, a turbiditic sequence corresponding to the Baixo Alentejo Flysch Group. The northernmost sub-division of this last Group is the Mértola Formation, practically the only one present in Mértola Municipality, followed by the Mira and Brejeira formations, more to SW (Oliveira, 1988; Silva, 1989).The Pyrite Belt corresponds to a large region that comprehends all the outcrops of the Volcano-Sedimentary Complex and the rocks that are in between, even if they are part of other domains. This is the area of the pyrite mines and also the area where the actual mineral prospection takes place.

3. Hydrogeology

Hydrogeologically the SPZ consists of hard rock aquifers. The low permeability rocks such as schists, phylits, greywackes, metavolcanic rocks, among others, associated with thin alteration layers are responsible for the low aquifer yields, usually less than 1 L/s. But there are some exceptions, especially in an optimal structural context, where some high fracturing is associated with more competent rocks, namely quartzites and greywackes. In such cases, yields can reach more than 5 L/s. Behind quantity, quality is also a problem in SPZ. Here the main issue is the high groundwater mineralization. The electric conductivity (EC) is very high in some wells on special SPZ geo-structures, reaching values of more than 10000 μS/cm in some places.

In general, the samples collected in deep wells show concentrations 1.5 times higher then those collected in large wells and springs, exception for the chloride in the Volcano-Sedimentary Complex waters, where large wells and springs have a median value of 396 mg/L and the deep wells 292 mg/L (Chambel et al., 1999). This is probably due to great evapotranspiration ratios, which promote the concentration of salts resulting from leaching of rocks that have high contents in salts retained during underwater rock formation (Chambel and Almeida 1998; Chambel, 1999).

The water samples of the Volcano-Sedimentary Complex and part of the Pulo do Lobo Anticlinorium are more mineralised than those from the other geological formations as can be seen in Figures 3 and 4, by the EC water values. The explanation must be in the high degree of fracturation observed in both of them, which results from the occurrence of more competent rocks, when compared with the more ductile rocks of the Baixo Alentejo Flysch Group (Chambel and Almeida 1998; Chambel, 1999). This would permit the ascension of deep mineralized water, with chloride and sodium contents as the main ions. This is confirmed by researches in the pyrite mine of Neves-Corvo, some kilometres west of Mértola Municipality (Fernández-Rubio et al., 1988; Fernández-Rubio and Carvalho 1993; Fernández-Rubio et al., 1994), where the works go today at a deep near 700 m and where it was possible to detect the increase of mineralization with deepness, which corresponds to about 475 µS/cm in EC by each 100 m deep, 150 mg/l of chloride and 140 mg/l of sodium by the same 100 m deep.

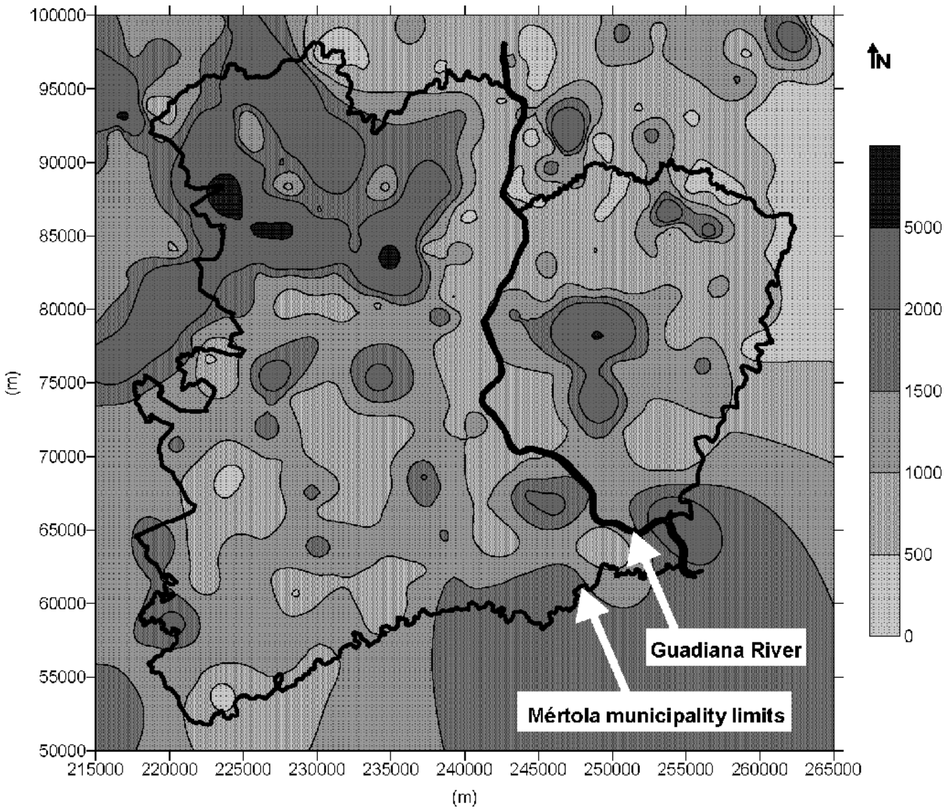

Figure 3. EC values for the groundwater of Mértola Municipality, in µS/cm.

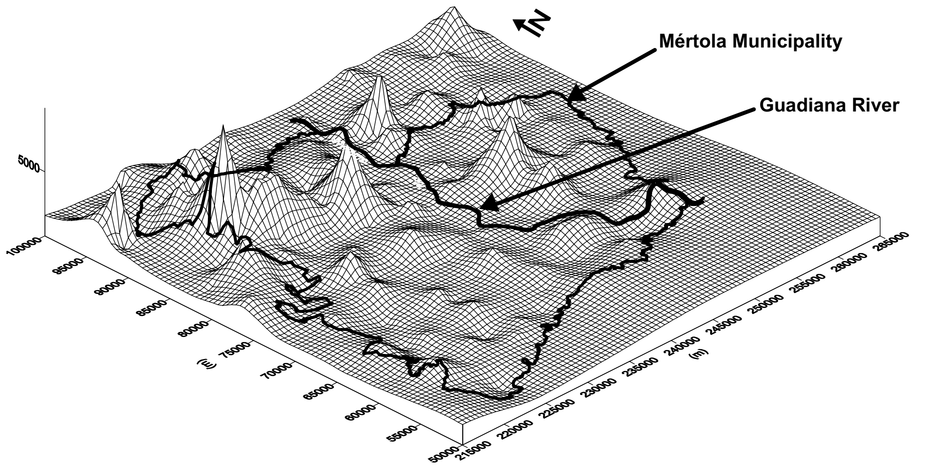

Figure 4. Three-dimensional representation of the groundwater EC values of Mértola Municipality, in µS/cm.

For the three groups of the SPZ on the area of Mértola, the waters of Pulo do Lobo Anticlinorium are clearly sodium or magnesium chloride type. Some of the salty waters occur inside this group, normally near the Volcano-Sedimentary Complex.

The waters of the Volcano-Sedimentary Complex are basically sodium chloride type, but some of them present some bicarbonate tendency. On the south part, on the Flysch Formation of Mértola of the Baixo Alentejo Flysch Group the waters are more sodium and magnesium bicarbonate type, but the sodium chloride waters continue to be present with some regularity. Rare are the calcium bicarbonate waters. Even so, these are the less mineralized waters of all the three groups.

The sodium and chloride tendency is clearer on the more mineralized waters, namely on the NW part of the municipality. The groundwater mineralization near the Guadiana River, on the places where the river crosses the Volcano-Sedimentary Complex, is much lower than in the flat areas, what is probably due to the higher hydraulic gradient on the vicinity of the river. Here, the quicker flow will drain the groundwater more rapidly and the evapo-transpiration processes are not so effective to sustain an increase on the mineralization.

4. Ecosystem Association

The existing plants and trees are adapted to this semi-arid environment, where specific species resist to these conditions, as the *Quercus suber*, the cork oak and the *Quercus ilex* trees. Investigation is now directed to the determination of specific flora associated with this very special kind of hydrologic environments. In fact, these mineralized waters are only present in certain areas and they seem to control the presence of salts in the soils, the natural vegetation and the behaviour of both animals and humans. The local of study was the NW part of Mértola Municipality, a place where the water mineralization is higher, as can be observed in the Figure 3 and 4. The Figure 5 shows the studied area.

Other of the consequences for the environment is the lost of soil capacity induced by humans when they use these kind of waters to irrigate. Due to the high ratio of evapotranspiration, the soils became highly salines and, after some years, they can't continue to produce, inducing desertification. With the desertification phenomena, the loose of soil is higher, and great part of South Alentejo are loosing soil till 1 cm each year, due to farming practices and soil depletion by concentrate rains, as shown in studies by Rosa (1980), Galvão (1982) and Brum Ferreira et al., (1993). In soils with less than half a metre in many places, some of these areas are now practically depleted of productive soils.

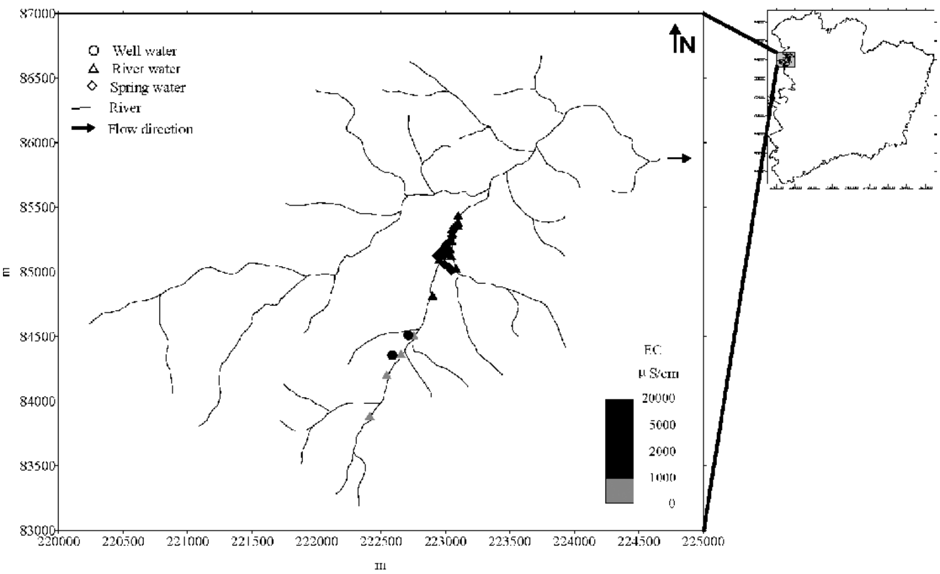

Figure 5. Selected area for the study of the ecosystems, showing EC interval values measured on the river, wells and spring waters, on 27 and 28 February 1995, after about 15 days without rain, showing a heavy component of the groundwater discharge to the river.

Comparing with the local flora, some special kind of plants occurs only in these special places where mineralized groundwater is present. This is the case of the *Juncus acutus* and *Juncus subulatus* (Figure 6), *Hordeum geniculatum* (Figure 7), and *Parapholis incurva* (Figure 8), which were identified by fieldwork in plain or depressed areas where the groundwaters have EC values upper than 3,000-4,000 µS/cm. The waters are basically Na-Cl type, as the previous investigation had detected. These waters can go up more that 10,000 or even 15,000 µS/cm, and the superficial waters in the areas near the springs rarely have less then 2,000 µS/cm, as it can be seen by the measurements on the area of investigation (Figure 5), where the photos 6, 7 and 8 where taken.

Figure 6. Juncus acutus and *Juncus subulatus*, plants that is present on this kind of environments.

Figure 7. Hordeum geniculatum, plant present on these ecosystems.

Figure 8. Parapholis incurve, other plant present on these ecosystems.

Also the behaviour of both humans and animals seems to be controlled by the water quality. The human settlements tend to avoid those areas, but the few people that live there, namely the shepherds, know exactly where the sheeps go to drink (they like the water with EC between 2,000 and 4,000 μS/cm). The shepard himself only drinks water from the rare wells with less than 2,000 μS/cm, and, for waters with more than 4,000 μS/cm, nor the shepherds nor the sheeps drink it. Attending this kind of behaviour, probably all the fauna is also controlled by the water quality. No agriculture is possible in these areas and even the cork trees are not present.

For the moment only this first species where identified, because the work was only done during summer of 2005, when the dry season don't permit to observe most part of the plants. The springtime will bring new light to the eco-system association in the area and the investigation will be completed.

5. Conclusions

Some species of plants are clearly related with the groundwater quality. In the south part of Portugal, on a geologic special environment called the Pyrite Belt, a strip on the Portuguese geology known for the presence of pyrite mines in a volcano-sedimentary complex, an association of plants normally found near shore is present. In effect, the groundwater in this area has some mineralization, namely sodium and chloride, responsible by the presence of these plants.

In these places the quality of the waters induces high contents of salts in the soils. Together with the use of mineralized water in agriculture, this induces increasing desertification processes, which results on soil depletion by erosion.

For the moment only some species where identified, because the work was only done during summer of 2005, when the dry season don't permit to observe most part of the plants. The springtime will bring new light to the ecosystem association in the area.

The special plants that were identified for the moment are the *Juncus acutus*, *Juncus subulatus*, *Hordeum geniculatum* and *Parapholis incurva* and are related with plain or depressed areas where the waters have EC values upper than 3,000-4,000 μS/cm, basically Na-Cl type. These waters can go up to more that 10,000 or even 15,000 μS/cm, and the superficial waters in the near areas rarely have less then 2,000 μS/cm.

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ASSESSMENT OF VULNERABILITY OF WATER RESOURCES TO CLIMATE CHANGE: ECOHYDROLOGICAL IMPLICATIONS

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Abstract: In this paper, the authors discusses the plausible impacts of climate changes on water resources with emphasis on ecohydrological implications. For this purpose, vulnerability of water resources to climate change was discussed. Available evidences indicate that regional changes in climate, particularly increases in temperature, have already affected a diverse set of physical and biological systems in many parts of the world. Based on the fact that water resources are an integral part of the global hydrologic cycle, they are considered among the most vulnerable natural systems to climate changes. Research since 1996 indicate that severe problems related to water will affect the globe around 2025 which will be intensifying to attain its peak by the year 2100. Undeveloped/developing countries where semi-arid climate prevails and whose water resources are not properly developed will be affected most severely from climate changes. An accurate impact assessment first necessitates analyses of parameters for their vulnerability to climate change for each system. This is achieved by construction of a conceptual hydrogeological model which is then transferred to mathematical model of the water resources system.

Keywords: climate change; ecohydrology; groundwater; recharge; vulnerability; water resources

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1. Introduction

If two of the major challenges that the word is to face in the near future were to be listed, population growth and global climate change would take place at the top of the list. This is, because, either population growth or climate change put direct stress on the water resources that are available for life. Water is essential for life not only because it assures survival of human being of assuring their drinking need and food security, but also due to the fact that it is essential to maintain water dependent ecosystems. The stress put on water resources by population is apparent and beyond the scope of this paper. However, the changes in climatic conditions may alter the hydrological cycle in even a basin scale which in turn, may affect the groundwater recharge regime.

The Working Group I of the Intergovernmental Panel on Climate Change (IPCC) has released a report on assessment the observed changes in climate, their causes, and otential future changes (IPCC, 2001). The report concludes that the globally averaged surface temperatures have increased by 0.6 ± 0.2ºC over the 20th century; and that for the range of scenarios developed in the IPCC Special Report on Emission Scenarios (SRES), "the globally averaged surface air temperature is projected by models to warm 1.4 to 5.8ºC by 2001 relative to 1990 and globally averaged sea level is projected to rise 0.09 to 0.88 m by 2100" (IPCC, 2001). The predicted climate changes are also evaluated by the Panel in terms of their impacts on water resources for the coming 100 years. The impacts are tabulated in Figure 1 indicating the confidence level of prediction.

The ecosystems, on the other hand are known to rely on subsystems such as the carbon cycle, temperature and soil moisture (UNEP, 2002). Any change in these sub systems may be reflected as a change in water-use efficiency which may consequently cause changes in plant productivity. Similarly, impacts on the wildlife are expected to result in changes in elevational movement, abundance, body size as well as shifts in breeding time etc. Although its magnitude and direction are not known, it is apparent that the future carbon storage in forests will be altered by climate change. Above all, lakes, rivers and inland wetlands which have important role in maintaining biological diversity are all water-dependent systems. Therefore, changes in climate may alter the habitat for many plant and animal species including those endemic or endangered.

2. Climate Change and Water Resources

Based on the facts that outlined above, water resources availability and quality and climate change are interlinked. However, assessments of impacts of climate change on water resources are generally documented in the literature on an approximate basis. This is, to great extent due to the fact that each natural hydrogeological system has its own uniqueness in terms of factors governing the occurrence and movement of water. Regarding the type of the water resources system, the effective parameters may change not only by category but also by degree.

Firstly, the response of water resources to climate change is controlled by the residence (or turnover) time of the system. As depicted in Figure 2, the residence time may range from days in small stream watershed to thousand years in deep confined continental aquifers. Secondly, total annual precipitation and although connected to the total precipitation more importantly the effective precipitation are the foremost leading parameters in water resources. Because, the effective precipitation is directly affected by climate change and directly affects the recharge of the water resources. Apparently, not only the reduction in precipitation but also its spatial and temporal distribution will have adverse effects on the availability and quality of the water resources. In addition to the climatic conditions, effective precipitation is controlled by the hydrogeological framework of the concerned system.

	2025	2050	2100
Water supply	Peak river flow shifts from spring toward winter in basins where snowfall is an important source of water (high confidence).	Water supply decreased in many water-stressed countries, increased in some other waterstressed countries (high confidence).	Water supply effects amplified (high confidenced). (high confidence).
Water quality	Water quality degraded by higher temperatures. Water quality changes modified by changes in water flow volume. Increase in saltwater intrusion into coastal aquifers due to sea-level rise (medium confidence).	Water quality degraded by higher temperatures (high confidence). Water quality changes modified by changes in water flow volume (high confidence).	Water quality effects amplified (high confidence).
Water demand	Water demand for irrigation will respond to changes in climate; higher temperatures will tend to increase demand (high confidence).	Water demand effects amplified (high confidence).	Water demand effects amplified (high confidence).
Extreme events	Increased flood damage due to more intense precipitation events (high confidence). Increased drought frequency (high confidence).	Further increase in flood damage (high confidence). Further increase in drought events and their impacts.	Flood damage several-fold higher than "no climate change scenarios."

Figure 1. Water resource effects of climate change-if no climate policy interventions are made (Judgments of confidence use the following scale: very high (95% or greater), high (67–95%), medium (33–67%), low (5–33%), and very low (5% or less)), (IPCC, 2001).

In order to establish strategies to cope with the expected problems in water-dependent systems it is essential first to define the problem in terms of vulnerability of water resources to climate changes. This requires a thorough understanding of the type and extent of the relationship between climate and water resources systems which are connected through the hydrological cycle. Water resources systems are constituted of two main part: hydrology (dynamic part) and the geological-physiographical configuration (static part). This approach is schematized in Figure 3. The term "vulnerability" here is used to define the 'extent to which the water resources system is susceptible to sustaining damage from climate change' following the definition by Intergorvernmental Panel for Climate Change. Thus, *vulnerability* differs from *sensitivity* which is defined as 'the degree to which a water resources system will respond to a given change in climate, including beneficial and harmful effects'. Apparently, vulnerability is a function of sensitivity (IPCC, 2001).

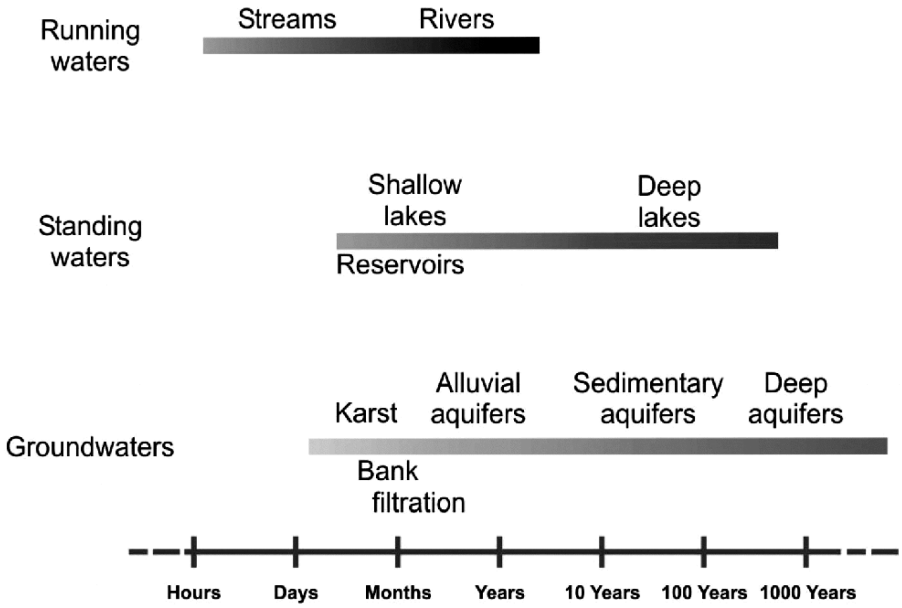

Figure 2. Residence times of water resources systems (from Chapman, 1992).

The hydrologic cycle states that precipitation originates as evaporation from land and the oceans. Soil moisture is used by plants, which return more moisture to the atmosphere. Water that does not evaporate or transpire or seep into subsurface runs off to form streams and rivers. Snow stored in winter in the mountains provides water for rivers and deltas in the spring and summer. Groundwater constitutes one portion of the hydrologic cycle. Water seeps into water-bearing formations known as aquifers that act as conduits for transmission and as reservoirs for storage of water. Among other climatic components of the hydrologic cycle, precipitation and evapo-transpiration are the two major parameters controlling the percentage of the water that runs off over the land (surface waters) and that seeps into aquifers (groundwater). However, knowledge of climatic change alone is not adequate for assessment of the impact on the water resources. Water resources may respond quite differently to the same climatic change due to their different hydrogeological framework. Therefore, thorough knowledge of the hydrogeological setting is essential in assessing the impact of climate change on water resources. This requires the definition of the hydrodynamic system and consequently the quantification of the vulnerability of the factors governing the occurrence and movement of the water in the system. Regarding the factors having role in the occurrence and movement of the water, water resources are classified as shown in Figure 4.

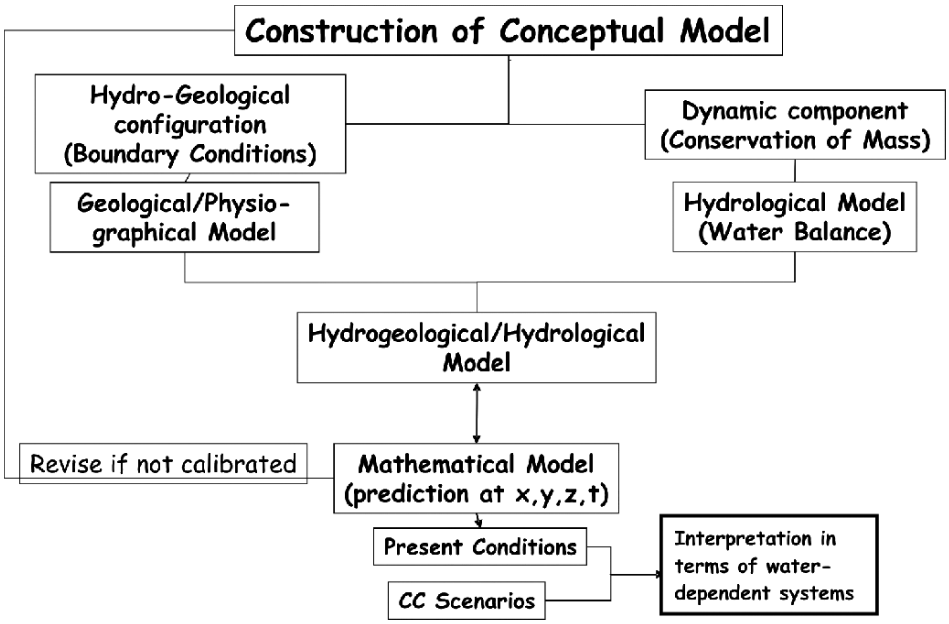

Figure 3. Methodology Applied in the Study of Assessment of Vulnerability of Water Resources Systems to Climate Change (Ekmekci et al., 2004).

3. Vulnerability of Groundwater Resources to Climate Change

The classification given in Figure 4 implies that the hydrogeological structure of water resources systems should be evaluated in terms of their role in capturing and storing capability of the precipitation. Recharge mechanism is thus of major importance. This classification also implies that the relative residence or turnover time of the water resources is important in assessing the impact of change in recharge regime. Residence time of water in the system indicates to a certain extent, the time lag between the response of the system to the changes in the recharge conditions. Apparently, water resources of shorter residence time are more vulnerable to any change in the climatic conditions.

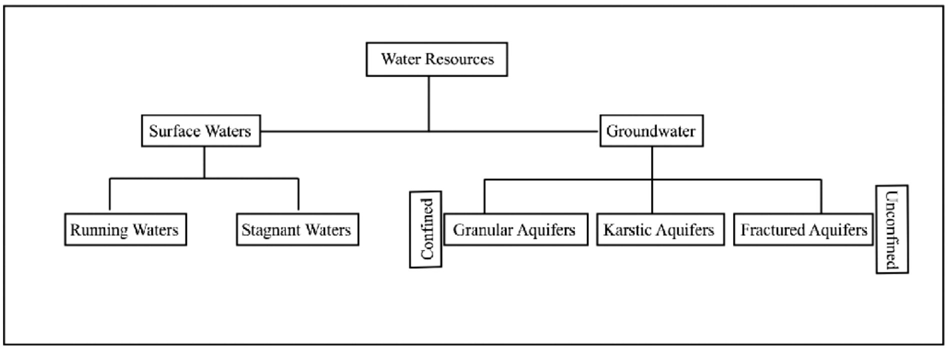

Figure 4. Classification of water resources according to the water occurrence and movement (Ekmekc et al., 2004).

Keeping in mind that each water resource has its own unique hydrogeological structure, parameters making the system more vulnerable may differ for each different system. Therefore, an accurate impact assessment first necessitates analyses of parameters for their vulnerability to climate change for each system. This is achieved by construction of a conceptual hydrogeological model which is then transferred to mathematical model of the water resources system (Tezcan et al., 2004). Once the mathematical model is calibrated for the prevailing conditions, it is possible to test every parameter used in describing the system for its response to change in recharge regime.

The recharge regime is closely related to the meteorological conditions such as the type and total amount of precipitation, spatial and temporal variation of precipitation, temperature and evapotranspiration. Recharge of water resources occurs when the water entering to the system (gain) exceeds the water leaving the system (loss). When the loss exceeds the gain, there will be no beneficial water. The major source of loss is the evaporation. Thus, the difference between precipitation and evapotranspiration (a function of temperature) can be regarded as the potential recharge and named as effective precipitation. Change in the temporal variation in precipitation and temperature due to climate change is then reflected in the effective precipitation. The ultimate consequence is the recharge of the water resources.

The effective precipitation has a major role in the process of making the water resources potential. The effective precipitation on the other hand, is sensitive to the changes in the magnitude, intensity and period of precipitation as discussed above. Considering these three principal factors, the impact of climate change on the effective precipitation can be evaluated together in the way shown in Figure 5. Effective precipitation is extremely low when a low total precipitation (magnitude) occurs in hot periods and in very short time (high intensity). On the contrary, when high amount of precipitation occurs in colder periods with an even temporal variation (low intensity), the effective precipitation is very high.

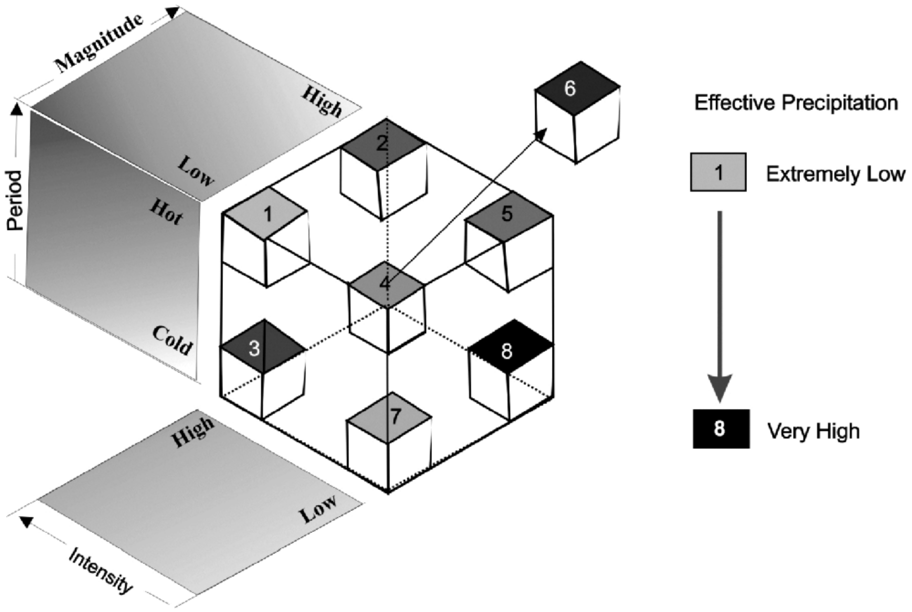

Figure 5. Principal factors controlling effective precipitation (Ekmekci et al., 2004).

Vulnerability of groundwater systems then depend upon the hydrogeological setting. In the former case, even if the setting favors high infiltration, recharge will occur in minor rate. To define the vulnerability of groundwater systems, another principal factor can be defined and related as depicted in Figure 6: effective precipitation, type of aquifer and turnover time. According to this picture, the unconfined aquifers with short turnover time are extremely sensitive to climate change and extremely vulnerable when this change reduces the effective precipitation. On the other hand confined aquifers with long residence time are much less sensitive to climate change and therefore their vulnerability is very low.

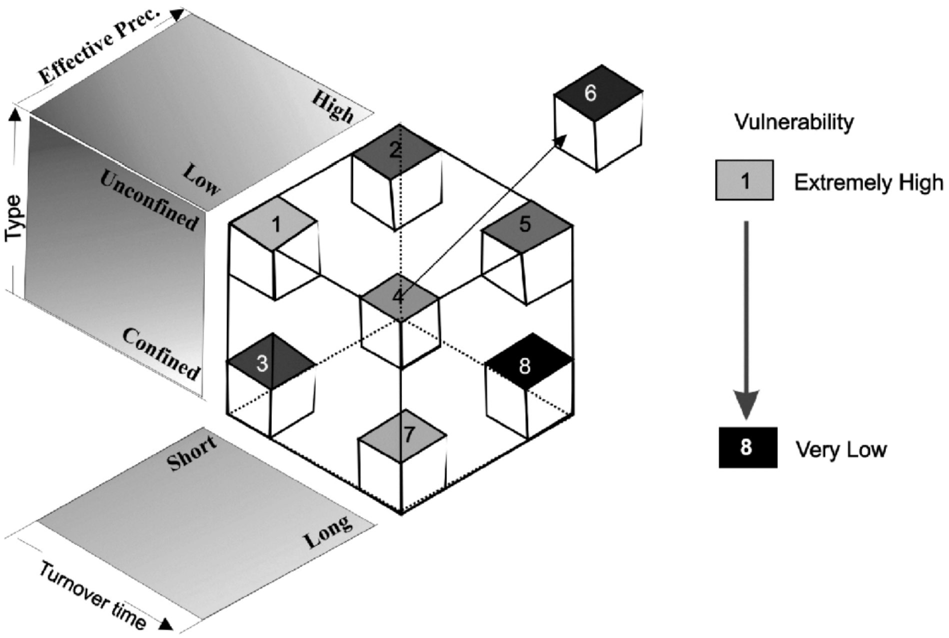

Figure 6. Principal factors controlling vulnerability of groundwater systems to climate change (Ekmekci et al., 2004).

4. Impacts of Climate Change on Basin Hydrology

The surface water resources require immediate attention in this respect although some specific characteristics of water resources may be more influential as they may affect the storage and discharge regime of the system. For surface water resources, size, shape, cover type, slope, drainage pattern and density of the basin and for the ground water resources, location and hydrological characteristics of the recharge area, type, depth and extent of the aquifer, hydraulic characteristics of the aquifer and the overlying vadose zone, boundary conditions of the aquifer are regarded as the important characteristics to be defined in vulnerability assessment.

When surface waters are concerned, it is more convenient to speak about reliability instead of vulnerability. *Reliability* refers to the 'manageability of the resource with low risk'. For instance, rivers with irregular flow regime are less reliable in this sense. It is apparent that, reliability of rivers then is dependent on the basin characteristics which are all combined in a single factor called here 'time of concentration', the temporal variation of precipitation and the total amount (magnitude) of precipitation. Figure 7 shows the reliability of river basins related to climate change.

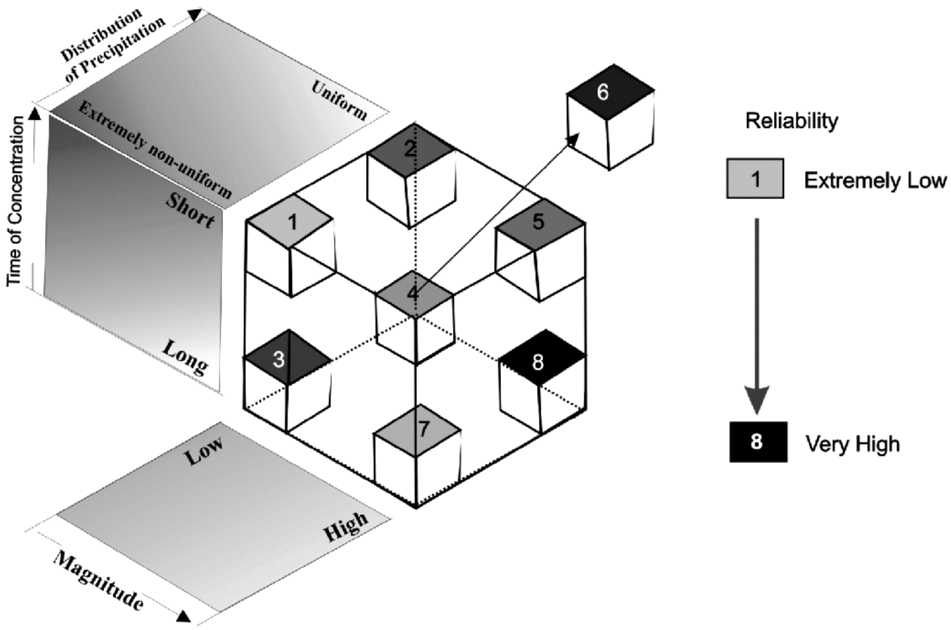

Figure 7. Principal factors controlling reliability of surface waters to climate change (Ekmekci et al., 2004).

5. Ecohydrology and Climate Change

The success of adaptation of ecosystems to climate change is dependent on the speed and the magnitude of the change. When the rate of climate change exceeds well the rate of resilience, the ecosystem could be seriously affected or even destroyed.

Ecohydrology, defined simply as the study of the interaction between the hydrological cycle and ecosystems, requires a thorough understanding of the response of the components of these systems to hydrological changes (Zalewski, M., 2002). The hydrologic cycle defines the dynamics of the water cycling between atmosphere, lithosphere and biosphere. From ecohydrological perspective, the “soil system” describing both the surface and subsurface hydrogeological environments, plays a crucial role in the dynamics between climate, soil and vegetation (Rodriguez-Iturbe, 2000). The soil moisture balance equation which is the fundamental equation in hydrology, explicitly quantifies this dynamics. Two variables of the balance equation, precipitation and evaporation incorporate climatic elements such as temperature, radiation, vapor pressure, humidity etc. Hydrogeological characteristics of the soil and underlying permeable material control the availability and storage of water.

Vegetation on the other hand has important influence on runoff, streamflow and effective precipitation due to interception as well as its major role in transpiration. This is perhaps the most challenging part of the mass balance equation. Because it requires a sound understanding of how plants affect runoff, stremflow, infiltration into soil and deep percolation to recharge the groundwater. Essential to ecosystems, soil erosion and water and nutrient interaction is also to great extent controlled by plants. Keeping in mind that all the interactions defined above are bi-directional, any change in any element of the climate-soil-vegetation system causes changes in the others, and assessment of this change is essential for a sustainable water resources management strategy.

6. Needs for Future Research

In the 2001 report of ICPP, the related working group also documented the estimated impacts on various ecosystems. These estimation were projected for the years 2025, 2050 and 2100 (Figure 8). However, these estimates are of geneal character and far from being specific. Therefore, it is essential to conduct research on specific sites on basin scale, first to assess the vulnerability of water resources n the basin of interest. Once vulnerability of water resources to climate change is assessed, the reactions of habitats and consequently the ecosystems to these changes should be estimated. This requires the essential knowledge of ecohydrological variability of ecosystems. The next step is then to establish a true interdisciplinary study highlighting the ecohydrological approach, to adequately address the questions related to the space-time links between climate-soil-water-landscape and vegetation. Response, resilience and adaptive capacity of different types of ecosystems in micro-scale habitats are among the topics of future research. Because the ultimate objective is to predict the future response of any system to any change, all sub-systems should be defined in terms of mathematical models that are capable to define the interactions among systems and their environments on at least a reasonable uncertainity.

	2025	2050	2100
Corals	Increase in frequency of coral bleaching and death of corals (high confidence).	More extensive coral bleaching and death (high confidence).	More extensive coral bleaching and death (high confidence). Reduced species biodiversity and fish yields from reefs (medium confidence).
Coastal wetlands and shorelines	Loss of some coastal wetlands due to sea-level rise (medium confidence). Increased erosion of shorelines (medium confidence).	More extensive loss of coastal wetlands (medium confidence). Further erosion of shorelines (medium confidence).	Further loss of coastal wetlands (medium confidence). Further erosion of shorelines (medium confidence).
Terrestrial ecosystems	Lengthening of growing season in mid- and high latitudes; shifts in ranges of plant and animal species (high confidence). Increase in net primary productivity of many mid- and high-latitude forests (medium confidence). Increase in frequency of ecosystem disturbance by fire and insect pests (high confidence).	Extinction of some endangered species; many others pushed closer to extinction (high confidence). Increase in net primary productivity may or may not continue. Increase in frequency of ecosystem disturbance by fire and insect pests (high confidence).	Loss of unique habitats and their endemic species (e.g., vegetation of Cape region of South Africa and some cloud forests) (medium confidence). Increase in frequency of ecosystem disturbance by fire and insect pests (high confidence).
Ice environments	Retreat of glaciers, decreased sea-ice extent, thawing of some permafrost, longer icefree seasons on rivers and lakes (high confidence).	Extensive Arctic sea-ice reduction, benefiting shipping but harming wildlife (e.g., seals, polar bears, walrus) (medium confidence). Ground subsidence leading to infrastructure damage (high confidence).	Substantial loss of ice volume from glaciers, particularly tropical glaciers (high confidence).

Figure 8. Ecosystems effects of climate change-if no climate policy interventions are made (Judgments of confidence use the following scale: very high (95% or greater), high (67–95%), medium (33–67%), low (5–33%), and very low (5% or less)), (IPCC, 2001).

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PREDICTING PROBABLE EFFECTS OF URBANIZATION ON FUTURE ECOLOGICAL INTEGRITY IN THE UPPER ILLINOIS RIVER BASIN, USA

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Abstract: A study was undertaken to predict the probable effects that future urbanization may have on ecological integrity in the Upper Illinois River Basin (Chicago area), USA. Biotic indices and sediment trace-element concentrations for 43 streams, determined by Illinois State agencies and as part of the U.S. Geological Survey's National Water Quality Assessment program, were examined along an agricultural-to-urban land-use gradient. The relations found among biotic integrity, sediment chemistry, and urbanization were associated with annual samples collected from 1982 through 1993. Because these annual samples were from different tributary basins with different urban percentages and geologic settings, the trends along the gradient suggest the absence of bias. Analytical equations were fit to bivariate relations, and probability density functions fit to residuals for use with the Monte Carlo technique so that stochastic modeling could be performed. Stability of stochastic modeling required 1,500 Monte Carlo trials; reliability of stochastic modeling was evaluated by comparing statistical summaries of measured to simulated biotic indices, and future predictions approximately validated against an independent AIBI score for Long Run Creek. Stochastic modeling of future urbanization-induced changes in ecological integrity for basins (Big Rock Creek, Des Plaines River, Mill Creek, and Flag Creek) along an urban gradient (1990 percent urban land use of 1, 5, 10, and 87 percent) resulted in a broad range of probable biotic resource quality (excellent to very poor). Predictors used to simulate changes in basin ecological integrity from 1990 to 2000 and 2000 to 2010 included fish and invertebrate biotic indices, and streambed sediment nickel concentration.

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Using these predictors, the degradation of ecological integrity in tributary basins occurred at differential rates and with a probable distribution of likely outcomes. For example, the AIBI median predictions of ecological integrity from 1990 and 2010 was 2 quality classes (good to poor) in the Big Rock Creek and Des Plaines tributary basins, and 1 quality class (poor to very poor) in the Mill Creek and Flag Creek tributary basins. A scale was devised for converting MBI scores to biotic resource quality classes for interchanging results with AIBI scores. This calibrated scale should be useful in more urbanized streams where it is not always possible to compute AIBI scores, and for comparison between biotic indices in other studies. Bed sediment nickel concentration was a useful predictor of ecological integrity and basin percent urban land use (and population density). Because the time and costs for determining nickel concentrations are much less than for determining biotic integrity scores, future studies could use this scale or other correlated variables as predictors.

Keywords: bed sediment; biotic indices; biotic integrity, ecological integrity; Monte Carlo technique; probable effects; stochastic modeling; uncertainty; urbanization; water quality

1. Introduction

The Chicago metropolitan area, one of the largest urban areas in the United States, is located in the Upper Illinois River Basin (UIRB). Because the plains surrounding Chicago contain some of the richest farmland in the world, an important concern is the conversion of agricultural lands to urban lands (called urbanization) and their affects on ecological (habitat and biotic) integrity in the urbanizing streams (Figure 1a). Most water-quality assessments of biotic health address either fish or macroinvertebrate community conditions (Booth and Reinelt, 1993; Dreher, 1997; Wang et al., 1997). In each case, multiple metrics are used to compute scores from which the degree of stream impairment is inferred (Table 1). For example, the respective index of biotic integrity (IBI), or modified version called the alternate index of biotic integrity (AIBI), and macroinvertebrate biotic index (MBI) represent composite scores based on a number of fish and macroinvetebrate community metrics (Karr et al., 1986; Bertrand et al., 1996). In general, the higher IBI (or AIBI) scores and lower MBI scores indicate better biotic integrity stream and therefore steam quality.

Table 1. Relation between biotic indices and resource classification [AIBI - alternative index of biotic integrity; MBI - macroinvertebrate biotic index]

AIBI score[1]	Resource quality[1]	Original MBI score[1]	Original MBI resource quality[1]	Modified MBI score[2]	Modified resource quality[2]
>50	Excellent	<5.0	Excellent	<5.0	Excellent
41 to 50	Good	5.0 to 5.9	Good	5.0 to 5.49	Good
31 to 40	Fair	6.0 to 7.5	Fair	5.5 to 5.99	Fair
21 to 30	Poor	7.6 to 8.9	Fair	6.0 to 6.49	Poor
<20	Very poor	>8.9	Poor	>6.5	Very poor

[1]Biotic integrity scores and resource quality classes defined by *Bertrand et al., 1996*

[2]Modifed biotic scores and resource quality classes determined by calibration in this study

There are several issues that may confound the use and interpretation of biotic indices for the evaluation of ecological integrity in urbanizing streams. *First*, the loss of habitat and biological integrity are complex and not well understood. Possible causes for diminished ecological integrity in urbanizing streams may include changes in quantity and(or) chemical composition of both surface and ground water. Some changes in water quantity that may occur in response to urbanization include increased runoff volume, flood frequency, and peak storm flows; and reduced groundwater recharge and baseflow contributions to streams. Some changes in water chemistry that may occur in response to urbanization include changes in applied chemicals; increased nonpoint pollution of pathogens, nutrients, metals, and sediment; and changes in temperature, oxygen content, and redox conditions. *Second*, studies by Seegert (2000), Houston et al. (2002), and Illiopoulou-Georgudaki et al. (2003) demonstrate the need for IBI calibration due to variation among IBI assessments. For example, variations in stream quality differ by up to three biotic resource quality classes for the same data set. *Third*, it is not always possible to compute an IBI (or AIBI) in more urbanized streams; consequently the use of alternate predictors of ecological integrity is important. One suggestion is to use the MBI and IBI interchangeably (Carroll and Jackson, 2005); however, using these biotic indices interchangeably requires some form of calibration to show the same stream quality. Presently, there are no published studies of calibrated biotic indices, and few studies exist that evaluate both fish and macroinvertebrate aquatic communities (Fitzpatrick, 2005).

According to Carroll and Jackson (2005), studies that further investigate the interconnections between biotic indices and other variables will be essential to developing effective future biomonitoring surveys. One means of exploring these relations is to use a gradient approach. A gradient approach is oftentimes used to analyze the effects of urbanization on individual aquatic communities (Booth and Reinelt, 1993; Dreher, 1997; Wang et al., 1997). In using a gradient approach, the measurement (biological, physical, chemical) responses are organized for analysis along a continuum of land uses; for example, agricultural to urban land use. Fitzpatrick et al. (2005) recently analyzed a subset of biotic community data from Wang et al. (1997) and bed sediment trace-element data collected as part of the United States Geological Survey's (USGS) National Water Quality Assessment (NAWQA) (Fitzpatrick et al., 1998) along two gradient measures of urbanization: population density and percent urban land use. In that study, the comparatively large variability and general trends in fish, invertebrate, and habitat indices, agreed with earlier studies in their relation to these measures of urbanization. No attempt was made to evaluate relations between AIBI and MBI indices and whether fish and macroinvertebrates communities respond similarly to urban stressors. In some gradient studies, however, attempts were made to identify thresholds of stream impairment based on trends in the data. Unfortunately, the comparatively large variability and nonlinear nature of these trends render it difficult to delineate specific thresholds of stream impairment (Fitzpatrick, et al., 2005).

This paper describes a stochastic modeling approach and results of simulated urban growth on future ecological integrity in tributary basins of the Upper Illinois River Basin (URIB). The use of a stochastic model is more appropriate than a deterministic model to provide simulated estimates of future ecological integrity because of the variability in biotic indices and streambed concentrations, and uncertainty in the rate of urban growth. For this reason, the specific objectives in this report are to: (1) develop and compare various stochastic equations for predicting urbanization effects on ecological integrity in the UIRB; and (2) quantify the probable effects of future urbanization on ecological integrity for the Big Rock Creek, Des Plaines River, Mill Creek, Long Run Creek, and Flag Creek tributary basins that occur a land use gradient during the years: 2000 and 2010. Accomplishing these objectives will provide information for holistic basin management and assessment of policy effectiveness with respect to urbanization and ecological integrity. Important questions that can be addressed using a stochastic model include: (1) Do suitable predictors exist? (2) How broad is the predictive range? (3) Can biotic indices be calibrated to each other? (4) Which tributary basins are more likely to degrade in 2000 and 2010?

2. Methods

A stochastic modeling approach is used to quantify the effects that urbanization might have on ecological integrity in the Upper Illinois River Basin (Figure 1a). The basic modeling approach involves developing various models, performing model validation, and scenario modeling of urban growth in selected tributary basins (Friedel, 2004). The stochastic models used in this study represent numerical stochastic equations that are developed through nonlinear regression, residual analysis, and application of the Monte Carlo technique. The regression analysis is performed on biotic indices and trace-element concentration data computed based on UIRB field measurements and analysis as described in the next section.

2.1. STOCHASTIC MODEL

2.1.1. *Measurements and Data Analysis*

The wealth of historical data in the UIRB made it possible to examine biological and contaminants data along an urban land-use gradient. Indices of biotic integrity determined from fish and macroinvertebrate community measurements collected in the early to late 1980's by the Illinois Environmental Protection Agency (IEPA) (IEPA, 1987 and 1994), and bed sediment trace-element concentrations and population density (1990, 1995, and 2000) determined as part of the U.S. Geological Survey's (USGS) National Water Quality Assessment study (Fitzpatrick, et al., 1998), were available for about 43 tributary basin sites in the 28,328 km^2 UIRB. The tributary basin sites in the UIRB have drainage areas that range 15 to 318 km^2 and are located in three subbasins: Des Plaines, Fox, and Du Page River. The full range urban land-use conditions exist at the Des Plaines River subbasin sites. In contrast, most of the Fox River subbasin sites are in the range from 0 to 20 percent urban land use, whereas the Du Page River sites are in the range of 60 to 100 percent urban land use (Figure 1b). The UIRB tributary basin sites fall into four main ecoregions delineated by surficial geology: Wheaton Morainal Country, low permeability (Des Plaines and upper Du Page); Wheaton Morainal Country, moderate permeability (upper Fox); and lastly Bloomington Ridged Plain, low to moderate permeability (lower Fox and lower Du Page). Data analysis is undertaken to identify potential predictors and ensure that unbiased and independent measurements are used in developing the stochastic models. The high degree of correlation between certain variables suggests dependence and suitability as predictors (Table 2). For example, the relation between population density and percent urban land cover is almost perfect with a correlation coefficient of 0.96. This suggests that either one of these urban measures can be used as the explanatory variable when deriving predictive equations. The next greatest relation to these urban explanatory variables is associated with the trace metals. The correlation coefficients for metals and urban criteria range from 0.7 to 0.79 and about 0.7 for relation to the biotic indices (AIBI or MBI).

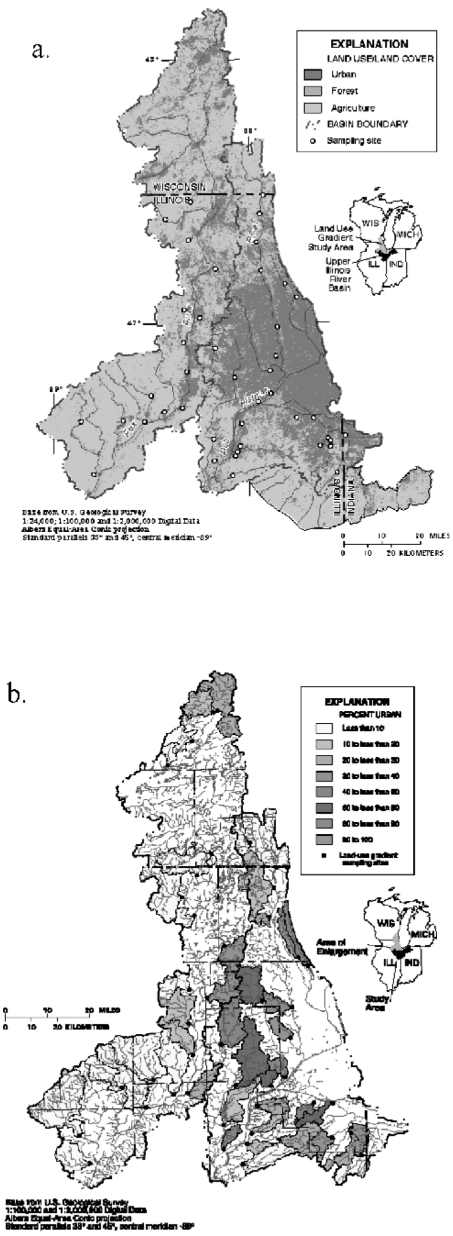

Figure 1. Upper Illinois River Basin: (a) land use, and (b) tributary basins along land use gradient.

Evaluating the relation of metals to the biotic criteria is important as the scores are associated with resource quality and therefore degree of stream impairment. The relation between the biotic indices and urban variables are slightly weaker with the absolute correlation coefficient values of about 0.64 to 0.68. Somewhat surprising is the weaker relation that exists between AIBI and MBI with a correlation coefficient value of −0.61. The comparatively weak relation between biotic indices may in part be a result of measurement noise or differences in time and space integrating abilities of fish and macroinvertebrates. The weakest relation among variables is between tributary drainage area and biotic integrity with a correlation coefficients of about 0.45. This very weak relation implies that the variability and (or) uncertainty is greater than other variables and for this reason will not be considered during model development.

Table 2. Correlation coefficients between certain urban measures, biotic indices, and bed sediment concentrations [AIBI - alternative index of biotic integrity; MBI - macroinvertebrate biotic index; Urb - percent urban land use, PopD - population density, people per square Kilometer Ni - nickel concentration, ppm in bed sediment, Cu - copper concentration, ppm in bed sediment, DA - drainage area of tributary basins]

	AIBI	MBI	Urb	PopD	Ni	Cu	Cr	DA
AIBI	1							
MBI	−0.61	1						
Urb	−0.68	0.65	1					
PopD	−0.74	0.64	0.96	1				
Ni	−0.71	0.65	0.79	0.76	1			
Cu	−0.55	0.70	0.67	0.61	0.86	1		
Cr	−0.59	0.70	0.73	0.71	0.89	0.94	1	
DA	0.45	−0.45	–	–	–	–	–	1

Because the biological data were collected during annual synoptic surveys (1982 to 1990) in different tributary basins and different ecoregions, these data were examined to ensure their independence and unbiasedness for use in stochastic modeling. For example, the annual biotic integrity scores (AIBI and MBI), plotted as a function of the 1990 percentage of urban land use (Figure 2), both depict comparatively weak trends. These trends indicate that the AIBI decreases and MBI increases with corresponding increases in percent urban land. The degree of weakness in these trends may be associated with what appears to be a shift (or bias) in the 1982 data and high degree of variability in the 1983 data. Whereas the removal of these data sets would reduce the variability and enhances trends in the biotic indices, the reasons for these aberrations were not apparent and therefore their removal could not be justified. The fact that these general trends reflect synoptic sampling that occurred in different basins and over a range of land use implies measurement independence. Likewise, no differences were seen in the response of biotic indexes to urban land use between the Fox and the Des Plaines River basins indicating that the insensitivity of fish and macroinvertebrates to ecoregions and therefore near-surface geology in the UIB (Figure 3).

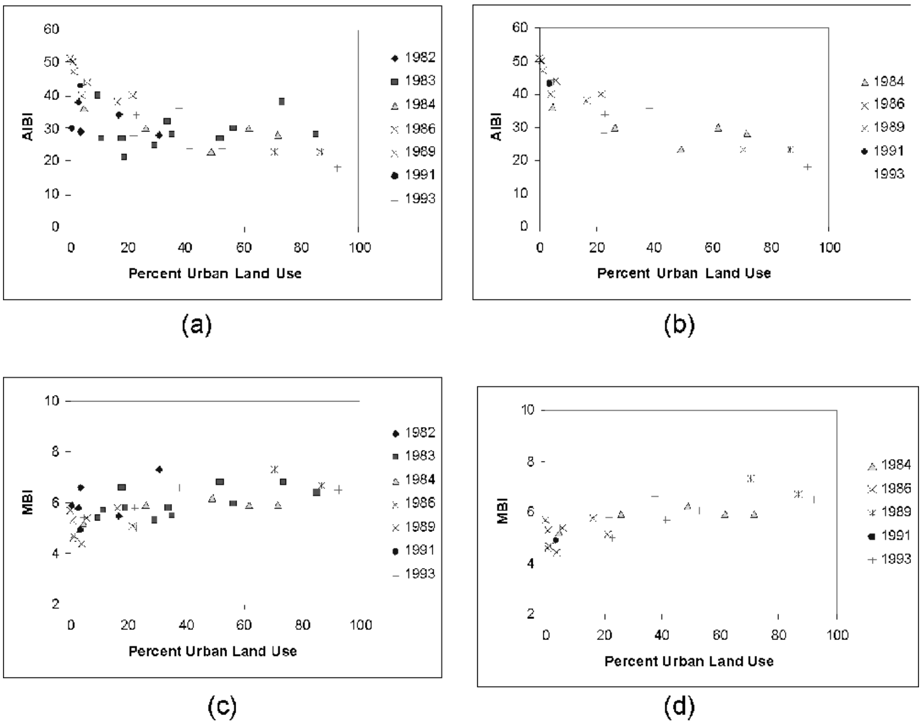

Figure 2. Annual biotic integrity scores: (a) fish AIBI (all years), (b) macroinvertebrate MBI (all years), (c) fish AIBI (1982 and 1983 removed), and (d) macroinvertebrate MBI (1982 and 1983 removed).

Priority pollutants (trace metals) in streambed sediments were explored for trends with respect to urbanization and biotic indices. In general, the concentration of trace metals in bed sediment increased with increasing urbanization. In principal component analysis (Fitzpatrick et al., 2005), the cadmium, chromium, copper, lead, mercury, and zinc concentrations grouped along a principal axis indicating their commonality in contributing to variability along that direction. In contrast, nickel concentrations were associated with an isolated principal axis indicating independence from the other metals. For this reason and the strong correlation between bed sediment nickel concentration with urbanization and biotic indices, and comparatively rapid and inexpensive measurements compared to fish and macroinvetebrate community assessments, nickel concentrations were evaluated with respect to urbanization (Figure 4a). Because a strong trend exists between bed sediment nickel concentrations and percent urban land use (similar to MBI and percent urban land use), bed sediment nickel concentrations could be used as an explanatory variable to predict biotic integrity scores and therefore degree of stream impairment. For this case, increasing nickel concentrations corresponded to increasing MBI (Figure 4b) and decreasing AIBI (Figure 4c) scores, indicating a degradation of biotic resource and therefore stream quality.

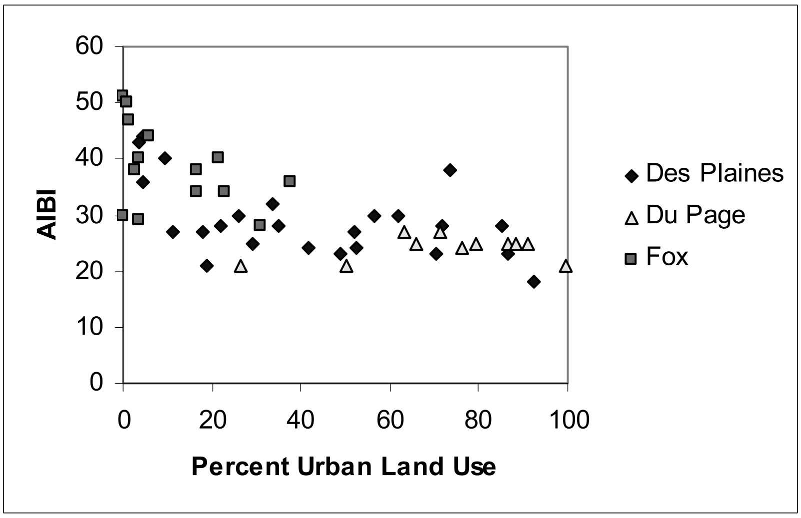

Figure 3. Annual fish AIBI integrity scores as a function of subbasin near-surface geology: Fox (sand), Des Plains (…), and Du Page (clay).

Other potential prediction variables that were explored for trends included relations among the biotic and habitat indices. Plotting the biotic indices against one another revealed a linear trend with a comparatively large degree of variability that was previously implied by having the lowest correlation coefficient among prediction variables (Figure 5). Habitat scores did not appear to be affected by the percentage of urban land use, thereby supporting previous studies by Wang et al. (1997) and Fitzpatrick et al. (2005). The finding that habitat indices were not related to the urbanization measures used in this study suggests that (1) the standard assessments may not be sensitive enough to quantify changes in hydrology, and (2) the definition of ecological integrity can be restricted herein to biological integrity. With the exception of habitat, all prediction variables appear to indicate a range (10-30% urban land use) in degradation threshold (Figs. 2, 3, and 4a) that is more or less consistent with other findings (Wang et al., 1997; Dreher, 1997; Fitzpatrick, 2005).

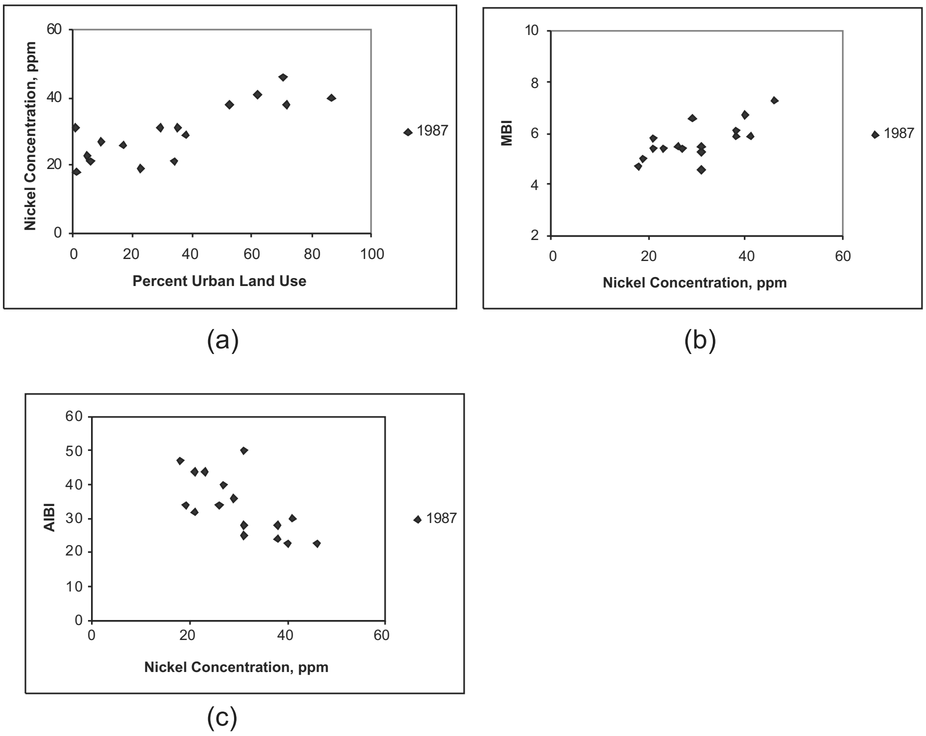

Figure 4. Relations between nickel concentration in bed sediment: (a) percent urban, (b) MBI, and (c) AIBI.

2.1.2. *Regression and Residual Analysis*

In this study, a nonlinear least-squares regression approach (Cooley and Naff, 1982) was used to estimate best-fit parameters for predictive equations that compute AIBI scores, MBI scores, and nickel concentration as a function of population density; AIBI scores as a function of MBI scores; AIBI scores as function of nickel concentration; percent urban land use as a function of nickel concentration; and change in population density as a function of basin size (urban growth). The equations and fitted-parameters for these functions are summarized in Table 3. Because the dependent variables are stochastic, residual analysis must be performed and the results incorporated into the regression equation.

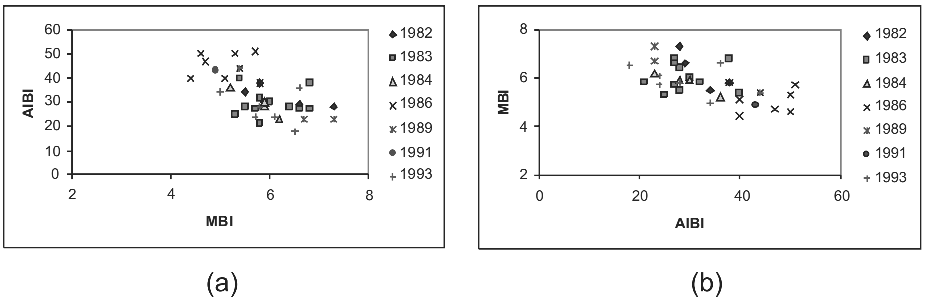

Figure 5. Relations between biotic indices: (a) AIBI as a function of MBI, and (b) MBI as a function of AIBI.

Table 3. The deterministic equations and fitted-parameters for predictive equations [AIBI - alternative index of biotic integrity; MBI - macroinvertebrate biotic index, Ni - bed sediment nickel concentration, %Urb - percent urban land use, PopD - population density, dPopD - change in population density]

Variable	Residual Equation	Parameters			
		a	b	c	d
AIBI (PopD)	= (c-d) * exp(–a * b * PopD) + d	16.84	0.0001	55.0	19.7
MBI(AIBI)	= AIBI(PopD) * a + b	–0.05	7.41	–	–
AIBI(Ni)	= $[\{1/1 + (a * Ni)^{b}\}^{-c}] *60$	0.05454	1.89	0.473	–
%Urb(Ni)	= $\{1.5*(Ni/a)^{b-0.5}\}*86.73$	48.36	1.15	–	–
dPopD(basin)	= basin * a + b	–2.76	8.66	–	–

By virtue of its formulation, nonlinear regression renders an otherwise stochastic process deterministic through the estimation of a single set of best-fit parameters. As described, the use of these best-fit parameters in the predictive equation results in a deterministic outcome; that is, a given input always produces the same output. Whereas deterministic equations are appropriate for describing nonrandom variables, these equations are inappropriate for predicting the range of behavior attributed to random variables such as AIBI, MBI, nickel concentration, and urban growth. To convert from deterministic to stochastic equations, the process variability are reintroduced. This variability and uncertainty is reintroduced by adding residuals back to the deterministic equation following random sampling of probability density functions describing the set of differences between the measured and predicted values (residuals). In this study, the various probability distributions of residuals were fit to probability density functions that are summarized in Table 4. The quality of fit was judged by using one of several goodness-of-fit criteria that included Chi-square, Kolmogorov-Smirnov, and Anderson-Darling (Werckman et al., 2001). The Monte Carlo technique used to select random residuals from these residual probability density functions is discussed in the next section.

Table 4. Summary table of probability distributions fit to residuals [AIBI - alternative index of biotic integrity; MBI - macroinvertebrate biotic index, Ni - bed sediment nickel concentration, %Urb - percent urban land use, PopD - population density, dPopD - change in population density]

Variable residual	Probability distribution	Minimum	Maximum	Loc/Shape, mean, or mode	Scale
AIBI (PopD)	Extreme value	–16.2	27.8	–3.54	6.41
MBI(AIBI)	Extreme value	–1.19	2.59	–0.26	0.48
AIBI(Ni)	Logistic	–0.34	0.32	–0.01	0.06
%Urb(Ni)	Gamma	–0.48	0.57	–1.11/39.3	0.03
dPopD(basin)	Extreme value	–284.7	621.6	–61.2	114.8

2.1.3. *Monte Carlo Technique*

The Monte Carlo technique overcomes challenges associated with devising and implementing analytical stochastic equations through the use of a random number generator. Generally, the Monte Carlo method builds up successive model scenarios (called realizations) using input values that are randomly selected to reduce the likelihood for bias from probability distributions already defined. In this study, the Monte Carlo method (Sargent and Wainwright, 1996) is used to draw random values from probability density functions for each model input variable used in the calculation. For example, by incorporating residual probability density functions into the predictive equations derived through regression, repeated sampling and calculation of the associated dependent variable results in alternate realizations (equally likely simulations) called stochastic modeling. Examples of stochastic modeling provided (including the three realizations and fitted line) include AIBI scores as a function of population density (Figure 6a), MBI scores as a function of AIBI

(Figure 6b), AIBI as a function of nickel concentration (Figure 6c), percent urban land use as a function of nickel concentration (Figure 6d), and change in population density as a function of tributary basin (Figure 6e). Qualitatively, the stochastic modeling appears to have reasonably replicated the random character for all of the variables evaluated. A more rigorous validation of the stochastic modeling process is provided next.

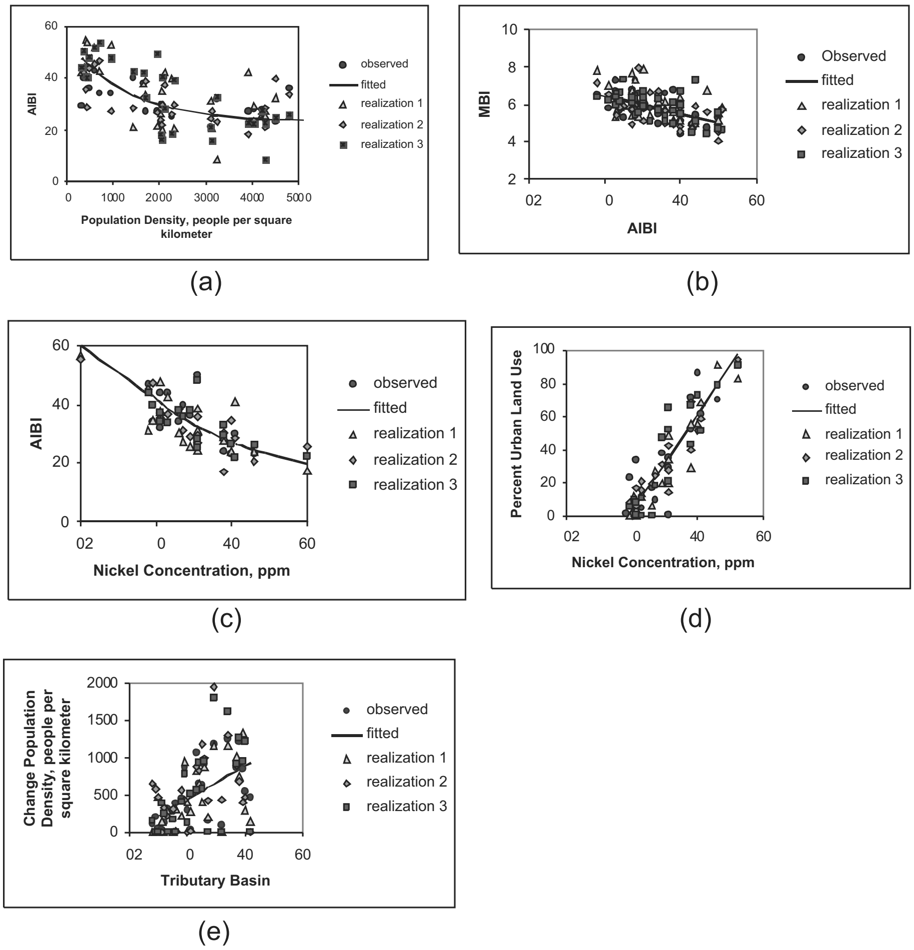

Figure 6. Stochastic modeling of selected variables: (a) AIBI as a function of population density, (b) MBI as a function of AIBI, (c) AIBI as a function of nickel concentration, (d) percent urban land use as a function of nickel concentration, and (e) change in population density as a function of tributary basin.

2.2. MODEL VALIDATION

Validation of the stochastic modeling approach involved three steps: (1) test the stability and convergence of a representative Monte Carlo prediction, (2) compare statistics for selected simulations to field measurements, and (3) evaluate a future prediction against an independent measure. The following sections describe these components and how they are used in model validation.

2.2.1. *Stability and Convergence*

The stability and convergence of individual model predictions is tested using the bootstrap approach (Werckman et al., 2001). In using the bootstrap approach, sample statistics (estimated mean, standard error, and confidence intervals) are computed from 200 independent (repeated) predictions of AIBI scores as a function of population density for the URIB. The bootstrap approach is repeated for an increasing number of Monte Carlo trials (500, 1,000, 1,500, 2,000) until the percent change in upper and lower confidence intervals is less than 1 percent. In this study, the lower and upper confidence intervals for all Monte Carlo tests agreed within about 1 percent. The standard error of average simulated AIBI, however, continued to decrease until the total number of trials was equal to or greater than 1,500. For this reason, the use of 1,500 Monte Carlo trials is deemed sufficient to ensure convergence and stability of predictions in this study.

2.2.2. *Comparison of Simulated and Measured Forecasts*

The relative basin accuracy of the stochastic model is evaluated by comparing statistical summaries of measured to simulated AIBI and MBI scores for the UIRB (Table 5). In general, good correspondence exists between measured and simulated biotic statistics, but particularly for the MBI. Whereas the median statistics appear to give the best correspondence between actual and simulated biotic scores, the extreme conditions represented by the minimum and maximum biotic scores are characterized by more uncertainty. One plausible reason is that the residual analysis for AIBI model as a function of population density is not as accurate as when using correlated random variables (Friedel, 2004). Another reason may be that an alternative regression equation and (or) improved residual probability distribution might exist. Because the stochastic models of biotic integrity provide adequate simulations of AIBI and MBI scores throughout the UIRB, the next section is used to provide insight on the ability of stochastic models to predict future ecological integrity.

Table 5. Summary statistics for simulated and measured biotic integrity [AIBI - alternative index of biotic integrity; MBI - macroinvertebrate biotic index]

Biotic Index	AIBI	AIBI[1]	MBI	MBI[1]
Statistics	Measured	Simulated	Measured	Simulated
Trials	37	1,500	37	1,500
Minimum	18.0	7.4	4.4	4.6
Maximum	44.0	60.8	7.9	7.9
Range	26.0	53.5	3.5	3.3
Median	30.5	29.8	23.2	6.0
Standard deviation	7.1	11.7	0.8	0.7
Coefficient of variability	0.31	0.10	0.31	0.10

[1] simulated as a function of population density.

2.2.3. *Long Run Creek 2000 Forecast Validation*

One means of testing the ability of a stochastic model to predict future events is through validation of the simulated results against an independent measure. Model validation is conducted herein by comparing summary statistics of AIBI forecast scores (future predictions) against an independent biotic measure for Long Run Creek (Table 6). The validation is only considered approximate, primarily because of the lag between the year 2000 simulation date and 1995-1999 computational period and secondarily because of uncertainty in the computed score. For this case, the simulated 2000 median AIBI score of 23.2 is in good correspondence with the computed 1995-1999 value of 26. This statement is particularly true because of the slight increase in population density (urban growth) that is assumed to have occurred during the lag time between the 1995-1999 calculation and 2000 simulation. As was implied in the comparison of basin simulated AIBI summary statistics, the Long Run Creek simulation had a broad range of probable AIBI scores: 9.2 (minimum), 23.2 (median - or 50th percentile), and 53.3 (maximum). The extreme values are very different than the late period calculation, however, when considering values between the first (25th percentile) and third (75th percentile) quartiles, the simulated AIBI values remain reasonable.

2.3. SCENARIO MODELING

The effects of future urbanization on ecological integrity in various tributary basins that occur along an urban land use gradient are evaluated through scenario modeling. These modeling scenarios permit quantification of the probable effects that future urbanization has on ecological integrity in the Big

Rock Creek, Des Plaines River, Mill Creek, Long Run Creek, and Flag Creek tributary basins during 2000 and 2010. The respective 1990 median (initial) amount of urbanization for the Big Rock Creek, Des Plaines River, Mill Creek, and Flag Creek basins are 1, 4.5, 9.5, and 86 percentage of urban land use. The urban growth scenarios used to evaluate the effects of future urbanization on ecological integrity are briefly described below.

Scenario 1: simulations of the AIBI as a function of population density for Big Rock Creek, Des Plaines River, Mill Creek, and Flag Creek over the two urban growth periods 1990-2000 and 2000-2010.

Scenario 2: simulations of the AIBI as a function of macroinvertebrate biotic index for Big Rock Creek, Des Plaines River, Mill Creek, and Flag Creek over the two urban growth periods 1990-2000 and 2000-2010.

Scenario 3: simulations of the AIBI as a function of bed sediment nickel concentration for Mill Creek over the urban growth period from 1990 to 2010.

Scenario 4: simulations of the percent urban land use as a function of bed sediment nickel concentration for Mill Creek over the urban growth period from 1990 to 2010.

Table 6. Summary statistics for the predicted 2000 AIBI and computed scores at Long Run Creek [AIBI - alternative index of biotic integrity]

Biotic Index Statistics	AIBI[1]	AIBI[2]
Percent urban land use	~28.5	28.5
Trials	1	1,500
Minimum		9.2
Maximum		53.3
Range	–	44.9
Median	26	23.2
Standard deviation		7.6
Coefficient of variability		0.31
25th percentile	–	21
75th percentile	–	28

[1] computed score , Bertrand et al., 1996

[2] simulated as a function of population density.

3. Results

In scenario 1, observed changes in the ecological integrity were based on simulated AIBI scores and changes in population density. In general, the simulated ecological integrity was degraded in response to urban growth after 1990 as indicated by a range of resource quality classes in all tributary basins for the years of 2000, and 2010 (Table 7). For example, there is an 85 percent chance that the quality of Big Rock Creek and Des Plains River basins might be degraded 1 or more resource classes due to urbanization from 1990 to 2000. Given the variability in measurement quality, this means there also is a 15 percent chance that the 2000 resource quality might be the same or slightly better than in 1990. Whereas Mill Creek had a 75 percent chance of degrading by more than one class due to urbanization from 1990 to 2000, Flag Creek, the most urbanized tributary basin, had only 55 percent chance that the resource quality might degrade 1 class. By 2010, it was likely that only Big Rock Creek, the least urbanized tributary basin, would continue to experience a loss of ecological integrity, but not more than 1 resource quality class. If cumulative distribution extreme values (minimum or maximum) were of primary interest then there would be no change in ecological integrity after 2000.

Changes in the 50th percentile (median) population densities for Big Rock Creek (least urbanized) and Flag Creek (most urbanized) tributary basins during 1990, 2000, and 2010 were 12 people per km^2, 420 people per km^2, and 700 people per km^2; and for Flag Creek during the same years the urban population densities were 2957 people per km^2, 3205 people per km^2, and 4096 people per km^2. Under these urban growth scenarios, the corresponding 50th percentile of AIBI scores indicates that by 2010 the ecological integrity would degrade in Big Rock Creek and Des Plaines basins by 2 quality classes (good to poor), whereas urbanization in the Mill Creek and Flag Creek basins would have degraded the ecological integrity only 1 quality class (poor to very poor). Stability of ecological integrity in Flag Creek is a function of the community being fully urbanized. Given that ecological integrity appears to degrade at about one quality class per decade, both Big Rock Creek and Des Plaines River also are likely to achieve the dubious status of very poor biotic health by the year 2020. Whereas these results imply that the ecological integrity in all tributary basins would be fully degraded in the UIRB by 2020, new management practices coupled with restricted or reversed growth trends may mitigate these future effects.

The purpose modeling scenario 2 was to: (1) demonstrate the ability to simulate the MBI scores as a function of AIBI scores; and (2) use these results with the simulated AIBI scores to establish a new MBI scale for future calibration and interchange between biotic scores. Because the characteristic trends for AIBI and MBI quality classes are reversed (negative correlation), the simulated results for MBI classes are presented for comparison to the AIBI classes in terms of a reverse cumulative distribution (Table 8). Comparison of the cumulative distributions for simulated AIBI classes and original (uncalibrated) MBI classes reveals a large disparity between biotic indices indicating the inability to interchange results. In this case, only 5 of 40 (4 percent) simulated resource quality classes were the same between AIBI and uncalibrated MBI results. By calibrating the simulated distribution of MBI to AIBI classes a new scale was derived (Table 1). Because the MBI scores were simulated as a function of AIBI scores, the factors associated with potential differences in integration of fish and macroinvertebrate communities were taken into account. Upon calibrating the MBI scores to the new scale, 36 of 40 (90 percent) resource quality classes now were the same indicating a higher likelihood for determining the ecological integrity using either biotic index.

Table 7. AIBI resource quality as a function of population density for Big Rock Creek, Des Plaines River, Mill Creek, Long Run Creek, and Flag Creek over urban growth periods: 1990-2000, 2000-2010 [AIBI - alternative index of biotic integrity; MBI - macroinvertebrate biotic index; G - good; F - fair; P - poor; VP - very poor; bold text indicates change from previous decade]

Cumulative Distribution Function of Alternative Index of Biotic Integrity Classes											
Tributary basin	0	10	20	30	40	50	60	70	80	90	100
Computed as function of 1990 population density											
Big Rock Creek	G	G	G	G	G	G	G	G	G	G	G
Des Plaines River	G	G	G	G	G	G	G	G	G	G	G
Mill Creek	F	F	F	F	F	F	F	F	F	F	F
Flag Creek	P	P	P	P	P	P	P	P	P	P	P
Simulated as function of 2000 population density											
Big Rock Creek	**VP**	**P**	**F**	**F**	**F**	**F**	**F**	**F**	**F**	G	E
Des Plaines River	**VP**	**P**	**P**	**P**	**P**	**F**	**F**	**F**	**F**	G	E
Mill Creek	**VP**	**VP**	**VP**	**P**	**P**	**P**	**P**	**P**	F	F	G
Flag Creek	**VP**	**VP**	**VP**	**VP**	**VP**	**VP**	P	P	P	P	F
Simulated as function of 2010 population density											
Big Rock Creek	VP	P	**P**	**P**	**P**	**P**	F	F	**F**	G	E
Des Plaines River	VP	VP	P	P	P	**P**	**P**	F	F	**F**	E
Mill Creek	VP	VP	VP	**VP**	**VP**	P	P	P	**P**	F	G
Flag Creek	VP	VP	VP	VP	VP	VP	**VP**	P	P	P	F

Table 8. Calibration of MBI classification scheme to the AIBI classification scheme for urban growth period: 1990-2000 [AIBI - alternative index of biotic integrity; MBI - macroinvertebrate biotic index; G - good; F - fair; P - poor; VP - very poor; bold text indicates difference from AIBI classes]

Cumulative Distribution Function of Alternative Index of Biotic Integrity Classes											
Tributary basin	0	10	20	30	40	50	60	70	80	90	100
2000 AIBI simulated as function of population density											
Big Rock Creek	VP	P	F	F	F	F	F	F	G	G	E
Des Plaines River	VP	P	P	P	P	F	F	F	F	G	E
Mill Creek	VP	VP	VP	P	P	P	P	P	F	F	G
Flag Creek	VP	VP	VP	VP	VP	VP	P	P	P	P	F
Reverse Cumulative Distribution of 2000 Macroinvertabrate Biotic Index (MBI) Classes											
Tributary basin	100	90	80	70	60	50	40	30	20	10	0
2000 MBI simulated as function of AIBI - original scale											
Big Rock Creek	**F**	**F**	F	**G**	**G**	**G**	**G**	**G**	G	G	**G**
Des Plaines River	**F**	**F**	**F**	**F**	**G**	**G**	**G**	**G**	**G**	G	**G**
Mill Creek	**F**	**F**	**F**	**F**	**F**	**F**	**G**	**G**	**G**	**G**	G
Flag Creek	**F**	**F**	**F**	**F**	**F**	**F**	**F**	**F**	**F**	**G**	**G**
2000 MBI simulated as function of AIBI - calibrated scale											
Big Rock Creek	VP	P	**P**	F	F	F	F	**G**	G	G	E
Des Plaines River	VP	**VP**	P	P	F	F	F	F	**G**	G	E
Mill Creek	VP	VP	VP	P	P	P	F	F	F	F	G
Flag Creek	VP	VP	VP	VP	VP	**P**	P	P	P	**F**	G

In scenarios 3 and 4, the purpose was to illustrate the utility of using alternate predictors to predict ecological integrity in future biomonitoring studies. For this case, bed sediment nickel concentrations were used to simulate the 2010 ecological integrity and percent urban land use. Using the simulated AIBI classes as a standard, the simulated resource classes based on nickel concentration appeared to underestimate the ecological integrity by 1 quality class (Table 9). Because the time and costs associated with determining nickel concentration are much less than for determining either biotic index, future studies should consider calibration of this and other more cost-effective variables that correlate to urban measures. Better correspondence was demonstrated between 2010 simulations of percent urban growth as a function of AIBI scores and bed sediment nickel concentration (Table 10).

Table 9. AIBI modeling of Mill Creek 2010 stream quality as a function of: (a) population density, (b) nickel concentration in bed sediment [AIBI - alternative index of biotic integrity; MBI - macroinvertebrate biotic index; G - good; F - fair; P - poor; VP - very poor; bold text indicates difference from AIBI simulated as function of population density]

Cumulative Distribution Function of 2010 Alternative Index of Biotic Integrity Classes											
Tributary basin	0	10	20	30	40	50	60	70	80	90	100
Simulated as function of population density											
Mill Creek	VP	VP	VP	P	P	P	P	P	F	F	E
Simulated as function of nickel concentration in bed sediment											
Mill Creek	VP	**P**	**F**	**F**	**F**	**F**	**F**	**F**	**G**	**G**	E

Table 10. Basin percent 2010 urban Land simulation for Mill Creek as function of: (a) population density, (b) nickel concentration in bed sediment [AIBI - alternative index of biotic integrity]

Cumulative Distribution Function of 2010 Percent Urban Land Use											
Tributary basin	0	10	20	30	40	50	60	70	80	90	100
Simulated as function of AIBI											
Mill Creek	15	17	17.6	18.1	18.5	19	19.4	20	20.7	21.6	25.2
Simulated as function of nickel concentration in bed sediment											
Mill Creek	16.3	17.1	17.3	17.5	17.7	17.9	18.1	18.3	18.7	19.1	21

4. Summary and Conclusions

A stochastic modeling approach was used to predict the effects of urbanization on ecological integrity. Analysis of historical data revealed urban correlated variables, absence of synoptic bias, different biotic scales and trends, and no specific urban thresholds with respect to ecological integrity. Because of the unexplained measurement variability in biotic indices and uncertainty in urban growth, numerical stochastic equations were developed, validated, and used to predict the effects of urbanization on ecological integrity in Rock Creek, Des Plaines River, Mill Creek, and Flag Creek tributary basins of the Upper Illinois River Basin. The important findings are as follows:

1. The comparatively high degree of correlation between copper, chromium, lead, mercury, and zinc with MBI scores but not AIBI scores suggests possible differential time and space integrating abilities of fish and macroinvertebrate communities.
2. The degradation of ecological integrity in tributary basins occurs at differential rates and with a probable distribution of likely outcomes. For example, the median predictions of ecological integrity from 1990 and 2010 is 2 quality classes (good to poor) in the Big Rock Creek and Des Plaines tributary basins, and 1 quality class (poor to very poor) in the Mill Creek and Flag Creek tributary basins.
3. A new scale is available for converting MBI scores to biotic resource quality classes for interchanging results with AIBI scores. This calibrated scale should be useful in more urbanized streams where it is not always possible to compute AIBI scores, and for comparison between biotic indices in other studies.
4. Bed sediment nickel concentration is a good but nonconservative predictor of ecological integrity. With the exception of the extreme values, the simulated resource classes based on nickel concentration overestimated the ecological integrity by 1 quality class. Because the time and costs for determining nickel concentrations are much less than for determining biotic integrity scores, future studies could use or consider calibration of this scale or other correlated variables as predictors.
5. Bed sediment nickel concentration is an excellent predictor of basin percent urban land use (or population density).

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GEOCHEMICAL MODELLING OF GEOTHERMAL FLUIDS - APPLICATION OF THE COMPUTER PROGRAM *SOLMINEQ.88*

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Abstract: The investigations were carried out based on hydrogeochemical data of the geothermal fluids and well fluid logs of the well KD 22. This well belongs to the Kizildere geothermal field in Western Anatolia. The aim of the investigation was to apply the geochemical model SOLMINEQ.88 on the reinjection as a tool to predict scaling. The production system, the (imaginary) heat exchanger, reinjection well, and reservoir were respected. To start the modelling, some preparations on the input data were necessary. Applying the pressure logs in dynamic state and respecting the steam pressure of water, the partial pressure of CO_2 in the reservoir was calculated to be 42 bar. All ingredients such as steam, CO_2, trace gases, and alkaline earth metals, which were lost on the path of up-welling were refilled. The composition of the reservoir fluid was recalculated respecting the steam loss. During up-welling and after silencing of the fluid, the main part of the CO_2 releases. Supersaturation occurs with respect to calcite and aragonite. Carbonate scaling is precipitated. During the modelled heat exchange down to 50°C, amorphous silica becomes supersaturated due to the lowering of the temperature. In the model for the reinjection well, silica is supersaturated, as well. The model for the mixing in the reservoir showed, that the supersaturation of amorphous silica disappears immediately after mixing. The pre-calculation of mixtures should be preferred, the mixing option of SOLMINEQ.88 does not produce suitable results every time. The modelling of geothermal fluids discovers the

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thermodynamic character. It gives no information about kinetic or mass balance aspects. Continental geothermal fluids can be modelled easily applying the Debye-Huckel theory due to their moderate ionic strength. Nevertheless, some problems occur due to the programming of SOLMINEQ.88.

Keywords: fluid geochemistry; geochemical modelling, Kizildere geothermal field; reinjection; reservoir

1. Introduction

Production of geothermal fluid without further reinjection causes often environmental problems. For instance, in the Kizildere geothermal plant, environmental problems such as (i) drop of the reservoir pressure and (ii) contamination of the Buyuk Menderes River (BMR) with boron occur. The reinjection is needed. Often, from the hydraulic and chemical view reinjection is problematical. According to the different physical character of hot production fluids and cold reinjection fluids, wells with suitable productivity indices (PI) can have too low injectivity indices (II). However, after solving the hydraulic problems, chemical problems might occur such as scaling. In the Kizildere area, the formation of carbonate scaling was observed in the production wells and in the separators. It was supposed that scaling will occur after reinjection, too. High contents of silicon in the fluids bear the risk that silica scaling can be formed in the heat exchanger, in the reinjection well, and in the reservoir. The mixing of the cold (and enriched) reinjection fluid with the hot reservoir fluid may cause fluid reactions and precipitation of scaling.

Applying geochemical computer models such as SOLMINEQ.88 (Kharaka et al., 1988), SOLVEQ (Spycher and Reed, 1990), or PHREEQE (Parkhurst et al., 1980), the equilibrium state of thermal fluids can be checked in every step of the facilities. Nevertheless, it must be faced that the quality of such a model depends on the quality of its input data, on the quality of the programming, and on the design of the program tools.

2. Investigated Areas and Methods

The investigations have been carried out in the Kizildere geothermal field near Denizli in the Turkish Aegean region, Western Anatolia. Kizildere is located in the Eastern Buyuk Menderes Graben (EBMG), 25 km western of the famous sinter terraces of Pamukkale. The Kizildere geothermal power plant produces mainly electricity from geothermal steam. The power amounts nowadays approximately 14 MW_e. All production wells are discharging around 1200 t of geothermal fluid per hour without pumping. After separating geothermal steam which contains the main part of the carbon dioxide, the installed low pressure turbine forces the generation of electricity. The waste water after separator is emitted partly into the BMR without further reinjection and partly reinjected.

Investigations on the thermal waste fluids after separator and on the separator fluids were carried out as well as analysis of the scalings. Gas analysis was done by the MTA (Çetiner, 1999). Several geothermal outcrops of the nearby region such as Pamukkale, Karahayit, Kamara, Yenice, Babacik, Golemezli, and Tekkehaman were taken into account, as well. Physical parameters and unstable constituents were determined onsite, the rest was determined in the laboratory. P/T-logs of the geothermal production wells of Kizildere were used to calculate the content of CO_2 within the reservoir fluid. These results were compared with the analytical results. Material balances were performed using preferably the data derived from the well KD 22.

Starting with the character of the waste water, the recalculation of the chemical character of the reservoir fluid needed information about material losses between inflow zone and separator outlet such as (i) the loss of steam, (ii) the loss of scaling, and (iii) the loss of non-condensable gases (NCG). In addition, energy conversion processes in the well and in the separator had to be evaluated. Applying energy balances and material balances, the chemical character of the reservoir fluid could be recalculated without downhole-sampling.

Basing on this data set, thermodynamic models were calculated applying the SOLMINEQ.88 program (Kharaka et al., 1988). Respecting the high temperatures and the moderate salinity, the Debye-Huckel theory was used. As the first step, by adding all ingredients and simulating the thermal and hydraulic character of the reservoir, the reservoir fluid was modelled. To test the model, the boiling and degassing processes within the production well and within the separator which were thermodynamically proven were simulated applying the boiling function of SOLMINEQ.88. The changes of the partial gas-pressures and the mineral saturation indices (SI) were investigated.

To simulate the reinjection fluid, the waste water after separator was adapted to the conditions of the (imaginary) heat exchanger and of the (imaginary) reinjection well. For the simulation of the heat exchange, the fluid was step-rated cooled from the conditions after the separator (100°C) down to reinjection temperature (minimum 50°C).

The character of the reinjected brine within the reinjection well (total vertical length 2000 m) was studied simulating the increase of the hydrostatic pressure and respecting a very gentle conductive heating-up by the surrounding rocks (ΔT = 10 K/ 2000 m). In addition, kinetic experiments were done to simulate the reaction of silicon (Neumann, 1997; Giese, 1997).

Finally, the mixing of the reinjected fluid and the simulated reservoir fluid was modelled using two strategies such as (i) direct mixing applying the mixing function of SOLMINEQ.88 and (ii) first calculation of the mixture's chemistry and modelling of this data set at reservoir conditions afterwards.

3. Results

3.1. RESERVOIR WATER

3.1.1. *Evaluation of c(CO_2) using the p/T-logs*

Two main anomalous characters can be observed within the logs which have been measured in the fluid during dynamic state. The pressure function starts linear and deviates as a curve. The temperature function deviates from the style of a linear function, as well. Assuming that (i) pressure losses by inner friction of the fluid due to turbulent flow-regime and by outer friction of the fluid due to the well resistance, and that (ii) density changes by cooling-down can be ignored, the pressure vs depth should be a linear function for a liquid. As soon as gas bubbling occurs, the function converts into a curve. At the bubbling point (IDP), the inner pressure of the fluid which consists of the steam pressure of water (p_{steam}) and the partial pressure of the NCG ($p_{partial,NCG}$) exceeds the hydrostatic pressure ($p_{hydrostat}$) of the column according to Giese (1997).

$$p_{inner} = p_{steam} + p_{partial,NCG} \geq p_{hydrostat} = p_0 + \rho g h \tag{1}$$

where p_0 is the outer pressure, ρ is the density, g is gravitational acceleration (9.81 m/s^2), and h is the length of the column. To analyse the depth of degassing (IDP) using the slope of the linear part of the function p vs depth (x), first the density of the fluid and the pressure loss in the interval between x_y and x_{y+1} which is regularly expected has to be calculated. This pressure difference dp_{reg} has to be compared with the measured one dp_{real}. The pressure anomaly d(dp) which occurs above the IDP is

$$d\left(dp\right) = dp_{reg} - dp_{real} \tag{2}$$

The function lg d(dp) vs depth x is a curve starting at the depth of degassing x(IDP). Inserting x(IDP) into the p vs x function, the pressure p(IDP) can be estimated. Subtracting the p_{steam} at the temperature T(IDP), the partial pressure of the NCG can be calculated.

Presuming, that the NCG consist only of CO_2, this partial pressure has to be inserted into the CO_2-option of SOLMINEQ.88. If the partial pressure exceeds the allowed limits, TIC can be added changed iteratively until the modelled partial pressure fits.

Figure 1 shows the results of the analysis of the pressure anomaly for the well KD 22 of Kizildere. The function of the so called primary anomaly of the pressure measured in dynamic state d(dp)(dyn) indicates that under test conditions above –600 m below WH the first degassing occurs. Respecting the steam pressure of water to be 14.9 bar at 197.9°C and the p(IDP) to be 57 bar, the partial pressure of the carbon dioxide was estimated to be 42 bar. The broken line remarks the x(IDP) (initial degassing point) at –600 m. Nevertheless, significant molar amounts of gasses appear just more upward. According to Olcenoglu (1986), the saturation pressure of CO_2 in the reservoir water was estimated to be 53.1 bar (applying Ellis and Golding, 1963).

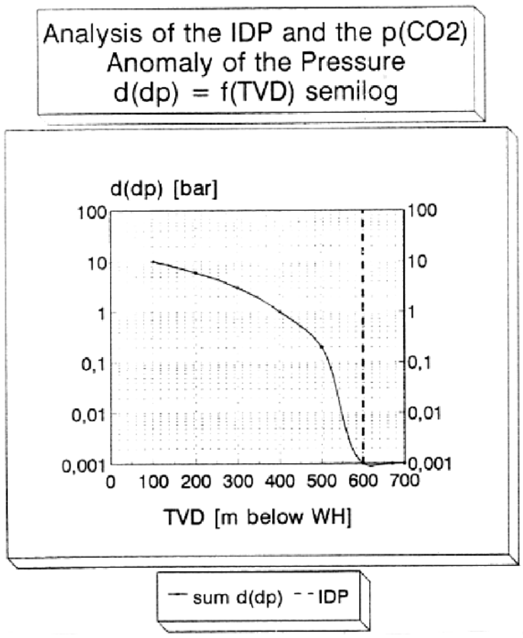

Figure 1. Analysis of the anomaly of the static fluid pressure $p_{stat} = \rho*g*h$ during production (=dynamic state). Lg d(dp) (dyn) vs depth x.

The analysis of the temperature changes works similar. Depending on the effects of conductive heat loss dT_{cond}, the function has to be corrected by subtraction of the non-adiabatic heat loss presuming that the loss of heat by volume work can be ignored. The interpretation of the logarithmic function of the adiabatic heat loss dT_{ad} vs path (= b - depth) shows often as good results as the analysis of the pressure anomaly (Giese, 1997).

3.1.2. *Enrichment by Steam Loss*

Using the results of the IDP analysis, the maximum steam loss of the fluid (i) during the ascent in the well, (ii) at the separator, and (iii) at the silencer can be recalculated (using the steam table; H: enthalpy of the liquid; ΔH_b: boiling enthalpy):

$$X_{b,\max} = 100\% \frac{H\left(H_2 0, l, T_{upper}\right) - H\left(H_2 0, l, T_{lower}\right)}{\Delta H_b\left(T_{lower}\right)} \tag{3}$$

Less mixing effects appeared within and after the exfiltration zone (between well bottom [WB] at –887.5 m and bottom of the casing at –575 m). According to the dynamic gradient log, the starting temperature was 197.9°C measured at -670 m below wellhead (WH). The final temperature at WH was estimated to be 188°C. The loss of temperature depends 90% on adiabatic cooling-down (steam loss) and 10% on conductive cooling (by the surrounding rocks). At the separator and silencer, the temperature is lowered in two steps (188° to 147°C, 147°C to 100°C) by adiabatic steam loss. From this point of view, 100°C (standardised to 0 m a.s.l.) were taken into account as the target of T (input to the heat exchanger). During the log measurement, the sum dT (–670 m TVD to WH) was 10 K, the sum (dT-dT[cond]) was 9 K.

The available heat for boiling down to 100°C amounts to 0.990 of the total heat loss. In fact, the steam loss X_b amounts to

$$X_b = X_{b,\max} \frac{96.86K}{97.86K} = 18.8\% * 0.990 = 18.6\% \tag{4}$$

down to 100°C, respecting the data of the steam table (Koglin, 1954). Within the temperature range of the production well, the steam loss X_b is 2.02% ($X_{b,max}$ = 2.25%).

3.1.3. *Recalculation of the Reservoir Water*

To correct the contents (c_i) in the waste water after silencer, first the lost steam had to be refilled. According to:

$$c_i\left(after\ boiling\right) = c_i\left(before\ boiling\right)\frac{100\%}{100\% - X_b} \tag{5}$$

the diluting factor had to be 100/81.4 = 1.229 from the waste water towards the exfiltration zone. The content of conservative species is 22.9% higher after silencing than below the IDP.

The partial pressure of 42 bar CO_2 was added at IDP conditions to the corrected data base mentioned above. After modelling, SOLMINEQ iterated finally 41.87 bar CO_2 at –600 m TVD to be equal to the content of 18600 mg CO_2/ kg fluid.

According to the results of the investigations at the separator and the balance, the contents of H_2S and NH_3 were corrected. Silicon was kept on the measured level. The highest alkaline earth metal contents were used to produce worst conditions. Calcite was set saturated, Mg and Sr were adapted to the results of the scaling analysis.

3.2. MODELLING OF CHARACTER CHANGES

The respected technical concept includes separation and silencing, and afterwards heat exchange down to 50°C, reinjection down to 2000 m below WH, and mixing with the reservoir fluid. Usually, in a closed doublette the steam would not be released to the atmosphere, only to the turbine. Released CO2 would be separated from the wellhead. On the contrary, for a test first the silenced fluid would be reinjected.

3.2.1. *Silencing and Heat Exchange*

Table 1 shows some basic input data and some modelling results. The modelled values of the partial pressure of carbon dioxide and water are shown in parentheses. The $p(CO_2)$ was inserted as 100% of the column pressure minus the steam pressure of water. The top line shows the data of the waste water after silencing.

Table 1. Basic input data (recalculated reservoir water and waste water after silencer) and selected result of the model applying SOLMINEQ.88

Data Input	Units	Reservoir	Waste Water
T	°C	197.9	100.0
P	bar	59.0	1.013
pH (at T)		5.89	8.05
$c(Na^+)$	mg/kg	1056	1298
$c(K^+)$	mg/kg	115	141
$c(Mg^{2+})$	mg/kg	0.31	0.22
$c(Ca^{2+})$	mg/kg	9.57	1.11
$c(Cl^-)$	mg/kg	104	128
$c(SO_4^{2-})$	mg/kg	605	744
$c(SiO_2)$	mg/kg	244	300
c(TIC[C])	mg/kg	5070	410
$c(Li^+)$	mg/kg	4.15	5.10
$c(Sr^{2+})$	mg/kg	0.82	0.01
$c(Al^*)$	mg/kg	0.58	0.71
$c(Fe^{2+})$	mg/kg	0	0
$c(Mn^{2+})$	mg/kg	0	0
$c(Zn^{2+})$	mg/kg	0	0
$c(F^-)$	mg/kg	16.8	20.6
$c(NO_3^-)$	mg/kg	0.08	0.10
$c(PO_4^{3-})$	mg/kg	0	0
$c(NH_3)$	mg/kg	14.4	3.50
$c(H_2S)$	mg/kg	0.98	0.50
$c(B^*)$	mg/kg	23.3	28.6
$c(As^*)$	mg/kg	0.86	1.06
E_h	V	–0.2	–0.2
Data Model	**Units**	**Reservoir**	**Waste Water**
$p(CO_2)$	bar	41.89	0.041
$p(H_2O)$	bar	14.87	1.013
SI (aragonite)		–0.058	–0.004
SI (calcite)		–0.001	0.096
SI (am. silic.)		–0.535	–0.064

The pH of the brine rises versus decreasing TIC. That means degassing of CO_2. Just if the TIC decreases, the critical carbonate content (better: activity a) increases as an effect of the pH change. The consequence is, that the saturation state of carbonate minerals such as aragonite and calcite ($CaCO_3$) increases, depending on the ion activity product (IAP) compared with the solubility product (Ksol)

$$IAP \Leftrightarrow K_{sol} = a\left(Ca^{2+}\right)a\left(CO_3^{2-}\right) \tag{6}$$

with a = c for low contents and the saturation index SI.

$$SI = \lg \frac{IAP}{K_{sol}} \tag{7}$$

Table 1 shows the changes of SI vs step, with respect to calcite, aragonite (carbonate scaling), and amorphous silica (silica scaling). At and below IDP, calcite is saturated and aragonite is undersaturated. Amorphous silica is undersaturated due to high temperature. The SI are rising towards the separation and silencing drastically (if no scaling would be lost). The reasons are:

- Enrichment after boiling (all).
- PH shift by CO_2 loss (calcite and aragonite).
- Cooling (amorphous silica).

To simulate the heat exchange, the fluid was cooled in three steps from 100°C down to 50°C. The TIC(C) was kept constantly on the conditions of the waste water. Figure 2 shows the results for calcite, aragonite, and amorphous silica. The SI of calcite and aragonite are rising after cooling-down below 100°C. That should not be expected, the minerals have an anomalous character of the solubility vs temperature. But the change of the pH forces this. Calcite is supersaturated in all steps, aragonite gets supersaturated below 100°C. The SI of amorphous silica is rising vs sinking temperature, as well. Below 90°C supersaturation occurs.

3.2.2. *Reinjection Well*

The output of the heat exchanger was taken into account as the input into the reinjection well. The chemical composition of the fluid was kept, temperature and pressure were modified towards the bottom of the well in two steps (–1000 m, –2000 m). Figure 3 shows that the SI are sinking as an effect of the conductive heating-up towards the bottom. Aragonite reaches undersaturation below –1400 m TVD, calcite remains in saturation state. The amorphous silica stays all the time supersaturated.

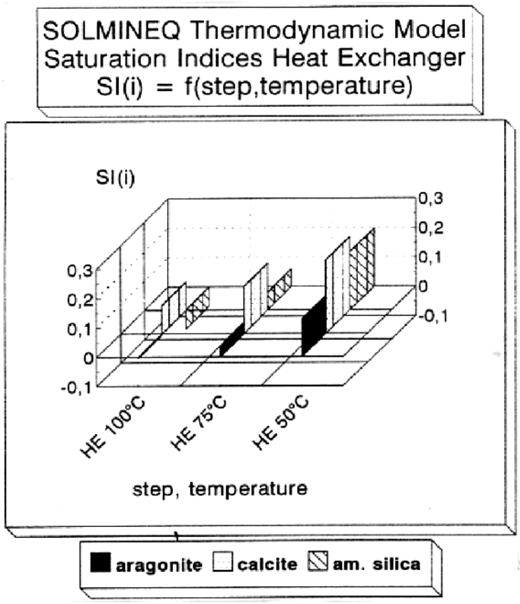

Figure 2. Thermodynamic model applying SOLMINEQ.88 for the heat exchanger between 100°C and 50°C. SI (calcite, aragonite, dolomite) versus step/temperature.

3.2.3. *Mixing in the Reservoir*

At last, the mixing of the reinjected brine (60°C, outlet of step 2 of the reinjection well, set on p[res]) and the reservoir fluid (here fluid like production well 2000 m below wellhead near 198°C) is a very critical point. Figure 4 shows the predicted SI of calcite, aragonite, and amorphous silica, derived from the model applying the pre-calculated data set. The application of the mixing option produced incongruent results. These were not taken into account. Depending on the very high TIC in geothermal brines and their buffer capacity, the effect of mixing corrosion (means: the mixture of two calcite saturated waters has to be undersaturated) could be depressed (compare Giese et al., 1998). However, the critical SI(aragonite, calcite) stay all the time negative. After mixing with a little reservoir fluid, the SI of amorphous silica decreases at once. On the contrary, the expansion of the cold injection front will bear risk in time.

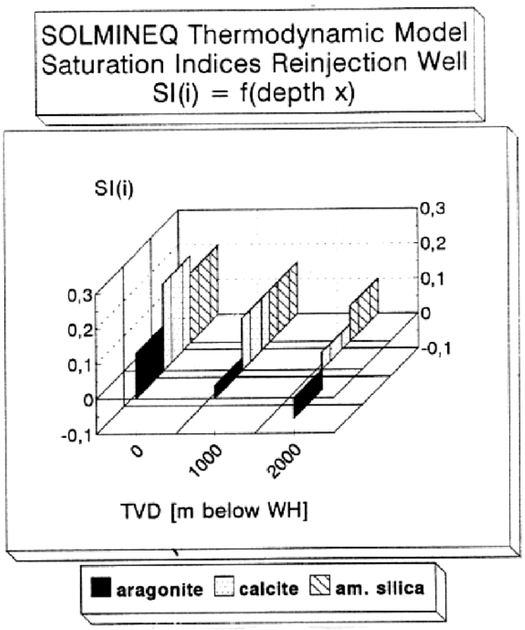

Figure 3. Thermodynamic model with SOLMINEQ.88 for the reinjection well between 0 m and –2000 m. SI (calcite, aragonite, amorphous silica) versus depth x.

4. Discussion

SI > 0 means supersaturation. From the thermodynamic view, this is the critical state, precipitation can be formed. But two more facts have to be respected which thermodynamic models such as SOLMINEQ.88 do not respect:

- Material balance
- Kinetics

If the potential of the total mass of the mineral which could be formed is very low, the risk is low. If there is an inhibition or a very slow chemical reaction velocity, no precipitation will occur from the kinetic view.

According to the results given by Dahms (1998) and Giese (1997), the precipitation of calcite should be depressed by

- the molar ratio Mg/Ca of up to 0.5,
- preferable precipitation of aragonite between 100° and 150°C, and
- high flow velocities (e.g. within production wells).

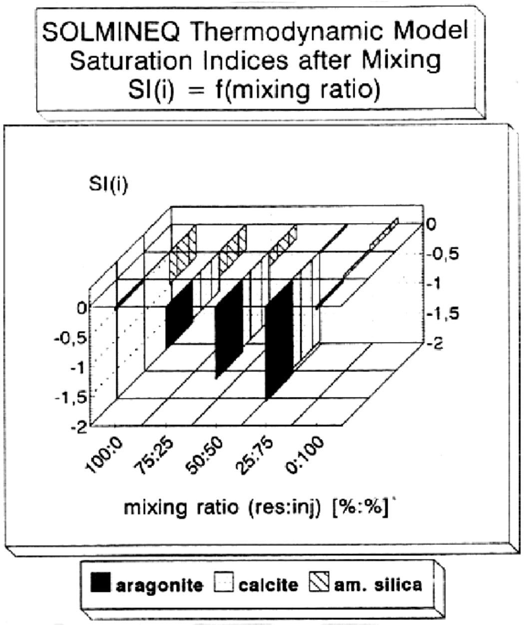

Figure 4. Thermodynamic model with SOLMINEQ.88 for the mixing after reinjection at 2000 m below WH. SI (calcite, aragonite, amorphous silica) versus mixing ratio.

On the contrary, lattice dotation of the lattice of $CaCO_3$ by Mg (in calcite) or by Sr (in aragonite) and the dotation of all carbonates by $(SiO_2)_n$ (amorphous silica or gel) can be observed during undersaturation state. This effect is called co-precipitation. Especially the amorphous silica as a sol and as a gel is very threatening by molecular filtering effects and surface adsorption.

After boiling and degassing of the reservoir fluid of Kizildere, in the production well, in the separator, and in the silencer carbonate scalings appear. The difference of the c(Ca) between down-hole and after silencer amounts up to 90%, the loss of carbonate scaling is significant. According to own analytical results, these scaling contain between 5% (well) and up to 20% (silencer) SiO_2.

By the shock-cooling, in the heat exchanger superficially an adsorptive layer of silica scaling can be formed. At the head of the exchanger system (100°C), a little aragonite and calcite can be expected, maybe with high contents of silica by co-precipitation.

According to the results for the reinjection well, the precipitation of amorphous silica followed by adsorption at the inner surface of the well can be expected (between 50° and 60°C). The silica scaling has to be seen as the main problem in the reinjection system. The reinjected brine contains not more than

5 mg $CaCO_3$ equivalents/kg fluid but more than 300 mg SiO_2 equivalents/kg fluid.

After mixing in the reservoir, the risk for silica scaling disappears at the mixing front by dilution and up-heating. Nevertheless, after a time the region surrounding the well will be cooled down and filled with unmixed cold reinjection-fluid. Molecular filtering might seal-up the aquifer and decrease the permeability.

5. Conclusions

Assuming the results of the computer modelling, proceeding a reinjection of waste water after silencing it should be kept an eye on the $(SiO_2)_n$ precipitation risk. From the view of the material balance, only the content of Si can affect significant scaling. Silica scaling can be expected

- in the heat exchanger,
- in the reinjection well, and
- in the cooled parts of the reservoir after reinjection.

Geochemical models such as SOLMINEQ.88 are an useful tool to solve very complex hydrogeochemical questions. In addition of data of p/T-logs in dynamic state and energy/material balances, the recalculation of reservoir brines is possible. Continental geothermal fluids can be handled easily with the conventional Debye-Huckel theory. The application of the SOLMIN-PITZ version using the combined Pitzer-equations/B-dot-equation often is not possible due to the too high T (more than 75°C), but not always necessary as well. Modelling tests below 75°C showed that the results for the fluids of Kizildere are closed to each other in both theories. According to the investigations on the application of computer models on geothermal waters and oil-formation waters (Kuhn, 1997; Thomas, 1994), the results of the SOLMINEQ fit until ionic strengths of 0.5 which was the I in these investigated brines.

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GROUNDWATER VULNERABILITY ASSESSMENT FOR INTERMONTANE VALLEYS USING CHU VALLEY OF KYRGHYZSTAN AS AN EXAMPLE

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Abstract: A groundwater vulnerability assessment using a standard index-and-overlay method is described for the city of Bishkek, Kyrghyzstan. The method adopted, partly dictated by the data available, used four weighted criteria. The resultant assessment is rational but rather subjective, a feature common to most such indexing systems. An analytical solution was therefore developed to attempt to reduce the element of subjectivity in the estimation of hydraulic inaccessibility, which is at the heart of groundwater vulnerability rationale. The resultant aggregated parameter (expressed as an index Ψ between 0 and 1) permits the estimation of the proportion of leakage occurring from the uppermost aquifer layer into a lower aquifer under pumping conditions, and so indirectly also the likely relative importance of contaminants in recent recharge. An example is shown of its application to the Chu Valley.

Keywords: Groundwater vulnerability; groundwater protection; hazard assessment; Kyrgyz Republic; Chu Valley; aggregate parameter of aquifer

A. Baba et al. (eds.), Groundwater and Ecosystems, 107–120.

1. Introduction

This paper proposes a method to calculate the set of geological factors used in ground water vulnerability assessment for the multi-layer aquifer conditions that are typical of intermontane valleys. Two approaches to the assessment of ground water vulnerability are described, using case studies of Bishkek city and the Chu Valley.

The Kyrghyz part of the Chu Valley is home to approximately 1,000,000 people, over a geographical area of 7,400 sq. km. Within the region, 401,700 hectares are arable land, of which 322,700 hectares has already been developed for irrigation (Sobolin, 1990). Bishkek, the capital of the Kyrgyz Republic, is situated in the Chu Valley.

The Chu basin has a very complicated hydrostratigraphy, resulting in complex water management conditions, typical of all intermontane basins in Central Asia. The groundwater setting of Bishkek and the Chu Valley includes the following key features:

- A semi-arid climate but extensive opportunities for recharge from rivers draining the nearby Alatau range of the Tien Shan Mountains.
- A complex unconsolidated fluvioglacial/alluvial aquifer system of Quaternary age, which is in excess of 350 m thick.
- Strong lateral and vertical variability. As a first approximation the system fines laterally northwards away from coarse clastic piedmont deposits composed of coalesced alluvial fans fronting the foothills into more stratified deep alluvial plain sediments.
- Unconsolidated sediments provide intergranular flow conditions, and there is hydraulic connection with surface flow in snow-melt rivers and associated canal systems, especially across the southern piedmont area where the aquifer system is considered to be both unconfined and to possess strong vertical connectivity.
- More complex semi-confined conditions are present in the northern part of the city where 3 aquifer systems have been identified by other resource investigation projects. Scope for significant pumping-induced vertical leakage exists, especially in the southern parts of Bishkek where low permeability horizons in the alluvial tract are thinner and less numerous.

Features of the Bishkek urban water infrastructure imposed upon this hydrogeological system include:

- 100% dependence on groundwater for drinking water, industrial and heating water needs.

- A very extensive piped water infrastructure (pressurised hot water as well as drinking water mains, plus piped sewerage and pluvial drainage), widespread use of on-site sanitation in single/two-storey residential areas and significant amenity irrigation of communal parts of residential areas.
- Supply wells located in a highly productive but very localised periurban valley-fill wellfield (Orto-Alysh production wellfield) and also throughout the urban area, at various depths.

Urban wells screened extensively in the middle aquifer (typically >120 m intake depth), but the lower part of the upper aquifer (40 m-120 m) is also widely tapped.

2. Vulnerability Assessment of Aquifer in the Bishkek Area Using a Four Criteria Approach

The data needed for the ground water vulnerability mapping (GVM) are largely prescribed by the general concepts of groundwater vulnerability mapping, by the assessment of the particular situation in Bishkek and by the technical data (in the form of maps and well records) which are currently available for the city. The following were identified as key criteria controlling aquifer vulnerability in Bishkek:

- **The presence & thickness of a low permeability surface layer** which will act to restrict infiltration of pollutants to the underlying aquifers and so protect them;
- **The geology of the aquifers**, and specifically their ability to transmit contaminants laterally and vertically from point of ingress
- **The depth to water table** (thickness of the unsaturated zone) because the presence of a thick unsaturated zone extends the residence time of infiltrating recharge and provides a medium for various attenuation processes to occur
- **Influent reaches of rivers/canals** crossing the project area, because linear recharge of contaminated river water can be a significant groundwater quality hazard

Each groundwater vulnerability criterion is classified into a number of zones reflecting relative susceptibility. Each zone is given a weight according to the relative impact on groundwater vulnerability. A map of the zones for each criterion was produced using GIS, with attribute information for each thematic layer describing the zones and the weighting given to each. These themes can be displayed singly over a base map showing the physical features of the project area, or combined to produce a composite map of relative groundwater vulnerability in which each polygon is in effect an area with the same point-score total.

The GVM is designed to illustrate those areas of the aquifer, which are intrinsically vulnerable to contamination. Hydrogeology, water usage and other information for vulnerability assessment was drawn from references (Grigorenko, 1979; Galanin et al., 1974; Galanin et al., 1982; Hydrogeology of the USSR, 1971; Karpachev et al., 1991; Levchenko et al., 1967; Litvak and Nemaltseva, 1990; Litvak, Nemaltseva and Burmin, 1991; Ljanov, 1994; Shestakov et al., 1982; Shestakov et al., 1980). The four component themes of the groundwater vulnerability map, with vulnerability classifications and vulnerability 'scores' for each parameter, are shown in Table 1 below. The composite map was then reclassified into areas of upper aquifer extreme, high, moderate or low vulnerability using the point-scoring system in Table 2.

Table 1. Point scoring system used for the GVM of the upper part of Bishkek aquifer system

Vulnerability Theme	Classification (i.e. component zones)	Relative Vulnerability	Vulnerability Score
Presence of low permeability surface layer	0 – 5 m thick	High	3
	5 – 10 m thick	Low	1
	Non-aquifer	Negligible	0
Geological units	Karabaltinsky	High	3
	Panfilovsky	Moderate	2
	Ala-Archinsky	Low	1
	Neogene alluvial inliers to south of city (non-aquifer)	Negligible	0
Depth to groundwater	0 – 5 m	Very High	4
	5 – 10 m	High	3
	10 – 50 m	Moderate	2
	>50 m	Low	1
	Non-aquifer	Negligible	0
Surface hydraulic conditions	Zone of influent rivers	Moderate	2
	Recharge zone, no influent rivers	Low	1
	Groundwater discharge zone and non-aquifer	Negligible	0

Table 2. Groundwater vulnerability map classification

Sum of 4 component theme scores	Vulnerability classification of upper part of aquifer system
0	Non-Aquifer
1-6	Low
7-8	Moderate
9-11	High
12	Extreme

This final stage of GVM production helps simplify the map for policy use without losing the underlying cumulative hazard principle. The results, shown in Figure 1, divide the city and its suburbs into five principal areas:

1. The northern half of the city (north of a line approximating to Prospekt Chuy/Prospekt Zhibek Zholy) overlies aquifer of low vulnerability. This is due to the presence of a thick low permeability surface layer, the frequency and thickness of aquitard layers and the low downward vertical head gradients. All of these features protect the producing horizons from penetration of contaminants that may be present in urban recharge.
2. Further south, the presence of a high water table across central parts of the city makes the system sensitive to change. As the low permeability layer thins and the aquitards become subordinate, the vulnerability rapidly changes to moderate and then high in an east west belt either side of the main railway. This central area appears complex because the edges of constituent polygons in several component maps interact to give zones with slightly different cumulative point-score. A subsequent map developed for planning/policy purposes simplified this central zone
3. Once the ground surface starts to rise more steeply from the old airport across the piedmont plains, the water table becomes much deeper and the vulnerability reduces to moderate as far south as the inliers of Tertiary alluvium. There are however linear high vulnerability features in the form of the channels of the Ala-Archa and Alamedin rivers, whose highly permeable beds are conducive to river leakage
4. Further south the valley of the Ala-Archa River narrows into the highly productive alluvial fill tapped by the Orto Alysh well field. As the water table rises, the vulnerability class increases in these exposed highly permeable deposits to High and locally to Extreme along the axis of the river channel (where pumping-induced influent conditions certainly occur). This area without doubt constitutes the most vulnerable part of the Bishkek

aquifer system, and it also coincides with the city's most important wellfield resource.

5. Even further south as the ground continues to rise, the water table becomes deeper and vulnerability rating lessens, although it should be noted that this area remains sensitive because it lies within the composite capture zones of the production boreholes comprising the Orto Alysh well field.

3. Modifying the Ground Water Vulnerability Assessment - The Ψ Index

Although parametric rating/index-and-overlay systems (used for ground water vulnerability assessment on Bishkek territory) are the most widely used method of classifying intrinsic aquifer vulnerability, they suffer from the criticism that the systems are subjective. This is because the indices ascribed are comparative, and although rational, cannot be quantitatively defined. Partly to test whether a different, less subjective approach can be employed and partly to extend vulnerability considerations to the assessment of the impact on lower members of a multi-aquifer a new, analytical, method of assessing vulnerability was developed

The need arises because intensive pumping from a dense array of private or public wells either distributed across a city, or in intensively exploited well fields, can induce significant head differences between a shallow unconfined aquifer and a deeper semi-confined aquifer, causing the leakage of polluted water down from the shallow aquifer. In one study of the groundwater-dependent city of Santa Cruz in Bolivia (British Geological Survey, 1997a) such vertical leakage had reached 90 m after about 30 years. In another study of the city of Hat Yai in Thailand (British Geological Survey, 1997b) polluted near-surface water in a highly stratified aquifer took typically 35-45 years to migrate from the shallow water table to the semi-confined aquifer. Thus even apparently well-protected deep aquifers can eventually be prone to degradation by persistent and mobile contaminants.

The method seeks to quantify the vulnerability to pollution of an imaginary well in a leaky aquifer system by estimating leakage to that well from the overlying saturated layer. Steady state is reached when leakage from the upper and lower layers and lateral inflow match flow to the well. A vulnerability index (in effect an aggregated aquifer flow parameter) could therefore be constructed based on the fraction of downward leakage in the total abstraction volume of the well. The index characterizes the triaxial ratio between filtration resistances. The higher the index is, the greater the proportion, at steady state, of leakage Q_1 (Figure 2) from the upper layer (presumed polluted) in the pumped water, and so the higher the vulnerability.

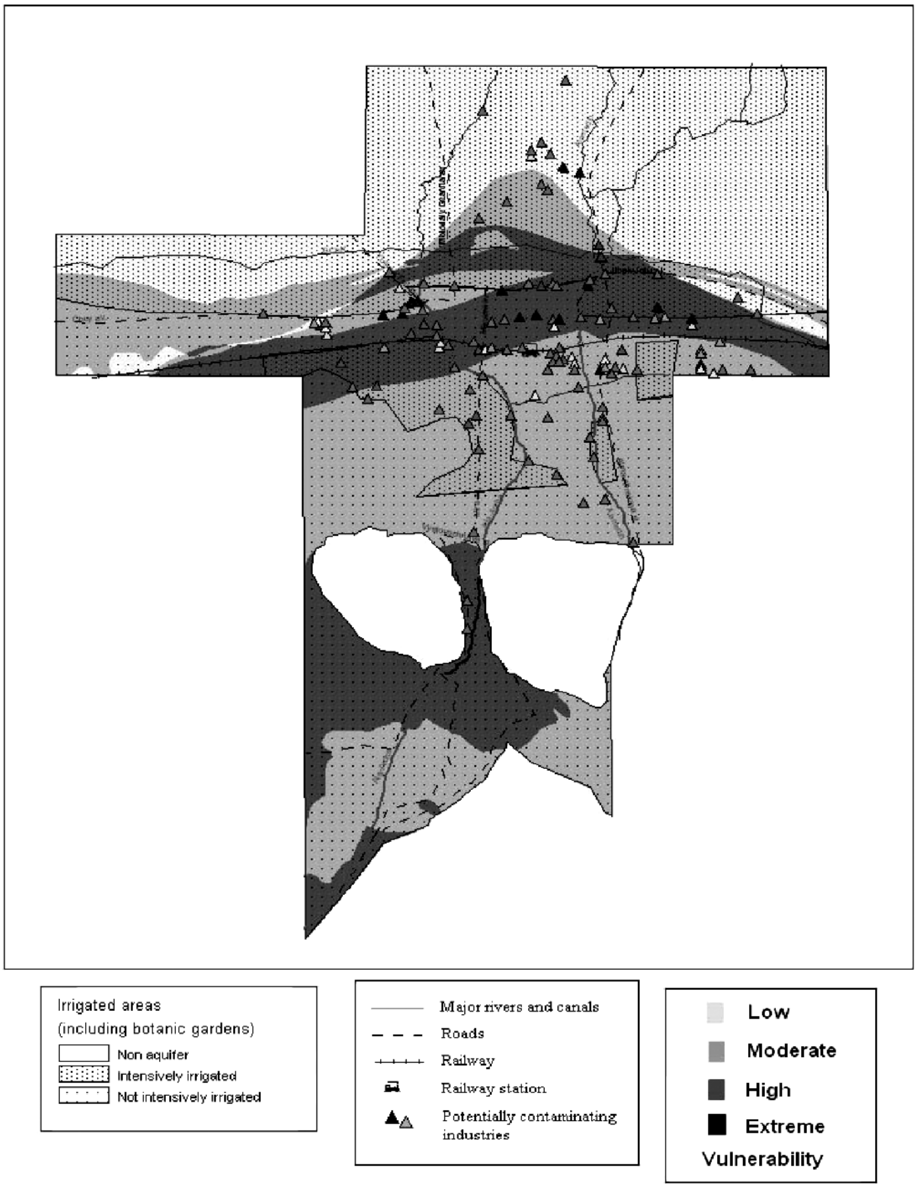

Figure 1. Groundwater vulnerability and potentially hazardous activities, Bishkek.

The index is defined as:

$$\Psi = \frac{Q_1}{Q} \tag{1}$$

where;

$$Q_1 = \int_0^R 2\pi\, r\, Kv \frac{Hv - H(r)}{Mv}\, dr \tag{2}$$

where Ψ is the vulnerability index (aggregate parameter of aquifer flow or Ψ index), Q is the abstraction rate, R is the radius of considered zone (usually 100-200 meters, approximating to zone where most leakage occurs), Kv is the vertical permeability of upper leaking layer, Hv is the head in upper layer, H(r) is the head in aquifer as function of distance from centre of the well, Mv is the saturated thickness of upper leaking layer and T is the transmissivity of the aquifer.

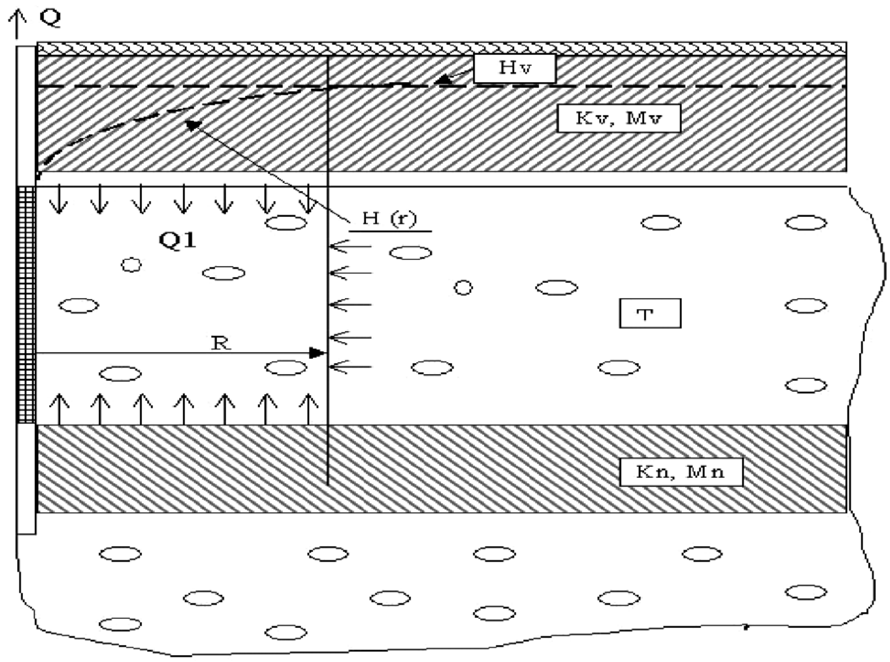

Figure 2. Typical filtration scheme with imaginary well in upper aquifer.

H(r) is the solution to the following problem:

$$T\frac{1}{r}\cdot\frac{d}{dr}\left(r\cdot\frac{d}{dr}\cdot H(r)\right)+\left(Kv\cdot\frac{Hv-H(r)}{Mv}+Kn\frac{Hn-H(r)}{Mn}\right)=0 \tag{3}$$

$$\lim_{r\to\infty} H(r)=\text{const} \qquad \lim_{r\to 0} 2\cdot\pi\cdot r\cdot T\cdot\frac{d}{dr}H(r)=Q \tag{4}$$

$$H(r)=\left(-\frac{Q}{2\cdot\pi\cdot T}\cdot K0\left(\sqrt{P1}\cdot r\right)+\frac{P2}{P1}\right) \tag{5}$$

where;

$$P1=\frac{1}{T}\left(\frac{Kv}{Mv}+\frac{Kn}{Mn}\right) \tag{6}$$

where K0(x) is zero order modified Bessel function of the second kind. This kind of solution has been used previously for well calculations in layered aquifers (Bochever, 1976; Hantush and Jacob, 1955). Taking into consideration the above derivation:

$$\Psi(R)=\frac{Kv}{T\cdot Mv\cdot P1}\cdot\left[1-R\cdot\sqrt{P1}\cdot K1\cdot\left(\sqrt{P1}\cdot R\right)\right] \tag{7}$$

where K1(x) is the first order modified Bessel function of the second kind. Thus a **Ψ** index can be constructed ranging from 0 (no leakage) to 1 (all abstraction derived from local leakage). The **Ψ** index can then be used instead of two of the criteria (thickness of low permeable upper layer and expert evaluation of geological unit) to better approximate the likely hydraulic response under pumping conditions. The other two vulnerability criteria from the procedure described earlier (depth to water table and surface hydraulic conditions) are then used together with the **Ψ** index, which can then be scored comparatively in the same way as previously, in terms of comparative vulnerability. An example scoring table is shown in Table 3.

Table 3. Example vulnerability scoring using **Ψ** index

Ψ	Vulnerability Score	Comparative Vulnerability
0.0-0.1	0	Non-aquifer
0.1-0.2	1	
0.2-0.3	2	Low
0.3-0.4	3	
0.4-0.5	4	Moderate
0.5-0.6	5	
0.6-0.7	6	High
0.7-0.8	7	
0.8-0.9	8	Extreme
0.9-1.0	9	

The method can be used quantitatively if the aquifer physical properties are known (vertical permeability of upper [leaking] and lower [leaked-to] aquifer, thickness and piezometric contours for each aquifer so as to derive average heads). If these are not available, estimation or interpolation from similar settings can give an approximate answer.

4. Applying the Ψ Index in a Vulnerability Assessment - Chu Valley Example

This method has been applied to a vulnerability assessment of the entire Chu Valley, whose Quaternary aquifers have 13 geological units (Table 4). Hydrogeology and water usage data came from the Kyrgyz Hydrogeology Survey Chu Basin Water Economy Department and other sources (Galanin et al., 1974; Galanin et al., 1982; Karpachev et al., 1991; Levchenko et al., 1967; Shestakov et al., 1980). The aggregated vulnerability classes used the Ψ index and the two other criteria employed for the Bishkek study for all the Kyrgyz part of the Chu Valley, Table 5. The resultant groundwater vulnerability map of the Chu Valley is shown in Figure 3.

The map can be used for development planning for towns and industry. In the north part of the valley there are several patches of very low ground water vulnerability, due to the presence of a thick upper loamy layer and comparatively deep ground water levels. The high vulnerability ribbon along the boundary between discharge and recharge zones is of particular planning importance especially for decisions on the siting of industrial plants and for policies to control agricultural pollution problems (agrichemicals, animal wastes management etc.).

A subsequent research study of the Bishkek part of the Chu Valley aquifer system using environmental tracers (Morris et al., 2005) confirmed the high vulnerability of the piedmont area, where vertical infiltration rates of 5-10 m/year were demonstrated and induced leakage from river and canal channels under intensive pumping conditions was deduced to be an important component of recharge. In these circumstances, boreholes with deep screen settings do not necessarily abstract old water, an important practical consideration when protecting a groundwater resource for drinking water purposes.

Table 4. Point scoring system for the Chu Valley GVM using ψ index

Groups of natural parameters	Classification of natural parameters	Vulnerability Score
Ψ index	Karabaltinsky (IV)	4
	Novotroitsky (V)	5
	Panfilovsky (VII)	5
	Ivanovsky (VI)	4
	Borolday-Tokmaksky (I)	5
	Ala-Archinsky (VIII)	3
	Chatkul-Stavropolovsky (IX)	2
	Sargow (XIII)	1
	Atbashinsky (XI)	1
	Georgievsky (XII)	2
	Issyk-Atinsky (X)	2
	Tokmak-Chumyshsky (II)	4
	Chumysh-Tashutkulsky (III)	2
Depth to ground water level	0-5	4
	5-10	3
	10-50	2
	>50	1
Surface hydraulic conditions controlling recharge potential	Zone of river infiltration	2
	Recharge zone	1
	Discharge zone	0

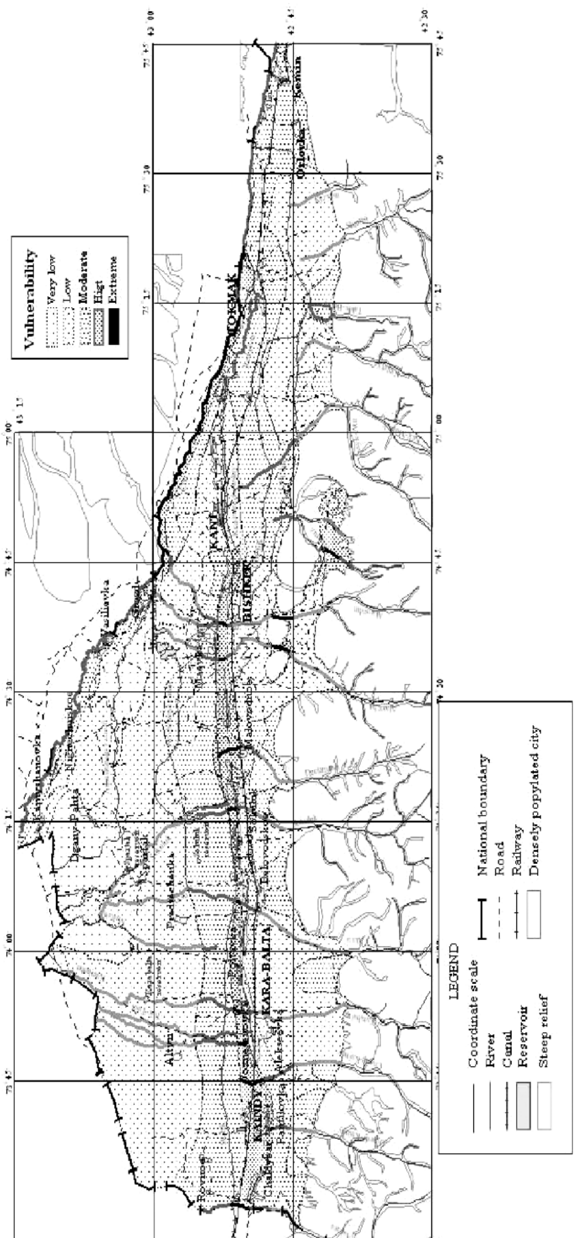

Figure 3. Schematic map of the Chu valley with ground water vulnerability assessment.

Table 5. Groundwater vulnerability map classification using aggregated scores from Table 4

Sum of 3 component theme scores	Vulnerability classification of upper part of aquifer system
1-4	Very low
5-6	Low
7-8	Moderate
9-11	High
12	Extreme

5. Conclusions

The **Ψ** index used in conjunction with other measured criteria such as depth to water table offers the opportunity to reduce some of the subjectivity that enters inevitably into the assessment of aquifer intrinsic vulnerability. An example is shown where the necessary aquifer characteristics have been recorded from previous water resource studies, enabling the index to be applied to an extensive regional aquifer system (the Chu Valley). The system could be more widely employed, especially in intermontane valley systems, where multiple aquifers are common.

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SURFACE/SUBSURFACE INTERACTIONS: COUPLING MECHANISMS AND NUMERICAL SOLUTION PROCEDURES

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Abstract: The interactions between surface and subsurface waters have long been an important topic in hydrological research. In general, these interactions are considered to be one of the most difficult areas of the discipline, particularly for the modeler who intends to simulate the dynamic relations between these two major domains of the hydrological cycle. In essence, one major complexity is the spatial and temporal variations in the dynamically interacting system behavior. The proper simulation of these variations requires the need for providing an appropriate coupling mechanism between the two components of the system. This study discusses the fundamental differences between the numerous coupling techniques that the hydrologic modeler can use to couple these two components. The details associated with their numerical solution procedures as well as the selection criteria of the most suitable coupling technique are also presented with particular emphasis on the spatial and temporal scales of subprocesses, the accuracy of the output required, and the numerical and computational complexity allowed.

Keywords: interactions between surface/subsurface domains; coupling mechanisms; numerical techniques; simultaneous coupling; iterative coupling; non-iterative coupling

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1. Introduction

The analysis of the interactions between surface and subsurface waters has become an important topic in hydrological research in the last two decades. For quite a long time, these inherently linked systems are artificially treated as two separate domains in order to provide the simplicity required in understanding the complexities of the hydrologic cycle (Gunduz, 2004). Today, science has reached to a point where the details of these two systems are uncovered to a certain degree and the focus is geared towards identifying the interactions between them. In general, these interactions are considered to be one of the most difficult areas of the discipline, particularly for the hydrologist who intends to simulate the relations between the two domains. The difficulty lies not only on the accurate formulation of the interactions but also in the temporal and spatial variations in the dynamically interacting system behavior (Gunduz and Aral, 2005). Therefore, the proper simulation of these variations requires the need for providing the appropriate coupling mechanisms between the surface and subsurface components of the system.

The accurate coupling of the two domains is indispensable when the overall hydrology of the system is to be analyzed (Sophocleous, 2002). The volumetric and mass fluxes that exchange between the surface and the subsurface provide the linkage and the dynamically interacting behavior of the hydrologic system. Without these fluxes transporting mass and volume in between, the hydrologic cycle is not complete and the surface processes are not linked to the subsurface processes. As there are no analytical techniques available for the solution of these interactions in real-world problems, the modeler is generally forced to use numerical methods for simulating the coupled behavior of the two systems. Consequently, it is important to mention that these coupling procedures are to be defined precisely and the associated numerical solution techniques are to be tailored for best accuracy and performance.

Based on these fundamentals, this study is focused on presenting an overview of the coupling mechanisms between the surface and subsurface domains with regards to flow and contaminant transport. All available techniques are described in details with their advantages and disadvantages. The numerical solution techniques associated with each technique are then discussed with particular emphasis on the potential drawbacks of each one of them and the spatial and temporal scales of hydrological subprocesses. The paper also presents the fundamentals of the criteria used in selecting the appropriate coupling technique based on the requirements of the particular study such as the required accuracy of the output, the data needs and the computational power requirements as well as the feasibility of implementing the particular technique as opposed to others available.

2. Fundamentals of Coupling Mechanisms

Generally, the coupling of surface and subsurface processes is done at a number of interfaces. These interfaces are defined to be the boundaries where surface flow/transport is in direct contact with subsurface flow/transport. Based on this description, one could identify the surface processes as the flow/transport in rivers/creeks and over the land surface. Similarly, the subsurface processes are defined to be the flow/transport below ground surface. The subsurface processes can further be classified into the flow/transport in the unsaturated zone above the water table and in the saturated zone below the water table. Since each one of these subprocesses have different characteristics such as representative physical dimensions as well as spatial and temporal scales, their coupling can become quite a cumbersome phenomena. As an example, while one can model the entire subsurface as a variably saturated domain with the groundwater table coming out as a part of the solution, it is also possible to model it as two separate layers linked at the groundwater table interface. Furthermore, the physical dimensions of the subprocess can be reduced changing the entire understanding of the system and the modeling practice such as the case in surface flow that can be modeled as a one-dimensional event in a river with the assumption of depth and width averaging or as a three-dimensional event without averaging any one of the physical dimensions. Hence, the coupling procedure is strongly related to the fundamental model to be used in the simulations.

A commonly accepted method for hydrologic modeling involves the linkage of surface and subsurface flow processes at the ground surface and the river/lake bottom interfaces. The ground surface interface is generally known to be the most obvious interface and is defined to be the boundary where overland processes are linked to unsaturated zone of the subsurface domain via the infiltration/exfiltration flux. The direction of the interacting flux is not only dependent on the overland flow conditions but also a strong function of the level of saturation of soil moisture. The two overland flow initiation mechanisms (i.e. saturation from above and saturation from below) are strongly related to these interactions as well as other factors such as the topography, land cover/use, hydraulic conductivity of soil and rate of precipitation. Another major interface linking surface and subsurface flow processes is the river channel or lake bottom. Here, the seepage flux is responsible for providing the linkage between the two systems. The direction of the flux is a function of the relative values of groundwater head and river/lake water stage. It is also important to note that the interfaces mentioned within this study are considered to be 'zero-width' interfaces with no major thickness and hence are not analyzed as a separate layer. This study is based on the assumption that the subprocesses of surface and subsurface are linked at these zero thickness interfacial boundaries and the flow and mass transport between these boundaries are instantaneous.

Depending on the accuracy required and numerical and computational complexity allowed, there are numerous techniques that the hydrologic modeler would use to couple surface and subsurface components and to evaluate the interacting fluxes. From the most complicated to the most simple, these methods include (i) true simultaneous coupling (Gunduz and Aral, 2003a; 2003b, 2005), (ii) semi-simultaneous coupling (Gunduz, 2004), (iii) iterative (internal) coupling (Morita and Yen, 2002), (iv) non-iterative (external) coupling (Motha and Wigham, 1995); and, (v) sink function type coupling (also known as "no" coupling) (Akanbi and Katapodes, 1988). Except for the sink function type coupling, all four methods are based on linking partial differential equations defining surface and subsurface flow via infiltration and seepage fluxes as the internal boundary conditions. In sink function type coupling, however, infiltration is simulated with empirical equations that are derived from soil characteristics and is incorporated in the volume/mass balance as a source/sink term.

It must also be mentioned that the surface and subsurface domains are coupled very similarly for the analysis of flow and contaminant transport. Essentially, the coupling of contaminant transport is strictly linked to the coupling of flow processes. Once the magnitude and the direction of the interacting volumetric flux is determined as a function of space and time, the mechanisms that provide the mass transport could then be quantified and modeled. In essence, the advective portion of the transport processes requires that flow coupling is already established and the hydrodynamics of both domains are fully understood. Hence, the coupling of flow processes is the initial step to be completed for a successful modeling effort.

Regardless of the numerical discretization technique used (i.e. finite difference, finite element, finite volume and others), the coupling procedures discussed above enforce certain requirements on the numerical solution of the model. While these computational requirements are more strict and numerically demanding in the case of advanced coupling techniques such as the true simultaneous solution, they are relatively less demanding in simpler techniques such as the non-iterative and sink function type coupling.

3. True Simultaneous Coupling

The true simultaneous coupling method is the ultimate, most advanced method of interacting surface and subsurface flows and is based on the simultaneous solution of the surface and subsurface flow/transport matrix equations formed as a result of the spatial and temporal discretization within the same global matrix structure (Gunduz and Aral, 2003a; 2003b, 2005). The final global matrix equation that is formed as a result of the numerical discretization process contains the unknowns from both domains and thus, the global coefficient matrix and the global load vector sizes are naturally much larger than their discrete counterparts. As the interacting flux terms are incorporated in the governing equations of both the surface and the subsurface domains, the hydraulic interactions are automatically reflected in the matrix elements. The solution of the global matrix equation then yields the values of the unknown terms and the interacting flux values are easily computed from these values.

One of the most critical issues of the simultaneous solution procedure is the time step used in temporal discretization of the surface and subsurface domains. It is important to note that the simultaneousness of the solution requires that both domains are temporarily discretized with the same time step, which is the main reason why the solution of the global matrix requires much more computational time and power when compared to other coupling techniques. Despite its difficulty and computational complexity, this method is thought the best technique that mimics the natural phenomena (i.e. simultaneous presence of surface and subsurface processes) and provides the most accurate results. The simultaneous solution of the two sub-matrices within a single global matrix is a relatively new technique and is believed to become more popular as access to high performance computing becomes widespread.

4. Semi-Simultaneous Coupling

The semi-simultaneous coupling method is an extension of the true simultaneous coupling procedure that has arisen from a numerical necessity. It is used in the mass transport simulation where the advective flux simulation must be separated from dispersive flux simulation and other source/sink terms. It is well-known that the numerical solution of advective transport requires the use of explicit techniques and demonstrates significant errors when implicit methods are used. This is particularly the case where advection dominates other transport means mainly in highly advective transport of contaminants with sharp fronts, where the numerical methods start to lose accuracy and computational efficiency. While dispersion favors implicit solution algorithms with possible use of large time steps, advection modeling generally requires an explicit algorithm with time steps limited by the Courant number criteria. Hence, the two major contaminant transport processes essentially behave in a contradictory manner (Gunduz, 2004). This phenomenon is one of the biggest problems of the numerical solution of contaminant transport yet to be solved. Since dispersion modeling could also be done with an explicit algorithm, a fully explicit scheme for the entire advection-dispersion equation is possible. However, such a scheme would not allow the simultaneous solution for the coupled surface/subsurface transport equation as explicit schemes don't involve solution of matrix equations and is based on sequential treatment of nodes at each time step. In contrast, the matrix solution of implicit schemes is necessary to simultaneously solve the two transport systems.

Except the problematic advection component of the surface transport, all transport mechanisms could be efficiently modeled with implicit schemes that involve matrix solution. The only exception to this setup would be the problematic advection component that can not normally be solved using an implicit scheme. It is this motivation that forces the modeler to separate the two processes and solve them in two steps. Using a fairly recent development in the area that results in the formulation of the so-called 'split operator' approach, one can now separate the advection operator from the dispersion and the rest of the operators. Consequently, the problematic advection operator is isolated from the rest and is solved using the most suitable explicit scheme possible. Although this approach appears to be a violation of the principle of "simultaneous presence" of these processes in nature, it provides a very powerful tool to handle the numerical difficulties associated with highly advective transport problems. Essentially, this procedure provides a sound methodology that gives mathematically identical results to the traditional compact operator methods. Consequently, one could discretize the equation by evaluating the advection term explicitly in time and the remaining terms implicitly in time. Originating from this need, Gunduz (2004) has developed the semi-simultaneous coupling method in which advection is treated separately from all other terms of the coupled surface/subsurface transport simulation according to the operator splitting technique.

5. Iterative Coupling

The traditional iterative (internal) coupling is one of the most widely applied coupling procedure in the last couple of decades (Pinder and Sauer, 1971; Freeze, 1972; Akan and Yen, 1981; Morita and Yen, 2002). In iterative coupling, the equations of surface and subsurface flow/transport are solved separately within their separate matrix equations but iteratively at each time step of the solution. Since each system is solved independently with respect to the numerical solution procedure, it is possible to use different time steps for surface and subsurface components based on the different temporal scales of each one of them. Consequently, while the more dynamic surface flow must be solved with a smaller time step, the relatively more static subsurface flow can be solved with a larger time step. This procedure allows that several surface time steps are evaluated within each subsurface time step. Like any iterative solution procedure, the iterative coupling requires the use of a pre-determined tolerance value below which the solution is assumed to converge. Hence, the technique can provide accurate solutions at the expense of computational cost.

6. Non-Iterative Coupling

In non-iterative (external) coupling, the surface and subsurface components are again solved separately at the same time step but in a non-iterative fashion (Smith and Woolhiser, 1971; Abbott et al., 1986; Motha and Wigham, 1995). In this technique, the surface flow model is generally solved first with time steps equal to or less than the time steps of the subsurface flow model. The results of the surface flow model are used to compute the interacting flux and this flux is then used in the solution of the subsurface flow model providing the coupling between the two domains. Once the solution procedure of the subsurface component is completed at the same time step, the control is progressed to the next time step without entering an iterative loop in which the model tries to satisfy the convergence of common flow variables such as the case of iterative coupling procedure. Since no convergence check is done, the solution is less accurate but comparably faster than the iterative coupling (Gunduz, 2004). Even though the accuracy of the solution from a non-iterative coupling technique is less than the solution from an iterative technique, this method has found wide applicability among modelers due to its comparably less computational time requirements.

7. Sink Function Type Coupling

Finally, the sink function type coupling is regarded as a further simplification of non-iterative coupling where interacting flux is now considered as a sink/source for the surface flow component. Due to its computational ease, there exist many models that used sink function type coupling such as the works of Akanbi and Katapodes (1988), Tayfur et al. (1993), Esteves et al. (2000) and Yan and Kahawita (2000). In all models developed by these researchers, the interacting flux is modeled using a semi-empirical algebraic equation and does not involve the solution of a partial differential equation. Hence, it is much simpler and numerically less demanding. In general, Horton, Philip or Green and Ampt formula is the method of choice for evaluating the infiltrating/exfiltrating flux. Moreover, the subsurface component is not even modeled with models that apply sink function type coupling. In rare cases where it is modeled, infiltration is included as a source to groundwater flow without solving the surface flow model. In this regard, it is clear that there is no direct link between surface and subsurface components and sink function type coupling is therefore known as the "no-coupling" approach (Gunduz, 2004).

8. Selection of the Appropriate Coupling Technique

Most of the time, the selection of the coupling technique is based on the limitations of computational and data resources as well as the objectives of the study. This is one of the reasons why the most primitive sink function type coupling is still finding wide application area in the field of hydrologic modeling. Numerically, it has no extra cost on the modeler and is often in harmony with the limited data available for a particular application. Furthermore, it is very convenient when the focus is only at one domain and the interest on the other domain is only limited to compute some approximate effect on the domain of focus. The non-iterative coupling, on the other hand, is the minimum level of coupling that must be implemented when both domains are equally important for the modeler. Since this technique does not iteratively improve the solution, it does not provide a precise solution but is generally acceptable due to the slow response of subsurface processes when compared to the surface counterparts. Consequently, the contribution from surface processes is not assumed to predominantly influence the conditions in the subsurface within a time step thus making it unnecessary to iteratively improve the solution. When such influences are crucial for the overall hydrology of the system, the modeler must implement an iterative coupling algorithm. In this method, the interacting fluxes are significant and have a direct influence on the overall volume/mass balance of either domain within a single time step. Without iteratively improving the unknowns in a time step interval, the model can not establish a converged solution particularly in highly dynamic conditions. Being a fairly sophisticated technique with moderate computational requirements, the iterative coupling algorithm is implemented by many modelers. On the other hand, the true simultaneous and semi-simultaneous coupling methods are the most sophisticated coupling mechanisms that require significantly high computational power due to the solution of larger matrices. However, this cost is counterbalanced with more realistic and accurate results since the process is essentially a replica of the natural law of simultaneous presence. With the wide-scale availability of the ever increasing computational power and easy access to spatially varied data, the popularity of the simultaneous solution techniques will be much higher in future. Until then, it will be the major dilemma of the modeler to choose between coupling techniques of higher accuracy and techniques that demand less data and computational power.

9. Scale Issues

In general, the term "scale" refers to the characteristic spatial or temporal dimension at which processes can be observed and characterized to capture the important details of a hydrologic event. Different scales of space and time govern the physical flow and transport phenomena in the hydrologic cycle as shown in Table 1. For integrated surface/subsurface models, these scales vary by several orders of magnitudes in terms of the computational step size, the simulation extent that is necessary to capture the important aspects of the hydrologic process modeled as well as the proper scales that are necessary to interpret the input data. These issues create additional problems within the numerical solution of coupled models and have become a major research area in recent years.

Table 1. Spatial and Temporal Scales of Hydrologic Processes (Gunduz and Aral, 2003a)

Surface/subsurface process	Spatial scale (cm)	Temporal scale (sec)
Unsaturated zone groundwater flow	$10^{-2} – 10^{+1}$	$10^{-1} – 10^{+2}$
Overland flow	$10^{+1} – 10^{+4}$	$10^{-1} – 10^{+2}$
River flow	$10^{+3} – 10^{+6}$	$10^{+2} – 10^{+5}$
Saturated zone groundwater flow	$10^{+3} – 10^{+6}$	$10^{+3} – 10^{+6}$

Integrating the processes from two extremes with respect to time and space scales necessitates high computational power and detailed data in coupled modeling. Particularly, linking unsaturated zone models with saturated zone models via simultaneous solution enforce small time and spatial discretization that in turn limit the application extent of the model. Similarly, coupled river-overland flow models also face the difficulty of different temporal scales. As discussed by Gunduz (2004), the problems associated with scale issues are the major concerns of numerical models and are still a challenge for the modeler.

10. Conclusions

The fundamentals of the interactions between surface/subsurface domains are analyzed with respect to the coupling mechanisms and the associated numerical solution procedures. Accordingly, the true simultaneous coupling mechanism is considered to be the most sophisticated technique that mimics the natural physical phenomena despite the difficulties associated with the additional computational power and data requirements. Until computational power and data availability reaches to adequate levels and the scale issues are resolved, iterative coupling technique is believed to serve as the only viable alternative.

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GROUND-SURFACE WATER INTERACTIONS AND THE ROLE OF THE HYPORHEIC ZONE

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Abstract: The hyporheic zone describes a region beneath and lateral to the bed of a stream where groundwater and surface water interact. Although the existence of this zone is well recognised, the flow dynamics and mixing processes within the zone are not well understood. To investigate hyporheic zone behavior, numerical groundwater flow models were developed with MODFLOW and calibrated using data collected at a study site on the Magpie River, near Wawa, Ontario, Canada. These models were used to examine the uncertainties of hyporheic zone behaviour at pool-riffle sequences and the response of the hyporheic zone to stream flow regulation. The hyporheic zone was found to be complex and temporally sensitive to stream stage and to groundwater fluxes as determined by streambed permeability and the hydrogeological characteristics of adjacent aquifers. The size of the hyporheic zone was found to be inversely proportional to the flux of groundwater moving towards the stream, and rapid changes in river stage were determined to cause short-term reversals of flow within the hyporheic zone which have important implications on hyporheic zone organisms and their need to adapt to changing environmental conditions.

Keywords: hyporheic zone; groundwater; riffle; regulated watersheds

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1. Introduction

The hyporheic zone occurs in sediments found below the stream bed and within stream banks lateral to a stream channel. The actual extent of this region is defined in many different ways, normally as a function of biological, physico-chemical and ecological properties. In an ecological context, the hyporheic zone is described as the ecotone which bridges surface water and groundwater (Brunke and Gronser, 1997), while a more physical definition (Fraser and Williams 1998; Williams, 1989; Triska et al., 1989) considers the hyporheic zone as the saturated pore space in sediments beneath and lateral to a stream/ river channel, which is strongly influenced by the interchange of ground and surface water. Regardless of the definition used, the influx of stream water that carries nutrients, chemicals and organic matter through the hyporheic zone is essential to the organisms or "hyporheos" that permanently or temporarily inhabit the region (Williams and Hynes, 1974; Storey et al., 1999a, 1999b).

The ecological importance of the hyporheic zone has led to various studies focusing on both the ecological (Franken et al., 2001; Boulton et al., 1998) and physical dimensions (Stanley and Jones, 2000; Packman et al., 2004; Storey et al., 2003). In general, the studies have shown the hyporheic zone to be highly variable in size and very sensitive to changes in stream conditions such as differences in river stage (water level), seasonal fluctuations of potentiometric head in adjacent aquifers, and the hydraulic conductivity of the sediments.

2. Study Site

Research conducted by the University of Toronto in recent years has focused on the interchange of ground and surface water in the hyporheic zone, as well as the factors that determine the nature of the interchange. Early studies were conducted at a field site located at the Speed River, near Guelph, Ontario, Canada (Figure 1) which was heavily instrumented to produce a data record stretching over a period of almost 10 years. Storey et al., (2003) showed that factors such as the head difference between upstream and downstream of a riffle, the hydraulic conductivity of the streambed sediments and the rate at which groundwater enters the stream, all have important influences on hyporheic exchange flows. The Speed River studies were performed under the assumption of steady state flow, a condition that is rarely achieved in nature. To investigate transient flow behavior and the role of variable river stage, further studies were carried out at a field site on the Magpie River near Wawa, Ontario, Canada (Figure 1). The primary objective of this work was to understand the impact that short-term, human induced changes in river stage can have on flow within the hyporheic zone, thus enabling development of management tools to minimize impacts on the hyporheos.

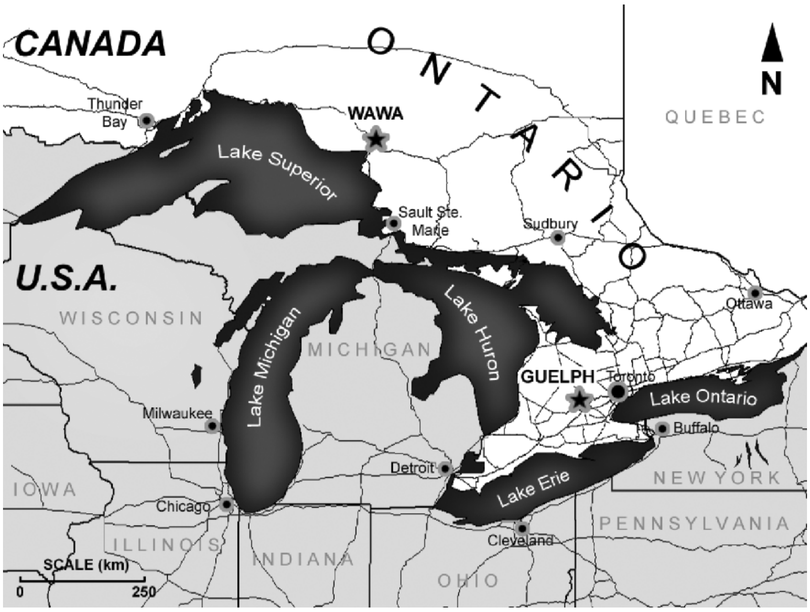
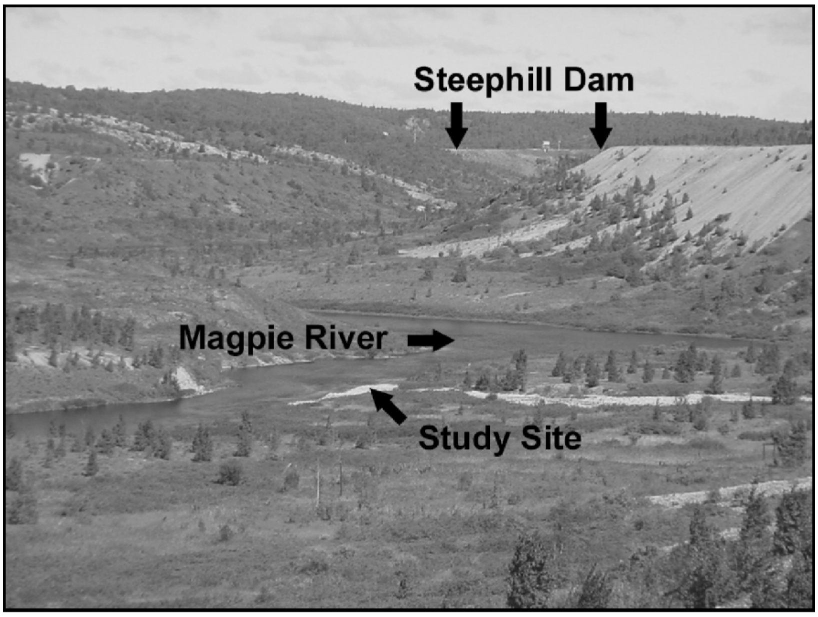
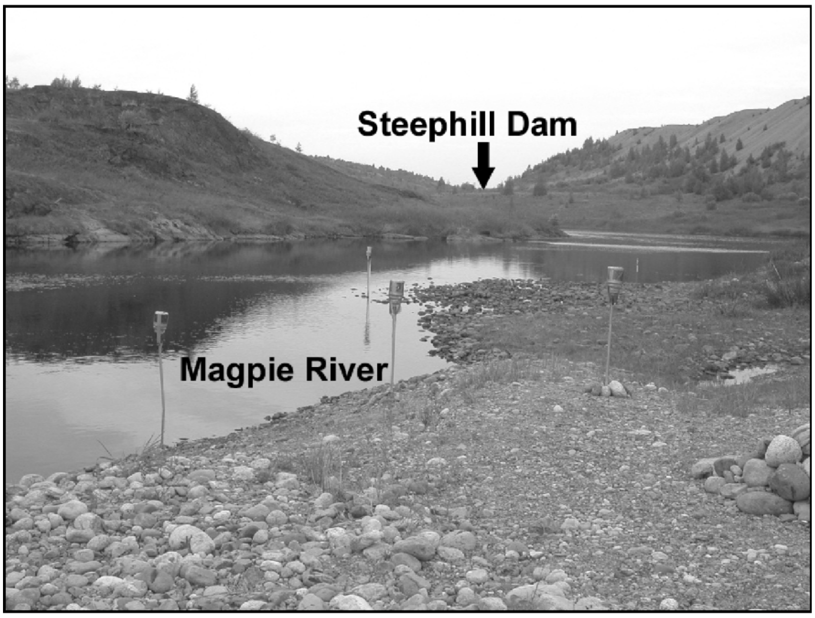


Figure 1. Locations of Speed River site (near Guelph, Ontario) and Magpie River site (near Wawa, Ontario).

The Magpie River is located within the Lake Superior watershed and is a highly regulated river with three hydroelectric dams along its reach. All three dams are located upstream of the study site, but only the Steephill Dam influences water levels at the site. Although there is always continuous flow in the river downstream of the Steephill Dam, water from the reservoir upstream of the dam is released approximately every 24 hours to generate hydroelectric power. The release of water creates a rapid elevation in stream stage. Figure 2a is a view to the north showing the study site in relation to Steephill Dam, and Figure 2b shows the piezometer installation.



a) Site in relation to Steephill Dam



b) Piezometer installation

Figure 2. The Magpie River study site.

3. Models

3.1. REGIONAL FEFLOW MODEL

During the first stage of the study, a 3-D finite element model was developed using FEFLOW to represent regional groundwater flow in a sub-catchment of the Magpie River below the Steephill Dam (Figure 3). The purpose of this model was to develop an understanding of the regional water balance and key boundary conditions, particularly in the immediate vicinity of the Magpie River. The model parameters were assigned to a super element mesh that was refined in close proximity to the river. Twelve layers were included in the model, the three most important layers representing sand and gravel, well fractured bedrock and weakly fractured bedrock. These layers were assigned values of 1 x 10^{-1} m d^{-1}, 1 x 10^{-5} m d^{-1}, and 1 x 10^{-7} m d^{-1} respectively for hydraulic conductivity in the horizontal direction (K_x and K_y). K_z values were set one order of magnitude lower. Once the FEFLOW model was calibrated, information on boundary conditions was incorporated into a series of more localized 3-D models developed using MODFLOW.

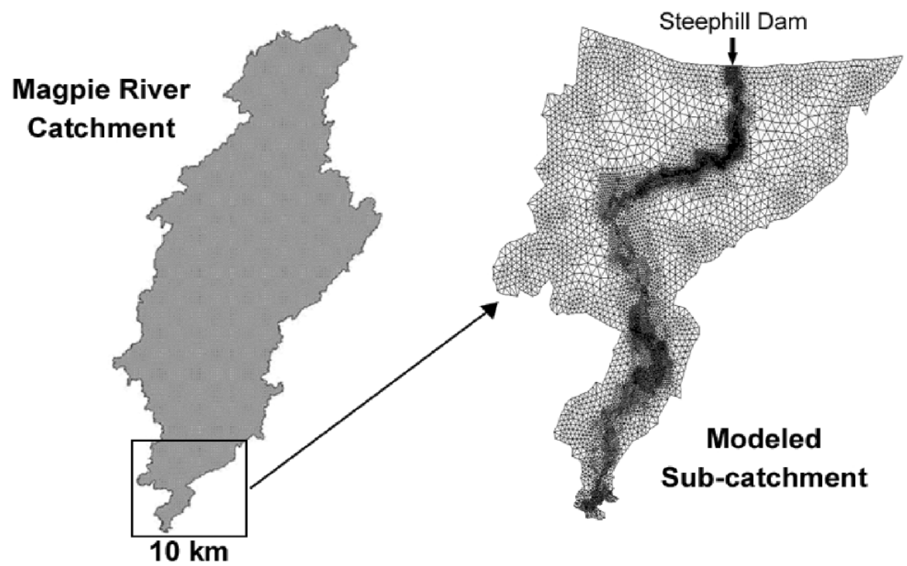


Figure 3. Model grid of the regional FEFLOW model of Magpie River area.

3.2. LOCAL MODFLOW MODELS

The 3-D finite difference MODFLOW models extended across a much smaller area than the regional FEFLOW model, but included a finer grid discretization, thus enabling the interaction between stream water and groundwater to be studied on a small, localized scale. The models were calibrated with data collected from pressure transducers installed in and adjacent to a 12 m long riffle of the Magpie River (Figure 2b) for two four-month periods in 2004 and 2005, each from mid-June until mid-October. The instruments recorded fluctuations in groundwater head within and adjacent to the riffle at 3 minute intervals.

3.2.1. *Riffle Model*

A schematic representation of riffle behaviour is shown in Figure 4. Just before the riffle, upwelling occurs from the groundwater to the stream. As the top of the riffle is approached downwelling occurs but then reverts back to upwelling on the down-slope side of the riffle. In the model, three successive riffles were simulated. These riffles were assumed to be about 1m high, 10 m in length and 50m apart. During the first set of model experiments the model was run at steady state. In later experiments, transient behavior was investigated.

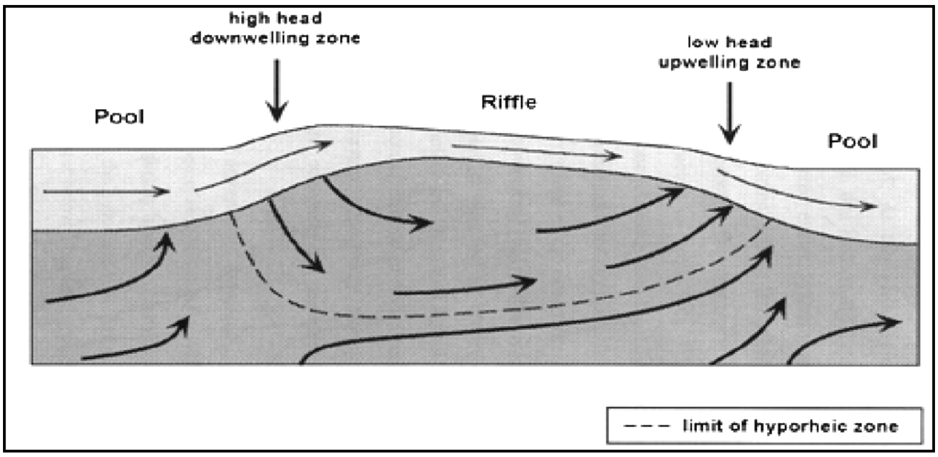


Figure 4. Schematic drawing of the hyporheic zone at a riffle.

The three-riffle model was 230 m in length, 110 m wide and 10 m deep. It contained 15 layers of various thicknesses and the horizontal grid discretization was 1m. The model included three conductivity zones representing a) the top

10 cm of the riverbed, b) the main riverbed and c) the surrounding geologic material. For the steady state simulations, river nodes and the aquifer boundary nodes were maintained by assigning constant head levels. The surface slope of the river was set to 0.002 with a rise and drop of 1 cm before and after each riffle to generate hyporheic flow.

A series of steady state simulations were performed for a range of hydraulic gradients towards the river. Results are shown in Figures 5, 6 and 7. In these figures, arrows represent flow directions and solid lines are equipotentials. With a relatively low hydraulic gradient (0.0001, representing a 1 cm head change over 100 m) net groundwater fluxes into the river are low and the hyporheic zone is quite pronounced (Figure 5). A moderate hydraulic gradient (0.001) reduces the size of the hyporheic zone significantly (Figure 6). When the hydraulic gradient is increased to 0.01, the hyporheic zone virtually disappears (Figure 7).

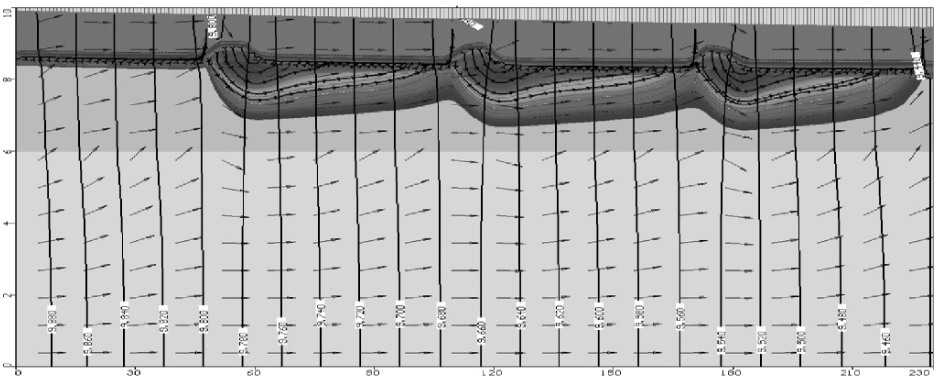


Figure 5. Extent of the hyporheic zones with hydraulic gradient towards the river equal to 0.0001. Model is 230 m in length and 10 m deep.

The three-riffle model demonstrates the influence of effluent flow on the development of the hyporheic zone. When the hydraulic gradient is high, groundwater flow to the stream is similarly high, which inhibits hyporheic zone exchange. Assuming steady state river stage conditions, the size of the hyporheic zone will have an inverse relationship with aquifer hydraulic gradient such that when the hydraulic gradient increases, the size of the hyporheic zone decreases. As discussed in Storey et al., (2003), seasonal variations in groundwater fluxes towards the stream can cause seasonal changes in the hyporheic zone. For example, in temperate climates, the hyporheic zone may only develop in summer when aquifer heads are low and baseflow contributions to the stream are minimal. During the fall when there is increased precipitation or during spring after recharge by snowmelt, hydraulic gradients to the river increase and the hyporheic zone may disappear. This seasonally dynamic situation is further complicated in regulated watersheds where drastic changes in the river stage can suddenly occur due to the opening and closing of sluice gates at control dams.

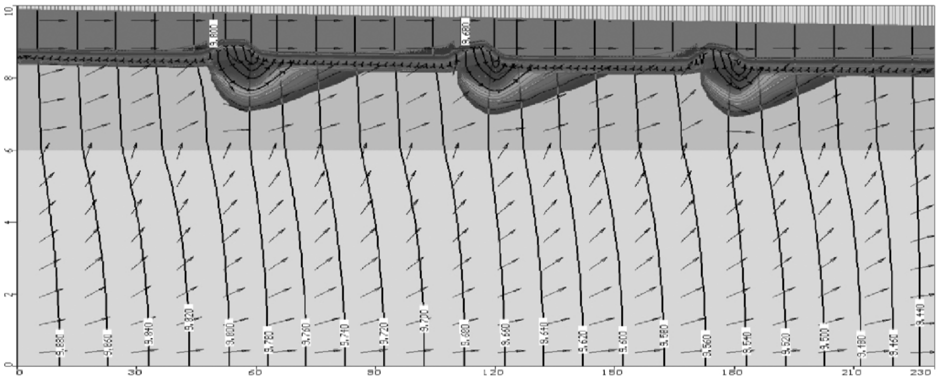


Figure 6. Extent of the hyporheic zones with hydraulic gradient towards the river equal to 0.001. Model is 230 m in length and 10 m deep.

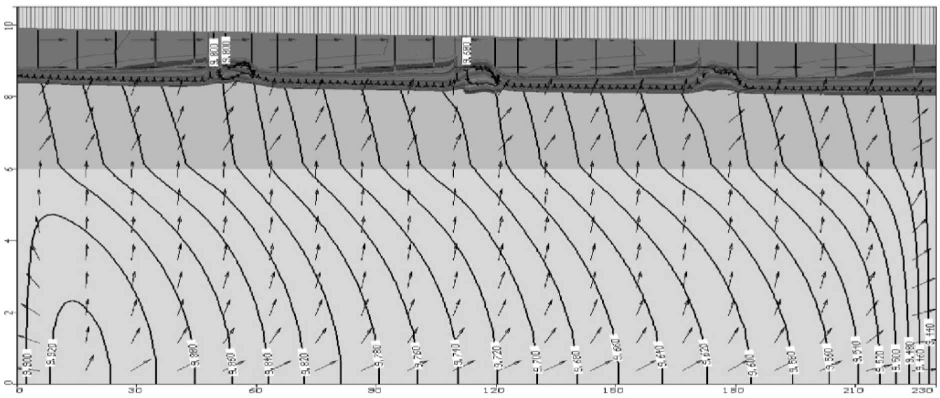


Figure 7. Extent of the hyporheic zones with hydraulic gradient towards the river to 0.01. Model is 230 m in length and 10 m deep.

3.2.2. *Regulated Watershed Model*

To better understand the effects of watershed regulation on the hyporheic zone, a transient finite difference model was developed using MODFLOW to simulate conditions at the Magpie River study site. The model was calibrated with data collected from pressure transducers installed in the riverbed to record groundwater potentiometric head and also from river stage data collected by Trent University at the same site. A small portion of these data is shown in graphical form in Figure 8. River stage varies over a much higher range than groundwater head, such that flow directions within the hyporheic zone may change dramatically over short time intervals. For example, downwelling occurs when river stage is higher than the groundwater head and upwelling occurs when river stage levels are lower than groundwater head levels.

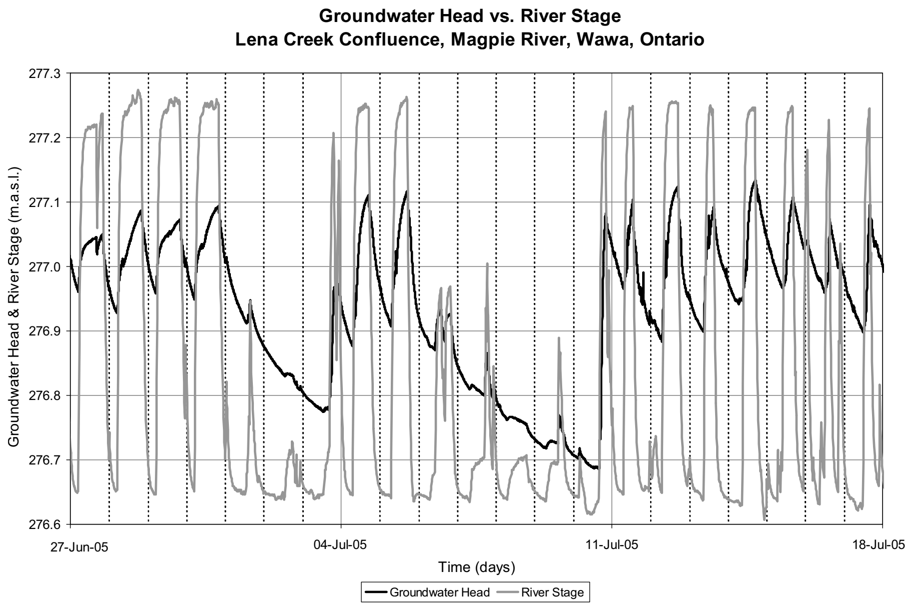


Figure 8. Changes in river stage and groundwater head showing upwelling and downwelling at a depth of 1m.

The transient MODFLOW model was developed using parameters and boundary conditions generated during calibration of the regional FEFLOW model. The model was 2000 m long, 70 m deep and 10 m wide and comprised six layers and three conductivity zones. In order to observe small scale changes in upwelling and downwelling in the hyporheic zone, the grid was refined in the river area of the model to provide a discretization of 1 m. Constant head conditions were applied along the lateral boundaries of the model and the river was simulated as constant heads that were raised and lowered by 0.6 m every 24 hours to represent the changing river stage levels recorded in the field data.

The model demonstrates that when the river stage is at its highest, downwelling occurs with river water flowing into the hyporheic zone from the river (Figure 9). When the river stage is at its lower level, hyporheic flows reverse, downwelling ceases and groundwater feeds the river.

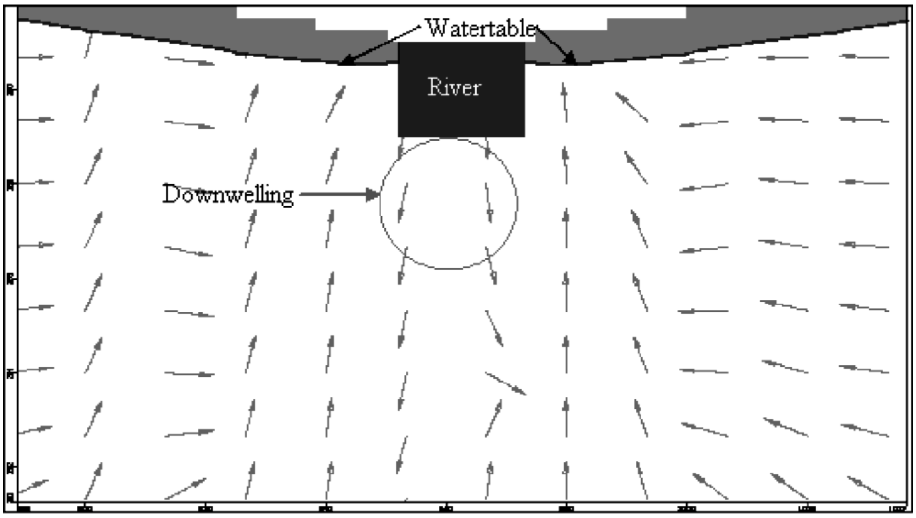


Figure 9. Model flow directions showing downwelling of water beneath the river during high river stage.

Figure 10 shows the relationship between the fluctuating river stage and the flux of water either entering or leaving the hyporheic zone. Five cycles are shown. When the river stage is at its peak of 290.6 m above mean sea level, very little groundwater enters the river and downwelling of surface water feeds the hyporheic zone. When the release of water from the dam ceases and the river stage is lowered to 290.0 m above mean sea level, the influx of surface water ceases, and the only observable flux is groundwater entering the river. Model results suggest that in such systems, the hyporheic zone is highly dynamic, growing during periods of elevated river stage and dissipating when the river stage is lowered. These observations are well supported by data collected in the field (Figure 8).

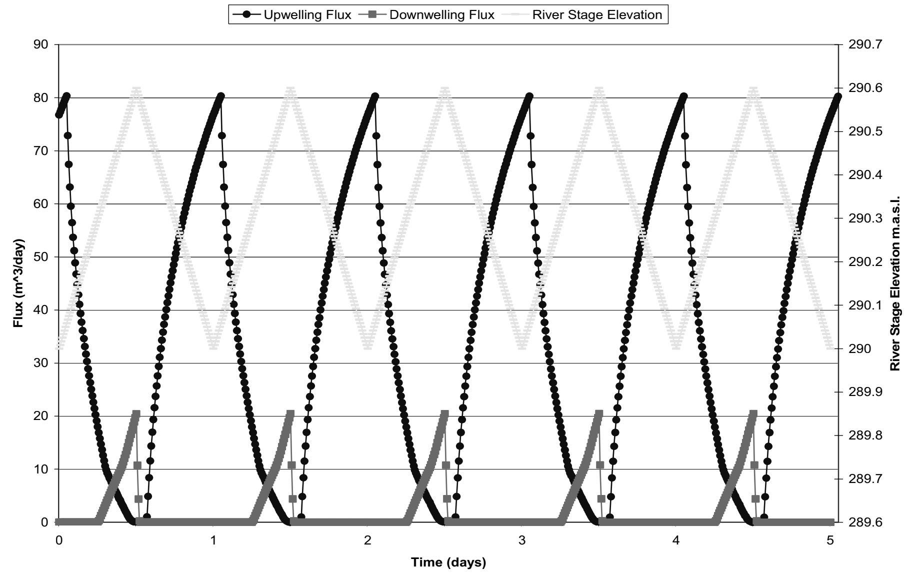


Figure 10. Upwelling and downwelling with 0.6 m river stage change.

4. Discussions

The use of numerical flow models, developed at different scales, to recreate the steady state and transient behaviour of the hyporheic zone, allows the influence of many variables to be examined and evaluated. The small-scale riffle model studies demonstrate the importance of potentiometric heads in the surrounding aquifer, and the resulting groundwater flow to the stream, on the size of the hyporheic zone. When hydraulic gradients towards the stream are high, groundwater fluxes are similarly high and the hyporheic zone virtually disappears. This may cause organisms that depend on the hyporheic zone to become stressed and leave the area in search of a more favorable refuge. Natural seasonal variations in hydraulic gradient and groundwater flows commonly occur in southern Ontario as described by Storey et al. (2003); thus, seasonal changes in the hyporheic zone must be expected. However, human-induced changes in aquifer heads can exacerbate the problem and, if excessive, could create a situation whereby the hyporheic zone disappears altogether, or stays present all year round. While the present study does not specifically address the extent to which human-induced changes to aquifers affect the hyporheic zone, it does highlight the need for further research into situations of this nature.

Large and rapid changes in river stage that occur in regulated watersheds may also have a significant impact on the hyporheic zone. Modeling supported by field data shows that extreme highs and lows in river stage caused by upstream hydroelectric dams may cause major changes in hyporheic fluxes and local reversals of flow directions. Organisms that are unable to adapt to these changes would struggle to survive, and additional research is necessary to better understand the ecological ramifications of rapid, human-induced changes in stream stage.

5. Conclusions

The hyporheic zone describes a region in sediments beneath and lateral to the bed of a stream where groundwater and surface water interact. This zone is not characterized by static, unchanging conditions, but rather by complex, dynamic interactions which make it difficult to fully understand. Improvements in this understanding have been accomplished by developing numerical groundwater flow models to simulate actual field conditions. The use of data from selected sites to calibrate these models ensures that the model simulations are realistic in relation to the processes that take place.

While streambed topographical features, such as pool/riffle sequences, are important in determining the location of hyporheic zones, groundwater flux is very influential in determining the size and extent of the hyporheic zone, with large groundwater fluxes into the stream having the potential to virtually prevent hyporheic zone development. Rivers and streams in regulated watersheds are further complicated by rapid and extreme changes in river stage, which in turn cause short-term changes in hyporheic zone behaviour including reversals in flow direction. Further work is required to fully understand this short-term transient behaviour and the nature and rates of hyporheic zone growth and collapse.

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GROUNDWATER MANAGEMENT AT IRRIGATED LANDS OF UZBEKISTAN AND ITS INFLUENCE ON ECOLOGICAL SYSTEM

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Abstract: An annual capacity of underground water resources in Uzbekistan is 66,342 thousand m^3 or 24.2 km^3. 86% water is formed in the overburden layers and connected to surface water. Total volume of underground water is 17079.94 thousand m^3/day. Fresh water having drinking quality (salinity is up to 1g/l) is 9 km^3/year. Underground water pollution also has increased. Comparing to 1965 underground water resources has reduced for 36% as the result of the man-caused factors. There are 4275 hectares irrigated lands in Uzbekistan. Over 50% of ground water in this area is situated in the layers up to 2 m depth. So, bad soil drain and low level of engineering drainage system has brought to rising of soil salinity and reduction of crop. The main feeding sources of ground water are a filtration of irrigated lands and different rate irrigation canals, precipitation and underground inflow of below-located layers. Ground water rising is caused by growth of irrigated lands due to population upsurge, traditionally accepted furrow watering practice, low technical rate of irrigation canals and horizontal and vertical drainage system. Total length of inter-farm irrigation network is 27619.7 km, on-farm network-167378.8 km. 62% of inter-farm and 79.5% of on-farm network have earthen channel. There are 136.7 th. km of drainage network, from which 29 th. km are main and inter-farm collectors, 107.7 th. km are on-farm drains (incl. 39.2 th. km of horizontal subsurface drainage). There are 9210 wells including 4214 vertical drains and 4996 wells are built for irrigation. At present water losses from main canals are 3197.3 mln. m^3 (13.2%); from inter-farm canals - 4931.3 mln. m^3 (20.4%); from on-farm canals - 8293.4 mln. m^3 (34.4%); field losses are 7724.2 mln. m^3 (31.0%). Total loss is 24146.2 mln. m^3. In this situation 20-23cu. km

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collector-drainage water is forming in the Uzbekistan territory. River water salinity in upstream is 1.5-3.0g/l, in midstream and downstream it fluctuates from 3.5-6.0 to 5-7 g/l. The half part of the collector-drainage water inflows to river and can be used again for irrigation in the lower located lands. Return water re-use is repeated several times along the river worsening river water quality. Irrigation water salinity in the midstream has reached to 1.0-1.1g/l, in downstream more than 2 g/l (against initial 0.2-0.3g/l). Annual economic damage in Uzbekistan caused by high location of ground water, soil and irrigation water salinity is over 920 million USD.

Keywords: groundwater; pollution; formation; irrigation lands; filtering; drainage; ameliorative regime; ecosystem; measurements; impact

1. Introduction

Natural resources of ground waters are assessed as 66342 thousand cubic meters per day or 24.2 cubic meters a year in Uzbekistan. 86 percent of it is formed at deposits of quaternary period and is directly related to surface water bodies. Total volume of ground waters is 17079.94 thousand cubic metes per day. Fresh ground waters (mineralization 1 g/l) are about 9 cubic kilometers a year. Relative to 1965 the quality of ground waters reduced. The amount is reduced to 36% due to technohen factors (Dukhovny et al., 2001).

Given that total irrigated area of Uzbekistan as 4275 thousand ha, on more than 50% of it the ground waters are at depth of less than 2 m, and due to insufficient natural drainage and poor artificial drainage this leads to Stalinization of soils and reduction of yield. The main sources ground waters at irrigated lands are infiltration from the fields, irrigation canals, rainfall and at some spot areas, inflow from lower stratums.

The main reason of rise of ground water level is increase of irrigated areas due to demographic rise of population. The other reasons are: practice of furrow and stripes; low technical condition of irrigation canals and poor technically condition of vertical and subsurface horizontal drains. Total length of interfarm irrigation canals is 27619.7 km, and onfarm canals - level of 16738.8 km. 62% of interfarm canals and 79.5% of onfarm network are unlined. At present, loss of water on main canals if 3197.3 mln. m^3 (13.2%) on interfarm canals - 4931mln. m^3 (20,4%) and on onfarm canals 8293.43 mln. m^3 (34.4%). Loss in field 7724.23 mln. m^3 (31%). The total loss is 24146.23 mln. m^3. at drained lands there are 136.7 thousand km of drainage network have been constructed. 29 thousand km of it belongs to main and interfarm collectors, 107.7 thousand km belongs onfarm drainage (including 39.2 thousand km of subsurface horizontal drains). There are 9210 wells including 4214 vertical drainage and 4996 well for irrigation (Ikramov, 2001a).

At current conditions there are huge quantities of collector- drainage waters (20-23 km3) are being formed, including 70 to 95% from ground water source. Their mineralization on upstream of rivers is 1.5 to 3.0 g/l, on middle reach 3.5 -6 g/l to 5-7 g/l. More then half of total volume of collector waters flow into the rivers and used for irrigation at downstream lands. Such repeated use may happen several times, thus rising mineralization of river, and harming the river ecological system. Mineralization of irrigation water in middle reach has reached 1.0-1.1 g/l, and at downstream at some periods it reaches 2 g/l and more (compared to natural 0.2-0.3 g/l) as stated by Yakubov et al., (2001).

Only total annual agricultural losses in Uzbekistan, due to shallow ground water table, Stalinization of soil and rise of mineralization of irrigation water are assessed as 919mln US dollars (JEF Agency, 2002).

2. Groundwater Management Principles in the Irrigated Area of Uzbekistan

Management of ground waters at irrigated lands is a main part of management of land reclamation regimes. Reclamation regimes are created by complex of hydromeliorative, agro technical, agrochemical and other measures aimed at formation of optimal conditions of soil forming process and ground water regime guarantying maximum yield with minimal expense and damage to environment (Ikramov, 2001b).

Reclamation measures (irrigation, leaching and drainage) impacting the soil formation process, directly effects the water-salt regime of soils and ground water. According to Reshetkino N. M. there are four types of reclamation regime can be created on irrigated lands. Theses are: hydromorphouse, semi-hydromorphouse, semi-automorphouse and automorphouse - characterized by different regime of ground waters, share on participation in soil formation and by feeding agricultural crops and also by specific structure of total and partial water - salt balances.

Types of reclamation regimes formed depending on share of ground waters on total waters consumption of agricultural crops, which at the same time depends on water-physical quality of aeration zone soils (texture, height and speed of capillary rise, water holding capacity etc.), type and phase of development of crops, the amount of water supply, draining capacity and irrigation technique.

Parameters, related to rational reclamation regime for the period of desalinization of lands (depth of ground waters by period of year, share of outflow from offtake of ground water from sawing stratum to lower well drained stratum, increase of water supply to field above evapotranspiration.) obtained from generalizing the experimental systems in different farming conditions, demonstrated in Table 1.

The parameters for the period after reaching desalinization of lands are given in Table 2. To justify the necessity for establishment of any regime it is necessary to make many optional forecast estimations to define the optimal depth of ground waters on annual basis, including the content and parameters reclamation measures, under which achieved favorable conditions for achieving high yield with minimal negative impact to ecological system.

The forecasting includes total water balance of reclamated territory, water - salt balance of aeration zone of irrigated field, salt balance of aeration zone, surface layer of ground waters, zone of raising flow and root zone. In order to make such complicated estimations there have been developed an algorithm and computers program by A. A. Adilov. The results of such estimations are given as diagram on Figure 1.

3. Main Measures for Reduce of Groundwater Negative Impact on Ecosystem

Organizational-technological for short perspective (2006-2011) with minimal capital investments:

- During transfer period with financial - economical difficulties in agriculture more worthwhile is furrow irrigation and irrigation by stripes, and improvement of water management in field. Selection of optimal elements of irrigation technique applicable to specific conditions (possibly transfer to short furrows with the length not more than 150-200 m, concentrated irrigation with duration not more than 1-1.5 day). With surface irrigation the special importance is land leveling. Such measures allow reduction of infiltration feeding of ground waters in the field. Progressive methods of irrigation (drip, sprinkling, discrete, high frequency irrigation) have to be established at representative areas as an experimental pilot sites
- Market changes in water economy complex. There are 2 possible ways: 1) charging for water services (partial and total charging for water), keeping state property of water sites; 2) privatization of water sites in farm and district levels with organizing Water User Associations as cooperatives and with participation of users, state, and local administration. At present, the preference is given to the second approach. This requires improvement of legislative base. An intensive preparation works are carried on first approach

Table 1. Parameters of reclamation regimes, recommended according to results of field experiment studies conducted by SANIIRI (efficiency of vertical drainage under different natural – farming condition of Uzbekistan and South Karakalpakstan) (Ikramov, 2001a)

Geomothological structure	Soil reclamation and hydrogeological conditions	Reclamation regime	Recommended depth of ground waters on annual basis. м				Relationship of volume of off take to intake -	Excess of supply to total evaporation %	Share of water consumption of crops covered by ground waters %
			X-XI	XII-II	III-V	VI-VIII			
Pre-Mountain plains mid-mountain valleys represented by two and multy stratums of alluvial and prolluvial complexes	**Heavy**. Moderately and highly saline lands at area of more than 50% with mineralized graound waters (10g/l). thickness ***m*** = 20-25 м; ***К*$_Ф$** = 0.1 м/сут Golodnaya Steppe, Fergana valley	Semi-automorphouse	3.5-4.5	1.4-1.5	2.2-2.7	2.7-3.5	 35-40 50-80	25-30	2-12
	Moderate. Moderate and highly saline lands at area of 30-50 % ***m*** = 15-30 м; ***К*$_Ф$** = 0.1-0.2 m/day. Zarafshan oasis, Fergana valley, Karshi steppe	Semi-automorphouse	3-4	1.4-1.5	2.2-2.5	2.5-3.0	 30-35 40-50 30-35	20-25	5-15
	Light. Moderate and slight saline lands at area of less than 30 %, ***К*$_Ф$** = 0.2 - 4 m/day Fergana valley, Zarafshan oasis	Semi-hydromorphouse	2.5-3.0	1.4-1.5	1.8-2.4	2.4-2.5	 35-45 30-35	15-20	20-40
Low desert plains, river deltas represented by single and multy stratum alluvial deposits	**Moderete**. Solonchak top layer (1-1.5 м) slightly saline. GWT 4-4.5 м; ***m*** = 4 - 10 м; ***К*$_Ф$** = 0.32m/day (middle and low reach of Syrdarya and Amudarya	automorphouse	>4				10-15	10-15	
	Slightly. and moderate saline soils and ground water ***m*** = 1.5 - 13 м; ***К*$_Ф$** = 0.14 - 6.9 m/day (lower reach of Syrdarya, Amudarya, small rivers)	hydromorphouse	2.5-3.0	1-1.5	1.5-1.8	1.8-2.2	40-45	25-30	

Table 2. Reclamation regimes and main criteria of reclamative proficiency for different types of soil profile. (During operational period of drainage) (Ikramov, 2001a)

Reclamation regime	Type of soil profile					$\frac{ET_Г}{ET_П^B}$	$\frac{Q_{\hat{I}}^{\hat{I}}}{\hat{A}+\hat{O}}$	$\frac{\hat{I}_G+\hat{I}_{\tilde{N}}}{\hat{A}\grave{O}_{I}^{\hat{A}}}$
	Thick relatively homogeneous sandy soils (fine-moderate, dune) / 1-в /	Sandyloam and slightly loam soils (0.5-1.0 m) above sand and pebble layers / 1-а /	Homogeneous slightly and moderately loam / 2-б/, / 3-а/, / 2-в/, / 3-в/	Sandy loam sand, above heavy loam and clay / 3-б/, / 4-б/	Heavy loam and clay / 4-а/, /4-б/			
	Depth of ground waters from ground surface, m							
Hydromorphouse	0.6-1.2 (0.9-1.5)	0.6-1.2 (0.9-1.5)	0.6-1.8 (0.6-2.1)	0.6-1.2 (0.9-1.5)		0.50-1.0	0.30	1.05-1.1
Semi-hydromorphouse	1.2-1.5 (1.5-1.8)	1.2-1.5 (1.5-1.8)	1.8-2.5 (2.1-2.8)	1.2-2.0 (1.6-2.3)		0.20-0.5	0.30	1.05-1.1
Semi-automorphouse	1.5-2.2 (1.8-2.5)	1.5-2.2 (1.8-2.5)	2.5-3.5 (2.8-4.2)	2.0-2.5 (2.3-2.8)	1.5-2.5 (1.8-2.8)	до 0.20	0.30	1.05-1.1
Automorphouse	More than 2.2 More than 2.5	More than 2.2 More than 2.5	More than 2.5 More than 4.4	Lower than 2.5 Lower than 2.8	More than 2.5 More than 2.8			

$\frac{ET_Г}{ET_П^B}$ - Share of ground waters in water consumption of crops;

ET^B - evapotranspiration during vegetation period;

$\frac{Q_{\hat{I}}^{\hat{I}}}{\hat{A}+\hat{O}}$ - share of drainage flow, formed from infiltration waters from surface (for example $\underline{П}$, $\underline{O}$, $\underline{P}$ *equal to 0* for condition of systematic vertical drainage);

$B+Ф$ - intake to the territory and infiltration from canals;

$ET_П$ - evapotranspiration from irrigated field in a year;

O_P - water supply to irrigated field;

O_C - atmospheric precipitation;

/ I-в / - type of soil profile according to classification in chapter 4;

0.6-1.2 - depth of ground waters during vegetation;

(0.9-1.5) – cotton field, bottom- same for alfalfa.

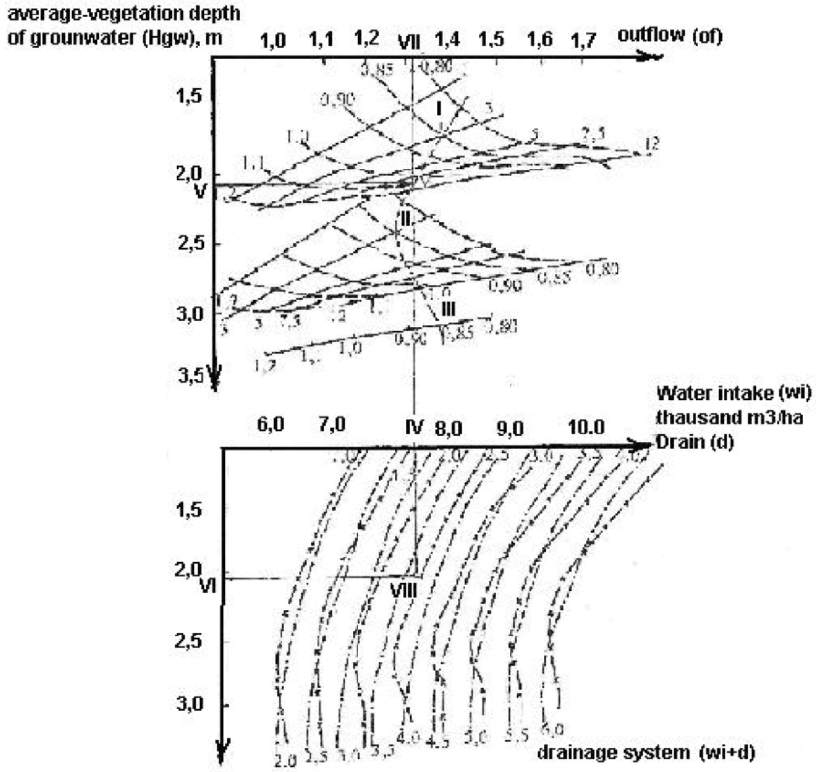

Figure 1. Diagram of parameters of "critical" reclamation regimes during vegetation period for soils of light texture in Khorezm region, given that subsurface inflow equals to zero (present condition). Mineralization of irrigation water – 1.2 g/l (1.2; 1.1; 1.0; 0.9; 0.85; 0.80 – relationship line r/α_{op}; 3; 5; 7,5; 12 – mineralization of ground waters $M_{гр}$, g/l).

- Strengthening the management of water resources on basin principle (irrigation-system) instead of administrative – territorial allows to settle management issues in complex (both water and land resources), management of water quality and impact of ground waters to ecological system

This requires improvement of legislative base on organization and functioning of basin management:

- Repair-rehabilitation works on drainage systems. Cleaning of interfarm and onfarm open collector drainage systems. Cleaning of subsurface horizontal systems with drain flushing machines. Cleaning of vertical drains with pneumatic impulse method, and supply floating pumps and other required technical means

- Repair-rehabilitation works on irrigation canals of different category. There are required sufficient amount of materials, technique and finances for timely fulfill of measures
- Thorough assessment of quality and resources of CDW with purpose of their environmentally safe use on irrigation and leaching
- Rational and even redistribution of irrigation from sources ground waters and CDW, between water users
- Optimization of structure of irrigation fields

Measures for mid and long term perspective. (2011-2025 гг):

- Optimal location of agricultural crops accounting natural and farming conditions of the region
- Taking highly and moderately saline lands out of production
- Capital leveling of lands
- Rehabilitation of drainage. Orientation to deep subsurface horizontal drain, combined and vertical drainage
- Local manufacturing of irrigation-reclamation facilities for construction of and maintenance water management sites
- Reconstruction of irrigation systems orientated to systems with maximum efficiency (none-pressure and pressure pipeline systems with use of local plastic films and plastic pipes, flumes and concrete canals)
- Development of irrigation technique and technology has to account at least partial water charging, under condition of severe deficiency of water resources and deterioration of environment. There have to be involved managed progressive means of irrigation allowing direct delivery of water to the root zone

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IMPROVED GROUNDWATER MANAGEMENT STRATEGIES AT THE AMU DARYA RIVER

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Abstract: The downstream part of the Amu Darya is still characterized by a number of world's most pressing environmental problems, such as water scarcity, toxic pollution, soil degradation, and serious impacts to human health. More than 3 million inhabitants live at this area without sufficient water supply. Without having access to centralized water supply of sufficient quality, uncontrolled use of polluted surface water and ground water remains as the only local option for the people. Fresh ground water resources are scarce and appear frequently as shallow groundwater lenses. These are of increasingly poor quality due to over-exploitation and at risk of being too polluted for further use. This paper revises our assessment of the current status of groundwater lenses and gives outline recommendations for an effective combination of pollution reduction and recharge. The work has been undertaken within the EU INTAS funded project 808 "Investigation of innovative pollution clean-up and avoidance strategies for surface water and groundwater resources at the 'Disaster Zone' of the Amu-Darya lowers" (OPAL).

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Keywords: Amu Darya river; wastewater; ground water lenses; quality; pollution; salinity; remediation

1. Introduction

In the Aral Sea basin, safe water resources are scarce and becoming steadily scarcer. In particular, high quality drinking water is and will continue to be a rare asset, in view of high population growth, increasing agricultural abstraction and rising requirements to water quality.

A particular focus is provided in this paper, to give an overview on the dimension and status of freshwater resources in alluvial parts of the lower Amu Darya region, the so-called “groundwater lenses” (GW-lenses).

Because of the high salinity of surface waters and shallow ground water resources, there is a need to explore alternative sources and storage of high quality waters. Storing low saline water resources from the Amu Darya summer flood by recharging local GW-lenses provides a promising way to protect drinking water against evaporation.

The paper begins by describing the status of the GW lenses for water management and to what extent this influences the water supply. It also revises understanding of the current situation for 10 study sites with respect to water availability, pollution, and possible remediation options of the GW lenses.

Using surface water resources for recharging the lenses under local conditions, however, requires a further removal of pollutants such as organic matter (indicated by COD/BOD), heavy metals and pesticides. An additional problem is the reduction of infiltration rates by the reduction of permeability due to the enrichment of suspended solids. For this reason, the paper introduces the capabilities of specific infiltration facilities (*Bio - Engineering System*) on the basis of constructed wetlands. Here, the effect of the improved infiltration rate and simultaneous degradation of pollutants becomes apparent. For this, field and laboratory experiments have been undertaken.

2. Status of Water Supply and Groundwater Resources in the Lower Amu Darya Region

The Amu-Darya river is one of the two main inflows to the Aral Sea. Its lowlands consist of three administrative areas: Khorezm, Karakalpakstan (Uzbekistan) and Tashauz (Turkmenistan). These cover the region between the Tuyamuyun dams (300 km south of the Aral Sea) and the former Aral Sea shoreline. Irrigation is the dominant economic sector in these regions and covers a total area of 1076000 ha (Ministry of Health, 2002; FAN, 1975). The extension of irrigation areas has led not only water use which exceeds the water availability, but also to severe land degradation and a notable reduction of water quality.

Particularly in the lower part of the Amu Darya, the situation must be be considered critical. Today, about 3 million people do not have access to safe drinking water resources (Aladin, 1992) and only 54.3% of the people in Karakalpakstan and 43.5% of the people in Khorezm had been connected to central water supply by 2000 (Ministry of Macroeconomics and Statistics, 2000; Zakhidova, 2004).

Uncontrolled use of these water sources which typically are of poor quality represents an important public health concern. After 40 years of steady decline, the incidence of tuberculosis has almost doubled since 1991 reaching 72.4 per 100,000 in 2001. This increase is reported in all regions, but is particulaly serious in rural areas, such as Karakalpakstan, where the incidence of 149.9 per 100,000 is more than twice as high as national average (Aladin, 1992). Over the last few years the region has experienced several outbreaks of infectious diseases which include diphtheria, viral hepatitis and typhoid (Ministry of Health, 2002).

Consequently, water and health problems, especially in rural areas of Khorezm and Karakalpakstan, are closely related to the need to maintain a decentralized water supply from local ground water and surface water resources.

The GW resources, like aquifer and lenses, depend on recharge from the Amu Darya river and the associated canals system. During recent decades, the quality of the surface and ground waters has suffered from anthropogenic deterioration due to such factors as agricultural and industrial return flows with contaminated and high saline waters.

Based on data from the State Committee for Geology and Natural Resources, Table 1 gives an overview of the characteristics of selected ground water lenses which are connected to the main canals. Most of the GW-lenses are located near the surface, with water tables at a deth of less than 5 m. This underlines the need to seriously consider protection against recontamination from the surface. A number of lenses are reported to be no longer usable (in Table 1 marked as x) because of high salinity.

Comparing the available information, the potential usable volume of groundwater lenses clearly exceeds the actual groundwater use. Even if there is still significant unaccounted water abstraction from local wells and canals in the rural areas, the groundwater lenses have the potential to contribute significantly to securing future water supply together with improved water supply from the Tuyamuyun Hydro complex.

Table 1. Regeneration of groundwater resources by source (O–operating X–out of operation)

GW lenses	Feeding source	Lens extension: width /length (km)	Water table/ thickness (m)	Usable volume, thousand m^3/day
Khojeyly (X)	Amudarya river left bank	0.8 - 3.7	2 - 7/25	42.3
Karabayliy (O)	Leninyab canal	0.4 - 3.1	1.3 - 10/4 - 15	4.32 - 13.8
Nukus(X)	Kizketken canal	1 - 5	5 - 15/3 - 30	8.4
Akmangit - (O)	Octyabr arna canal	0.3 - 5.5	2 - 15/5 - 35	3.2 - 6.25
Erkindarya (O)	Abadyarmish canal	0.3 - 2.5	1.5 - 3.5/28 - 30	2.5
Tokumbet Khalkabad, Avez,Imanch, (X)	Kegeyly canal	0.2 - 4	3 - 5/10 - 24	3.26 - 12.6
Kenes (O)	Beshyab canal	1.2 - 2.3	1.2 - 2/8.32	3.1
Shagalkupir, Alikul (O)	Kuvanishdjarma canal	0.3 - 1.6	1.5 - 3.5/10 - 26	1.3 - 3.9
Esim (O)	Esim canal	0.3 - 2.1	2 - 2.3/20	2.93
Beruny, Bybazar (X)	Amudarya river right bank	0.15 - 2.2	1 - 5.2/29 - 75	3.9 - 9.3
Akchaul (O)	Kirkkiz canal	1.1 - 2	1.5 - 3/ 20 - 95	4.3 - 26
Jambaskala (O)	Kelteminar canal	0.2 - 1	1.8 - 5.3/18	1.3
Akhunbabaev, Ellikala (O)	Amirabad canal	1.5 - 1.3	1.8 - 3.5/ 33	4 - 15
Klichbay (O)	Klichniyazbay canal	0.6 - 1.3	1.5 - 2.7/33	12.9

3. Assessment of Water Quality Status for Selected Lenses

To assess the current water quality status of 10 selected groundwater lenses, data from several monitoring stations (location indicated in Figure 1) were obtained from the Uzbek State Hydro-Geological Institute.

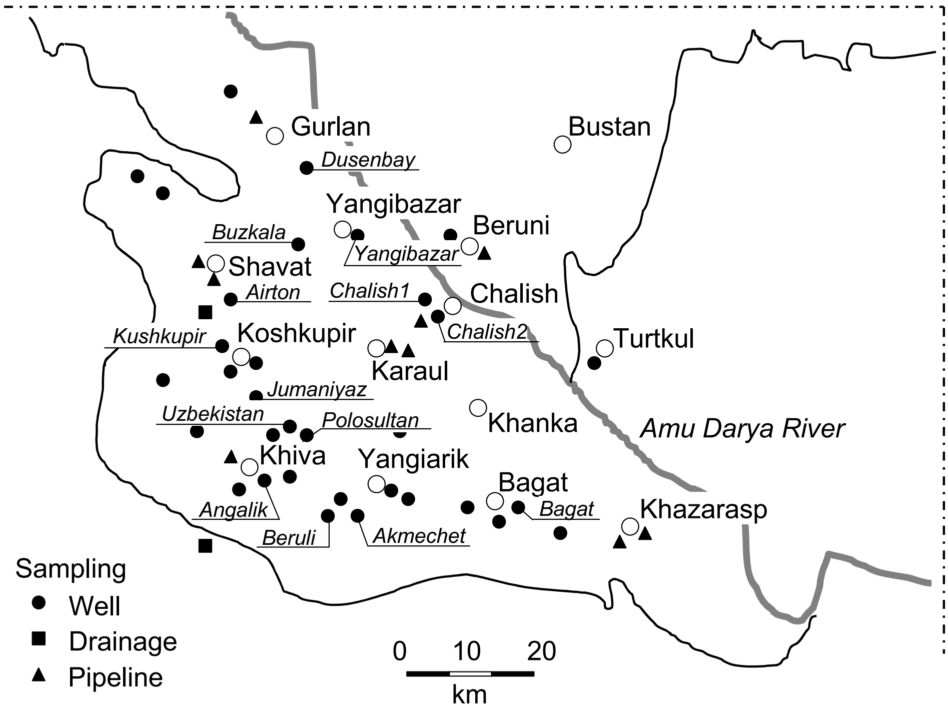

Figure 1. Site map of lower Amu Darya region and location of monitoring stations for the investigated GW lenses.

The data provided comprise GW levels, water balances, recharge rates and hydrochemical analyses for the following ground water lenses: Chalish (13), Bagata (8), Yangiarik (12), Polosultan (15), Angalik (18), Akmechet (19), Shavat (Uzbekistan kolkhoz) (21), Jumaniyaz (23), Airton (25), Buzkala (28). Samples where obtained by pumping from a depth of 5 to 15 metres in November 2002, September 2003, and May 2004.

For each samples, information has been provided on salinity, hardness, heavy metal concentration, cysts of parasites such as *helminthes* or *Giardia lamblia*, and pesticides (HCH, DDT, DDE and DDD) and oil products.

The results reveal high salinity and hardness as well as raised concentrations of heavy metals and parasites. Figure 2 shows the average concentration for selected constituents. The salinity in at most of the stations is in the range 1000 and 1600 mg/l and exceeds the Uzbek MAC of 1000 mg/l. The highest salinity is indicated at Chalish 1 with 2106 mg/l, at Dusenbay with 2525 mg/l, and Yangibazar with 2968 mg/l.

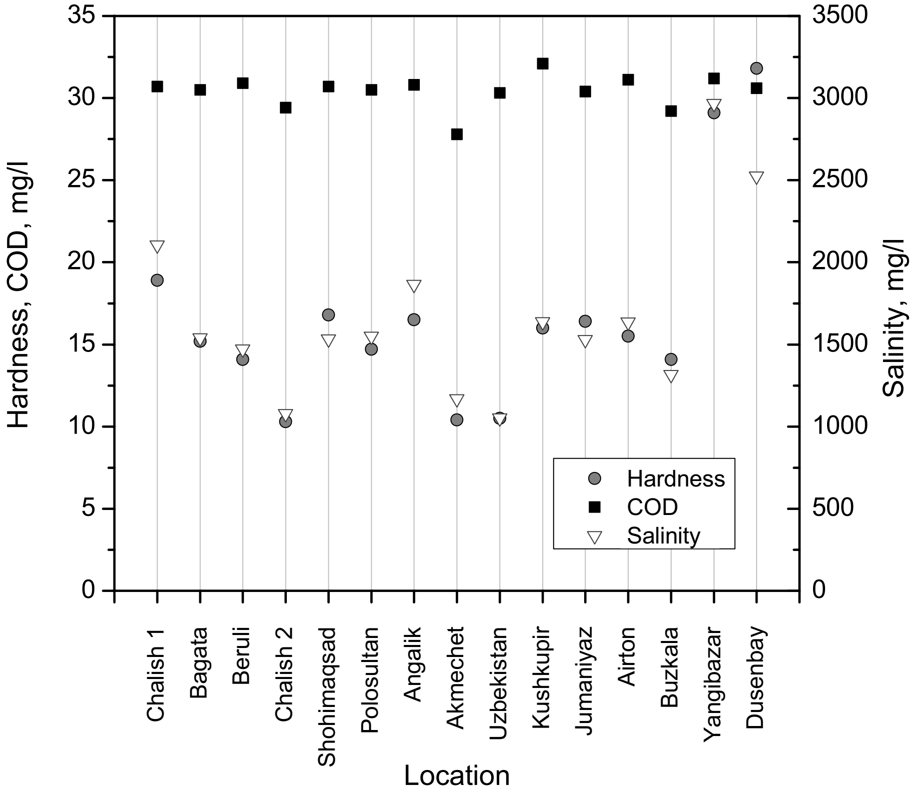

Figure 2. Water quality status of investigated lenses, average concentrations for hardness, Chemical Oxygen Demand (COD), and salinity.

The MAC for hardness of 7 mg/l is exceeded at all stations. The highest values are found at Chalish 1 with 18.9 mg/l, at Yangibazar (29.1 mg/l), and Dusenbay (31.8 mg/l.) At all monitoring stations, the measured COD concentrations were around 30 mg/l, significantly exceeding the MAC of 15 mg/l.

Data on current heavy metal contamination are shown in Figure 3. The highest Zn concentrations are shown for Beruli and Kushkupir. Cr is highest at the lenses Chalish 2 and Kushkupir, with both of these lenses also having high Cu and Fe concentrations. The highest Pb concentrations are found at Angalik, Kushkupir, and Jumaniyzaz.

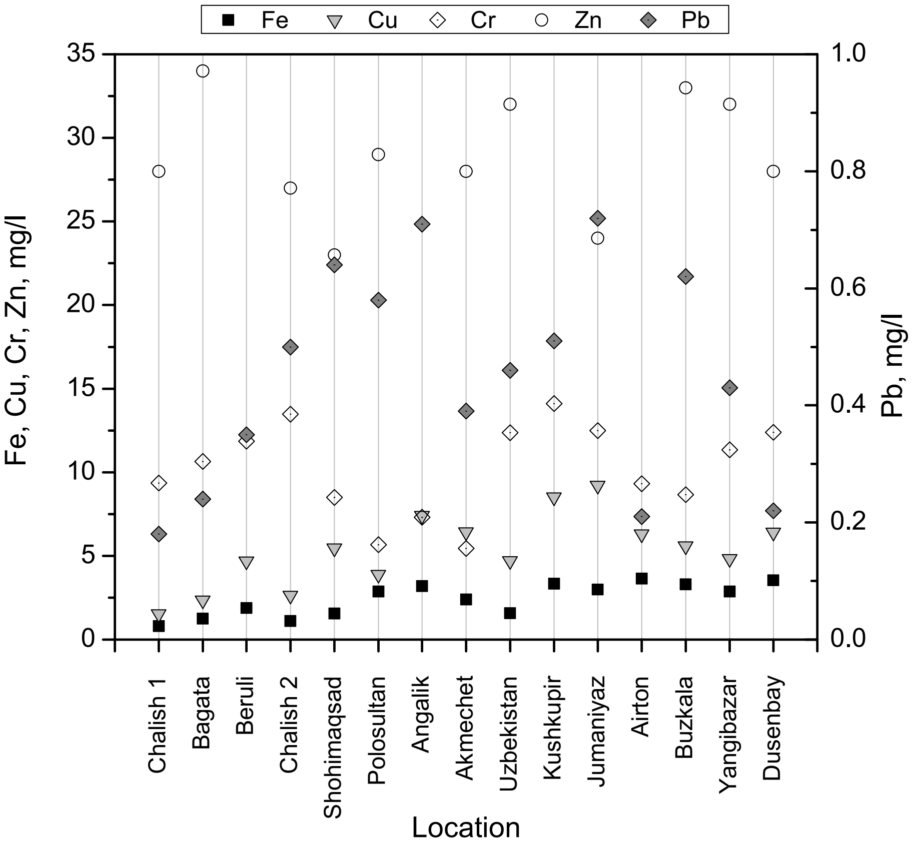

Figure 3. Heavy metal concentrations of investigated lenses, average concentrations for Iron (Fe), Copper (Cu), Chromium (Cr), Zinc (Zn), and Lead (Pb) (mg/l).

4. Improved Methods for Recharging Fresh Water Lenses

In order to feed the GW lenses, traditionally simple infiltration canals are used. The system consists of, parallel infiltration of canals and in-between lying wells. Past experience in the operation of these recharge systems has shown that high concentrations of suspended particles lead to clogging and a rapid reduction of permeability near the canals. An example of reduced permeability is given in Goncharenco et al. (2004). Following his results, it is to be expected that within a period of 20 to 30 days after the initial start of operation, the relative penetration rate is reduced by two to four times.

Alternative water resources to those from irrigation canals and collectors are rare in the lower part of the Amu Darya. Only a few municipalities have rudimental sewage system, which are generally in a poor operating condition.

Particularly in rural areas, latrines and other forms of dry deposition provide most of the sanitation.

Ideas for improving the recharge have to consider the availability of surface waters originated from the Amu Darya and irrigation canals. Both (i) the improvement of hydraulic efficiency by reduction of suspended matters and (ii) the reduction of pollutant concentrations are, therefore of critical importance.

The system tested by CEWM is termed a *Bio - Engineering System* (BES) referring to the combination of a constructed wetland for filtration and pollutant remobilisation together with an infiltration basin (Rhamonov et al., 1998, Razakhov et al., 2004).

The principle design of the BES is shown in Figure 4. It is based on shallow artificial ponds and consists of 3 layers: (1) the open water surface, facilitating sedimentation of suspended particles and biochemical reactions in the water phase, (2) the root zone layer (sand), enabling filtration, pollutant uptake by macrophytes and biochemical reactions at the root zone and (3) the anaerobic layer. Below the root zone, a gravel bed is constructed, acting as a drainage and infiltration layer for the recharge of the lenses.

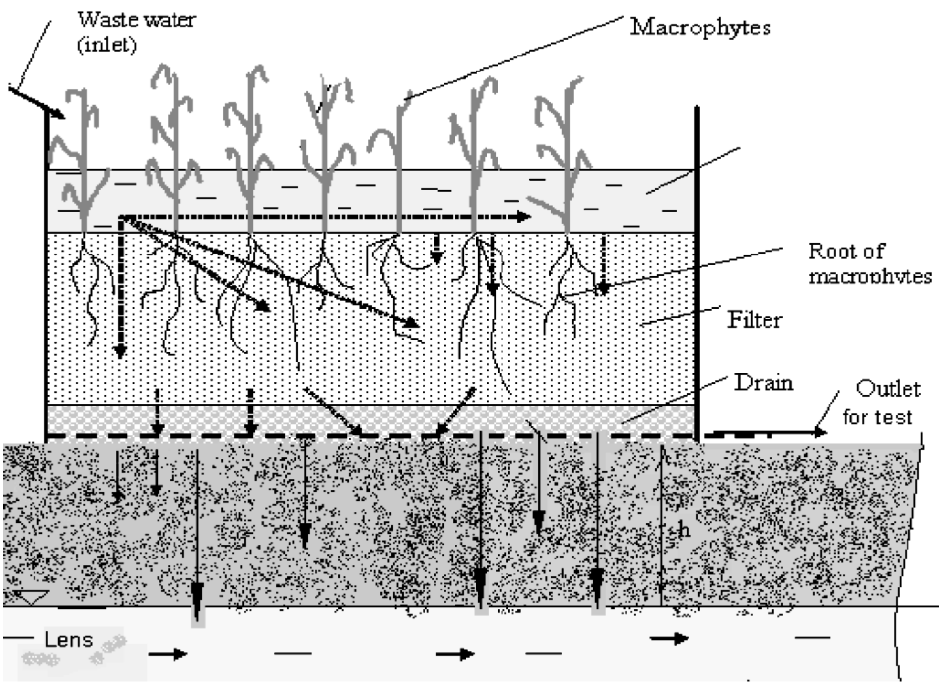

Figure 4. Main components of the Bio - Engineering infiltration System.

The construction of the BES allow a balanced interaction of biota involved in the clean-up process, like macrophytes, phytoplankton, periphyton, zooplankton, zoobenthos, bacteria, and enhance the effective combination of aerobic (in the root zone) and anaerobic (in the lower zone) processes.

Past experiments underlined the positive effect of increasing the permeability by a sufficiently developed root system. Figure 5 show results from testing the variation of permeability of different macrophytes over time, and depicts typical variations of permeability during the first 1.5-2 months for such systems.

During an initial phase, the situation is the same as without macrophytes (Figure 5, no vegetation). After 45 - 60 days the root systems are developed and have grown up to 0.5 - 0.8 m. From then on, infiltration to GW rises due to the loosening of the filtration bed and the partially achievement of the initial permeability. The perforating ability of roots differs amongst the species investigated. *Acorus p.* developed the most effective permeability, while *Typha lat*. and *Phragmatis comm*. revealed similar and slightly lower effects.

Rhizofiltration, a special case of phytoremediation, is considered essential as a mechanism for stimulating pollutant uptake by plants. The removal process is enhanced by phyto-degradation, in which the plants, along with enzymes, are able to degrade organic pollutants. Nitrogen reduction is facilitated by the assimilation of nitrates. The wetlands also creates favorable conditions for bacterial nitrification and denitrification processes. Reduction of heavy metals is due to metal absorption by macrophytes and phytoplankton. The residual matter is destroyed by microorganisms and *fungi.* Constructed wetlands also encourage a reduction of bacterial pollution. Experiments with reeds have shown a decrease of *colon bacillus* from 220 - 46000 cells/ml to 2 - 6 cells/ml.

Experiments with the BES have been conducted to demonstrate the effect under specific local conditions. Results of the experiment are presented in Table 2. Concentrations of BOD decrease by about 80 to 87%. In comparison to the lesser reduction of COD (68 to 86%), it is shown that the treated surface water contains more degradable than persistent constituents. Suspended solids were nearly all removed (98%) by sedimentation and adsorption within the through-flow to the filter layer. The intense reduction of NH_4, NO_2, and NO_3 indicate both effective nitrification and denitrification effects.

High uptake rates of heavy metals provided by hydrophytes and phyto-plankton result in a significant reduction in ionic concentrations. By interactive processes between sulphite and sulphide groups, the macrophytes facilitate an additional chemical adsorption of the metals. Sedimentation linked to sorption processes in the filter layer also contributes to the observed decline of the heavy metals. The results indicate a total reduction within a range of 95 to 100%. An exception is indicated for copper where only 86% reduction is reached.

Table 2. Pollutant contents reduction with the BES in one day

Description	BOD mg/l	COD mg/l	NH_4 mg/l	NO_2 mg/l	NO_3 mg/l	PO_4 mg/l
Input	24 - 65	53 - 180	10 - 20	0.2 - 5	14 - 48	0.9 - 3.2
Output	4.7 - 8.4	17 - 26	0.35	0 - 0.05	1 - 4.2	0 - 0.1
Reduction, %	80 - 87	68 - 86	99	92	98	98
MAC	2.5	15	0.1	3	45	3.5
Description	Sus. Solids mg/l	Fe mg/	Cu mg/l	Zn mg/l	Pb mg/l	Cr mg/l
Input	112 - 200	0.78	0.07	0.8	0.3	3.8
Output	3 - 4	0.05	0.01	0.01	0.005	0.0
Reduction, %	98	94	86	99	98	100
MAC	1.5 - 2	0.3	1.0	1.0	0.03	0.05

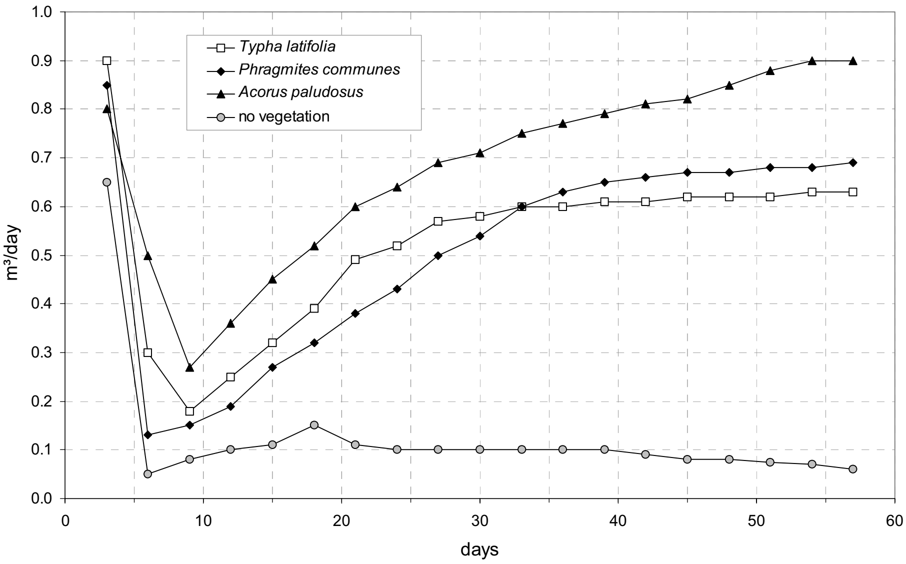

Figure 5. Temporal development of soil infiltration rates for different macrophytes.

5. Recommendations for Future Applications

At present, a constructed wetland pilot plant was planned by CEWM for demonstrating the practical applicability in drinking water purification. It is now implemented at Kazakhaul, which is a village in the Khojeyly district with 5,000 inhabitants (Figure 6).

In the past, the local population used surface water downstream of the Takhiatash hydrocomplex as drinking water. According to data from monitoring programs, the water quality at this location does not meet the Uzbek standards for drinking water supply and required alternative approaches.

The capacity of this BES is designed to provide 100 l/(cap ·d) and was built as a combination of a lagoon and a constructed wetland. Figure 6 outlines the basic concept of the treatment plant. Within the lagoon, a first sedimentation of particulate substances and uptake of nutrients by *Chlorelya sp.*occurs. The pre-treated water is then discharged to a vertical-flow constructed wetland.

According to the monitoring results, the following pollutant reductions might be achieved: TSS: 99-99.7%, COD: 29-71%, BOD: 54-83%, phosphates: 33-72%, NH_4 and NO3: 80-99%, NO_2: up to 100%, Pb: 43-83%, Cd: 53%, Fe-42: 52%, Mn: 39-96%, Zn: 31-75%, Cu: 33-81%, mineral oil and pesticides: up to 100%, bacterial pollution: 71-98%.

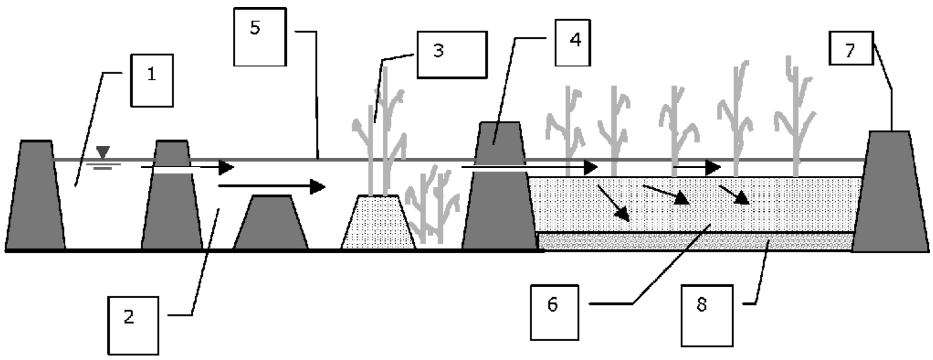

Figure 6. Cross-section of the pilot treatment plant at Kazakhaul: 1- water inflow 2- settling trap; 3- hydrophytes; 4- outlet; 5- lagoon; 6- constructed wetland; 7- dam; 8- filtering base.

6. Conclusions

The combined use of constructed wetlands and infiltration plants has a number of important advantages. These include (i) improved mitigation of clogging effects due to pre-filtration and development of root systems (ii) reduction of pollutants such as heavy metals, organic pollutants, decomposable organic matter, and cysts of parasites, which must be considered as important under local conditions.

While the purification capacity of BES has been verified in several laboratory and field experiments, the impact of extreme winters on the treatment capacities and biological functioning needs further investigation. Also the direct combination of treatment plants and recharge facilities must be tested in a pilot plant. However, the information already available strongly suggests that the combination of treatment of surface waters and the recharge of local groundwater lenses would overcome some of the major historical constraints.

The treatment facilities as proposed would allow maintenance of continuous pollutant monitoring in the recharge water used for recharge with storage in the lagoons, in case of non-functioning and problems.

Salinity reduction is best achieved through the use of lower saline summer floods for recharge. The storage of such water in ground water lenses, would provide the required supply of lower salinity water. At the same time, the wetland systems would be needed especially during the summer months, avoiding any of the anticipated complications due to very low temperatures during winter.

The recharge of lenses could focus on small individual GW bodies and hence minimize the potential recontamination of recharge water with inflowing ground water. Above all, the recharge of lenses with treated surface water would effectively reduce the evaporative water losses, as long as the water table was kept below a critical elevation.

It is recommended that the activities on combined surface water treatment and recharge of ground water lenses be extended to include practical demonstration plants.

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THE IMPACT OF GROUNDWATER PRODUCTION AND EXPLOITATION ON ECOSYSTEM IN AZERBAIJAN

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Abstract: The Azerbaijan Republic is located on the western side of the Caspian Sea, in the arid climatic zone and feels deficit of general water balance. Due to this, the rational exploitation of fresh and weakly saltish groundwater has great importance. The production and utilization of the groundwater caused negative consequences that affected on natural conditions of different regions. For example, the decrease of the groundwater level in a part of Qusar region has created some ecological problems. In the regions of development of the relict forests the dynamic levels of the groundwater decreased on 25-30 meters. This affected root-inhabitable stratum and as a result led to abrupt aggravation of the ecological situation of the regions. It was required to optimize the production of the groundwater. Another example is Qanikh-Ayrichay field. Here we can notice abnormally high hydraulic interrelation between ground and surface waters, and underground component of the river drain in the natural conditions forms approximately 55-65%. When infringing given conditions by production the groundwater the decrease of their average level forms approximately 50-60 meters. The appeared depression funnels cause tightening of the additional volume of river run-off to the groundwater intake wells, that is damage river run-off. Given volume of the river run-off must not be calculated twice when assessing the common water balance of the region. Opposite situation is observed in the Absheron peninsula where there are almost no fresh water resources. The water demands are met by surface water resources of Samur river and groundwater transfer from Guba-Gusar region of the Republic (more

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than 200 km from the demand areas). These sources are augmented with water from Kura river in the Ali-Bayramli region (more than 180 km from the demand area), at rates of up to 30 m^3/sec. Due to the specific hydrogeological conditions, absence of sewerage system and etc., it leads to land subsidence, landslide processes, flooding, and other environmental and social phenomena and abrupt aggravation of the geoecological situation of the peninsula, in the whole. As it has already been mentioned a big part of groundwater in Azerbaijan is utilized for irrigation of agricultural crop. Mostly we notice fresh (up to 1.0 g/l) or weakly saltish (up to 3.0 g/l) hydrocarbonate-sulphate and sulphate-hydrocarbonate waters with different cationic composition. The combination of cations form irrigation factors of the groundwater and as a result forms opportunity to utilize it for irrigation of concrete soils. Incorrect calculation of given qualitative parameters in the "water-rock" system has led to salinization of the soils of irrigated areas of Mil and Mugan-Salyan plains and accordingly led them to exit from the crop rotation. At present time, melioration of these areas is realizing. Thus, we can certify that production and utilization of the groundwater is essential anthropogenic factor, which influence on geoecological situation of the republic. This factor must be taken into account when forecasting water-related activities in the whole and concerning exploitation of the groundwater, in particular.

Keywords: water-bearing horizons, aquifer, water-saturated series, water-economy balance, gravel-pebble sedimentation, depression funnel, exploitative reserves

1. Introduction

The Azerbaijan Republic is located on the western side of the Caspian Sea, in the arid climatic zone and feels deficit of general water balance. Due to this, the rational exploitation of fresh and weakly saltish groundwater has great importance. Groundwater fields are the Quaternary water-bearing series of foothill and intermountain plains, which are unevenly distributed through the territory of the republic, and their exploitation reserves are not equal as well. In all, 11 fields with general exploitation reserves 14.2 mln. m^3/day of fresh and weakly saltish groundwater are utilized and approximately 49% of them were utilized during the most intensive exploitation period (1980-1988). However, in some fields (Karabagh, Mil, Jabrail) the volume of annual production has reached 85-95% of approved reserves. The most part of the produced groundwater (up to 90%) is used for irrigation of agricultural crops and only

10% is used for centralized and decentralized supply of the settlements with fresh water and for industrial-technical needs (Figure 1).

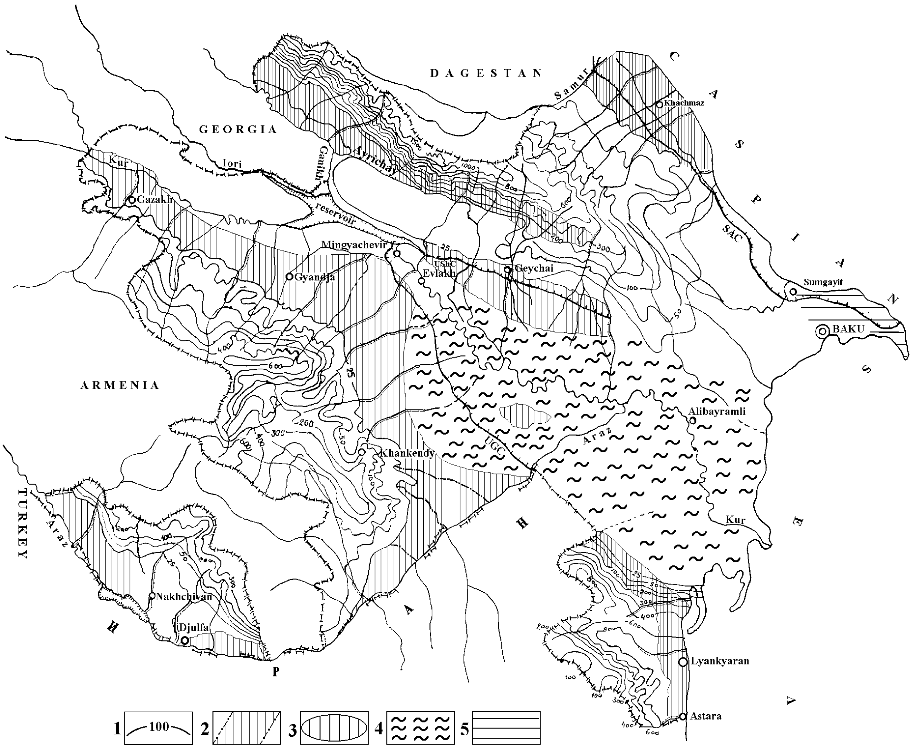

Figure 1. The scheme - map of the average annual river run-off of the rivers of Azerbaijan Republic with drawing the fresh groundwater fields. 1 - Average annual river flow, mm. 2 - Design countries between two rivers, which useful fresh groundwater resources were estimated by. 3 - Territories of fresh groundwater fields. 4 - Territories of mineralized groundwater development. 5 - Territories of sporadic fresh groundwater development.

2. The Current Situation in Azerbaijan

The production and utilization of the groundwater caused negative consequences that affected on natural conditions of different regions. These consequences are classified according to two signs:

- processes related with increase and decrease of the groundwater level;
- processes related with interrelation of quality parameters of the "water-rock" system.

The water-saturated series of foothill plains are composed of aquifer and some (usually 3-4) confined waters, hydraulically interrelated with each other. In addition, piezometric surface of the underlying horizons exceed heads of the overlying ones (owing to uniform area of feeding) and as a result, the unloading occurs from bottom to top. Reservoirs are gravel-pebble deposits with a lot of sandy inclusions, as well as different-grain sands. As a rule, coefficients of their filtration vary within 5-15 m/day, sometimes up to 25-30 m/day. During the production of groundwater by the water-wells all the aquifer and confined water are actually utilized (owing to their hydraulic interrelation). Therefore, their weighted-average levels are accepted in calculations (Figure 2).

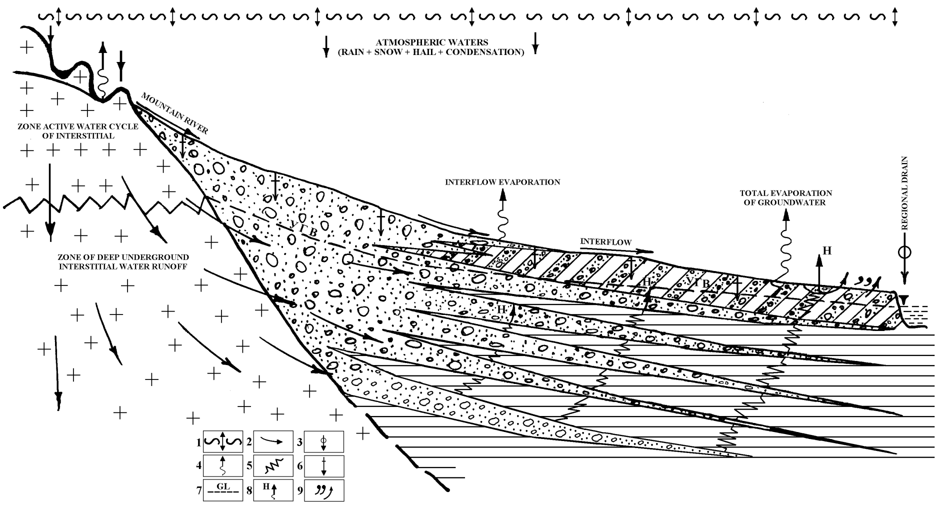

Figure 2. The basic scheme of formation and expenditure of stratum - pore groundwater of foothill plains. 1 - Upper boundary of the system. 2 - Groundwater runoff and interflow streamline. 3 – Regional drains (Kur, Araz, Ganikh, Ayrichay rivers, sea). 4 - Interflow and groundwater run-off water table evaporation. 5 - Lithologic holes. 6 - Atmospheric and surface water infiltration. 7 - Groundwater level. 8 - Pressure gradient. 9 - Groundwater run-off zone.

Regional depression funnels, generated under influence of the groundwater production, in its turn disturb the natural hydrodynamic balance of the water-saturated series and provokes new conditions of their interaction with the hydrosphere waters. If these changes have been foreseen when assessing the regional exploitative reserves, no negative consequences should be in the geo-ecological environment. However, in practice we often observe the opposite picture.

For example, Qusar foothill plain is rich with water, the exploitative reserves of which are approved by analytical calculations on the base of allowable decreases of the dynamic level, that is about 68.8 meters, and radius of the depression funnel of the calculated water-well is 15 km. At the same time within the mentioned depression funnel on the area of 320-hectare there are lowland woods with the relict trees, protected by the state. Calculations have shown that in 5 years of the groundwater exploitation within the large forests the weighted-average levels of waters will decrease on 11.6 meters (Figure 3). The laboratory on forestry of the Republic Institute of Botany made special researches concerning impact of the groundwater level decrease upon the woods. The tendency of drying of the trees of all classes has been defined and in case of the further decrease of the groundwater, the death of large forests is practically inevitable. Taking into account the mentioned above, the construction of the water-well in the present region has been suspended. Exploitative reserves were revalued (towards the reduction) with account of the environmental protection requirements.

Negative impact of the regional depression funnel is revealed within Qanikh-Ayrichay field. Here the river network is well developed, and the underground component of their weighted-average annual run-off is high and varies within 45-62%. Exploitative reserves of the field were estimated by analytical calculations of the forecast linear water-well, located perpendicularly to the groundwater run-off along its front. Allowable decrease of the water-saturated series with thickness to 500 meters was about 90-100 meters and has been limited by the technical opportunities of barrage pumps. Thus, the hydraulic interrelation of the river and groundwater run-off has not been taken into account in calculations. We have made additional researches on assessment of damage to the surface run-off from the sharp decrease of the groundwater level. The calculations were made according to the schedules and dependences, developed by E.A. Minkin and S.J. Kontsebovskiy (1979). It has been revealed that the calculated depression funnel will cause the additional inflow of the river run-off and on some rivers (Aliganchay, Turanchay, Ayrichay, Damarchik, etc.) the damage to their run-off will be about 65-78%. Thus, the depression funnel provokes the additional tightening of river waters to the groundwater water-wells the volumes of which should be accounted during the assessment of general water balance of the present natural-economical region of the Republic. In other words, the same water should not be counted twice.

In the areas of intensive production of groundwater, the generated depress-sion funnels are indices of reliability of the assessment of exploitative parameters of the fields. So, for the Ergi water-wells, within Mil foothill plain, exploitative reserves in volume of 0.75 m^3/sec with allowable decrease of water-wells (148 wells) within 25-30 meters were estimated. When utilizing the water-wells their dynamic levels began to exceed the calculated allowable decrease that has led to the breakdown of the pumps, abrupt reduction of the productivity and the general water-take-off has decreased up to 0.3 m^3/c. The present indices show that when designing water-wells Ergi the essential mistakes have been admitted. Similar cases can also be noticed on some water-wells of Ganja, Karabakh, Mil, Jabrail plains, where the allowable decrease, revealed during the investigation, exceeded during their exploitation (Table 1).

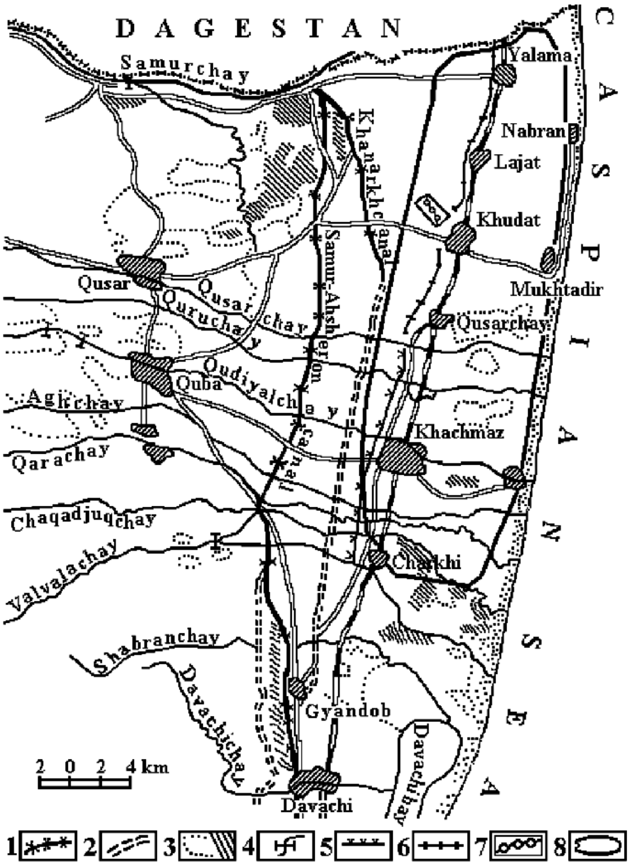

Figure 3. Qusar fresh groundwater field with allocation of forest massive. 1 - Channels for reconstruction. 2 - New founded channels. 3 - Areas of irrigation. 4 - Water-wells of the rivers. 5 - Khachmaz water-well. 6 - Designed water-well of III Baku water pipe. 7 - Shollar water-well. 8 - Forest massive

Opposite situation is observed in the Absheron peninsula, one of the most developed agroindustrial complexes and where Baku the capital and Sumgayit the third city on the population are located. Own water resources of the peninsula are rather insignificant, in this relation water supply of the region is carried out by drawing of the groundwater and surface water from other areas of the Republic. In a whole, from all sources of water supply about 30.0 m^3/day of water resources are involved to the peninsula. At the same time, hydro-geological conditions here are characterized by the development of the detached trough-like structures, composed of loamy-clay sands marine sediments, hydrorelief inclinations are very weak and are directed to the sea (Figure 4). Water-bearing series with thickness from 2-3 to 20 meters, are composed from unconfined and poorly confined interstratal aquifer with water-conductivity coefficients about 10-15 to 250 m^2/day. Intensive water-economic activities, absence of the regional collecting-drainage and sewer networks as well as specific hydrogeological conditions promoted accumulation of the groundwater in individual hydro-geological structures. Since 1956 (after commissioning the Samur-Absheron channel), the steady tendency of the unconfined water level rise is noticed in peninsula (Figure 5). For individual territories the basic regime-forming factors are infiltration from the irrigated areas and irrigational constructions, as well as leakages from underground communications (in the urbanized territories). As a result of significant anthropogenous loading on the groundwater the levels of unconfined waters had got over the critical mark of 3.0 meters from day surface that had led to underflooding, flooding and swamping of the vast areas (about 84,000 hectare) (Figure 6). The vast areas of settlements Sabunchi, Zabrat, Bina, Buzovna, Mashtaga, etc are flooded or are in state of the latent flooding. Engineering - geological conditions of the flooded territories have abruptly worsened. Changes of the physical properties of the ground have caused development of the slump phenomena and, as a result, destruction of some engineering constructions: bridges, railway and highways, residential buildings and other. Under threat is the main airport of the republic - named after H. Aliyev. Not acceptance of the effective measures on stabilization and regulation of the confined water level within Absheron natural - economic zone has led to the abrupt aggravation of the geoecological conditions entailed the negative consequences mentioned above, put the great loss to the economy of agroindustrial complex.

Thus, we can state that the depression funnels, formed during intensive production of the groundwater, allow assessing the correctness of the prospecting and designing activities during the estimation of the exploitative parameters of the concrete fields or the groundwater water-wells. At the same time, the revealed negative consequences, which are reflected on the quantitative and qualitative parameters of the groundwater and geoecological conditions, in a whole, point to the mistakes admitted when forecasting the operation of the water-wells. In addition, in our Republic more often nature protection factors are not taken into account: damages to an interflow and forest massifs, flooding and swamping of the territories, aggravation of the meliorative condition of the land, etc.

Table 1. Dynamics of the level regime on the water-intaking sites

Water-wells	Allowable decrease of the underground water level according to research data, m	Groundwater level in the exploitation wells, m		Odds between forecast and actual levels of the groundwater, m
		Forecast, according to research data	Actual, during diagnostic study in 1987	
1	2	3	4	5
Agdam	47	25	70	22
Barda	50.9	26	60	34
Yevlakh (Malbinasi)	23.1	3	37	34
Ujar	50	6	90	44
Naftalan	60	33	70	37
Fizuli group	38.3	6.5	44	37.5
Ordubad	22	22	22	0
Jabrail	65.7	24	80	56

3. An Offered Method of Water-Economic Balance Calculation

The analysis of natural and water-economic conditions, the degree of the exploitative reserves study, and also the necessity of account of restrictions on the environment protection have determined the necessity to modify the traditional methods of the exploitative reserves and resources of fresh groundwater of the Republic.

It is known, that intensive water-economic activities abruptly reflect on all system of the hydrosphere and on groundwater, in particular. Taking into account the hydraulic interrelation between aquifer and surface run-off, the whole hydrodynamical system of groundwater is subjected to intensive spatial-time anthropogenous impact, and as a result, the whole balance structure of resources of fresh groundwater is transformed.

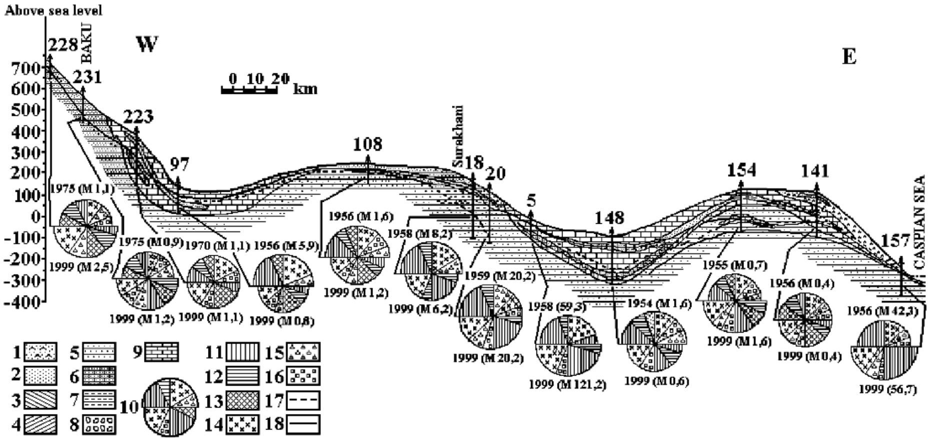

Figure 4. Lines of lithological - hydrogeological sections on line IV-IV. 1 - shells with sand. 2 - consertal sand. 3 - clay sand. 4 - loam. 5 - clay with sand inclusions. 6 - sandstone. 7 - clay with gypsum bands. 8 - gravel–pebbles. 9 - limestone–shells. 10 - comparative characteristics of chemical composition and mineralization of groundwater (half-round sectors describes the cations and anions content, half-round radius describes the increase or decrease of mineralization, numbers point to years, in brackets - annual average mineralization). 11 – Cl^-; 12 - SO_4^-; 13 - HCO_3^-; 14 - Na^+ $+K^+$; 15 - Mg^+ ; 16 - Ca^+ ; 17 - groundwater occurrence for 1955. 18 - groundwater occurrence for 1999.

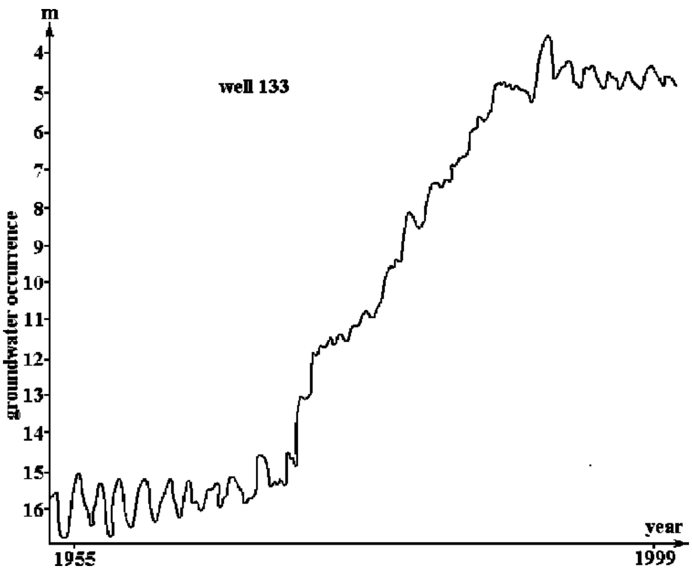

Figure 5. Typical chart of groundwater regime of the well #133.

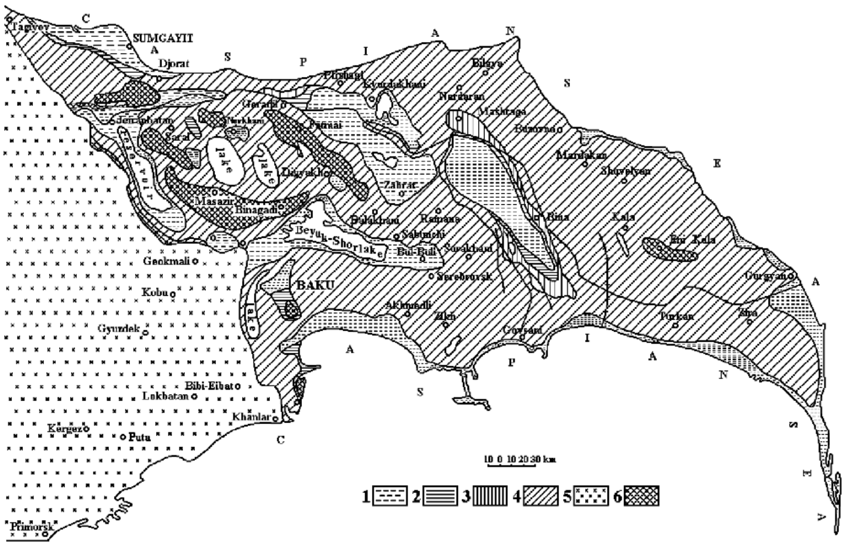

Figure 6. Map of existing and predictable underflooding of Absheron peninsula. 1 – Underflooded zones occurrence for 1982. 2 - Predictable underflooding development zones until 1990. 3 - Predictable underflooding development zones between 1990 and 2000. 4 - Zones, which are not subject to underflooding development. 5 - Region of sporadic distribution of groundwater. 6 - waterproof deposits.

It is necessary to accept the whole hydrosphere in conditions of realization of water-economic actions as the basis of the analytical model. Specificity of the hydrosphere reaction on the impact, which is characterized by a close feedback, requires the priority to the normative forecast of nature conservation activity model. Such estimation should base on the analysis of the process reflecting the basic ability of hydrosphere, namely: interrelation of natural waters the mathematical model of which is the equation of water balance, and in conditions of anthropogenous impact on hydrosphere - water-economic balance (WEB). In conditions of deficit of the general water resources, the important factor of this method is the opportunity to manage the state of the fresh groundwater resources on deficit of WEB, as the present equations serve as a substantiation of a choice of the corresponding actions, directed to the deficit liquidation. In addition, impact between surface and ground vectors of a natural run-off of studied hydrosphere is taken into account as damage to the surface run-off when operating the groundwater and enables to realize water resources management of the hydrosphere as a whole.

We also certify that the rational water-take-off in conditions of water resources deficit should consider only that volume of water, which allow receiving the filtration properties of the deposits, and its take-off will not lead to undesirable infringement of the environment. Therefore, the hydrogeological conditions of the territory, means of the water sampling and value of allowable changes in the environment, which are inevitable when producing the fresh groundwater, determine the value of resources of the groundwater. In addition, the estimated term of the revealed resources has no limits due to possibility of their management.

The most complete agricultural-balance approach during the estimation of the resources and the forecast of the hydro-geological conditions of fresh groundwater [5], where the scheme of consecutive tasks is substantiated according to their hierarchy. Let us define the sequence of tasks solution basing on the present scheme:

- I stage:
  - selection of the balance sites for joint estimation of the WEB of surface and groundwater;
  - selection of the water resources, for which infringement of developed balance of the water is inadmissible or allowable in the restricted limits;
- II stage:
  - set of restrictions;
  - drawing up of the WEB equation and definition of its natural and account parts;
- III stage:
  - estimation of WEB equations components;
  - forecast of consequences of withdrawal of the water resources by mathematical modeling method.

We shall preliminary group the sources of fresh water reserve (FWR) exploitative reserves formation in the following way. We shall place the sources having hydraulic connection with exploited water-saturated series in the group A, which are subdivided in:

A_1 - the groundwater, entering the boundaries of balance sites from the sides;

A_2 - the groundwater, flowing in boundaries of balance sites from mountain area;

A_3 - surface waters of the rivers, lakes, water basins, etc.

The group B includes the sources having infiltration connection with exploited water-saturated series. Here we distinguish:

B_1 - surface waters of the rivers, water basins, channels, etc.
B_2 - atmospheric precipitation (meteoric waters).

Then the base management of the WEB should be written in the following way:

$$Q_{for} - Q_{res} = Q_{\exp} \quad (1)$$

and;

$$Q_{esp} = Q_A + Q_B + Q_C \quad (2)$$

where: Q_{for}-the amount of water actual within the calculation area; Q_{exp} - exploitative reserves of the groundwater; Q_{res} - amount of water, which is necessary to leave from ecological and sanitary point of view; Q_A and Q_B - the parts of exploitative reserves, provided with the engaging of the resources of the groups A and B accordingly; Qc - the charge of the water, provided with the depletion of capacity reserves (without completion) of the groundwater.

Values Q_A, Q_B and Q_C represent discharges of the water-intaking systems, which involve the greatest possible quantity of water from the sources of the given groups. In connection with the precise orientation to the water source, the unified algorithm of the estimation of these sizes is developed and parameters of it are resulted below (Figure 7). This unified algorithm practically represent the multivariant forecast, directed on a choice of the reserve structure, providing selection of the greatest possible or required quantity of the groundwater at allowable infringement of water balance of the territory. At the same time, definition of the given sizes should be accompanied by the estimation of the consequences of withdrawal of the water from the completion sources and the solution of the question on their optimum quantity, which can be involved in formation of exploitative reserves.

With this purpose at the final stage, optimum exploitative parameters of the abstract water-wells are determined by the method of mathematical modeling and the total productivity of them makes exploitative resources of the researched water-bearing series. On the basis of the water-conductivity maps, depths of the burials, hydroisohypse, piezometric contours, chemical composition of the ground and confined aquifers, hydrological parameters of the interflow and other materials optimum sites and circuit (linear or areal) of the design water-wells locations are determined. When modeling the restrict-tions (Q_{res}), which are foreseen in the base WEB equation (1), connected with the protection of quantitative and qualitative parameters of the exploitative reserves of groundwater and environment should be taken into account. These restrictions can be: minimization of damage to the interflow from exploitation of groundwater; restrictions of allowable decrease in connection with the environment protection requirements and many others.

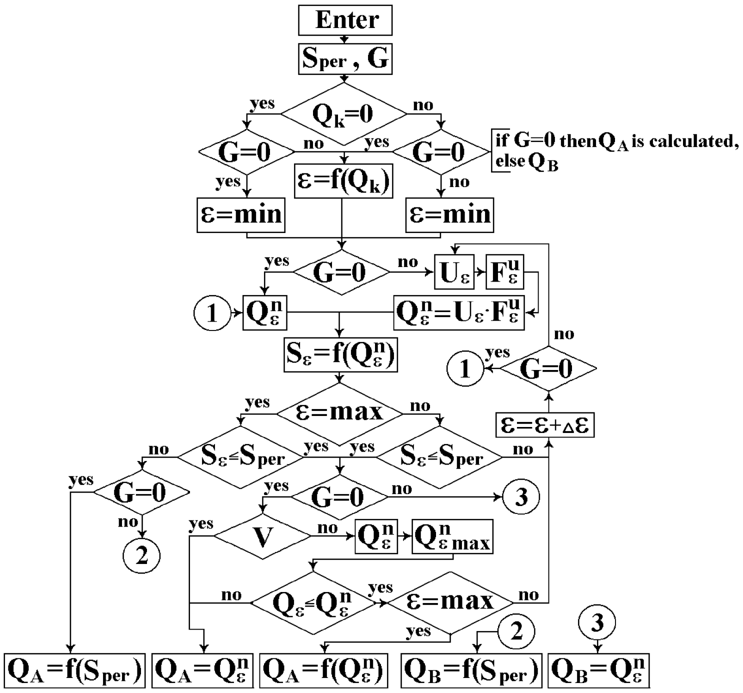

Figure 7. Algorithm of estimation of Q_A and Q_B. $\mathbf{Q_A}$ - component of exploitative reserves of the groundwater, provided with sources of completion of group A; $\mathbf{Q_B}$ - component of exploitative reserves of the groundwater, provided with sources of completion of group B; $\mathbf{S_{all}}$ - allowable decrease of aquifer level; $\boldsymbol{\varepsilon}$ - set of values of probability of exceeding, %; $\Delta\varepsilon$ - step under searching the values $\boldsymbol{\varepsilon_n}$; $\mathbf{Q_K}$ - the charge of water in sources of group A and B which obligatory presence is caused by economical and nature protection restrictions; $\mathbf{Q^n_\varepsilon}$ - the average-annual charge of the water in source of completion probability exceeding ε; $\mathbf{S_\varepsilon}$ - decrease of the level in exploited aquifer at withdrawal of charge $\mathbf{Q^n_\varepsilon}$; $\mathbf{Q^n_{\varepsilon max}}$ - the maximal daily - average charge of water in source of completion by probability of exceeding $\boldsymbol{\varepsilon}$; $\mathbf{Q_\varepsilon}$ - the charge of water which can be passed by sedimentations on contact of the aquifer and source of completion; $\mathbf{U_\varepsilon}$ - intensity of water inflow from sources of group B in exploited horizon within the area $\mathbf{F^u_\varepsilon}$; $\mathbf{V}$ - type of the power supply of group A ($\mathbf{Q_{A1}}$, $\mathbf{Q_{A2}}$, $\mathbf{Q_{A3}}$).

The set of works is devoted to mathematical modeling of the geofiltration. Adaptations of available techniques and programs for hydro-geological conditions of the republic are given in separate article [4]. It is optimum to create the permanent mathematical models of the geofiltration of all stratum-pore water fields of foothill plains of the republic, which will allow solving any direct, opposite and forecast hydro-geological problems based on MODFLOW software.

The mentioned above circuit of the consecutive tasks enables estimations of all natural and anthropogenous factors, which convert the structure of the groundwater resources, the forecasting of the reaction of underground hydrosphere on impact and management of the hydrosphere water resources in a whole.

4. Conclusions

Intensive production of fresh groundwater in different fields of the republic has led to negative consequences connected with the abrupt decrease or increase of the groundwater level. The damage is put to relict woods (Qusar field), river flow (Qanikh-Ayrichay field), geoecology (Absheron peninsula) and other environmental elements of the republic.

The analysis of the reasons of occurrence of negative consequences has specified essential defects in the process of investigation and engineering the water-wells of the groundwater, connected, mainly, with mistakes in accounting the restrictions on protection the environment.

Regional depression funnels, formed under impact of the groundwater production, transform natural hydrodynamical balance of whole hydrosphere waters and provoke new conditions of their interaction. Thus, new, so-called "developed" resources of groundwater are formed.

Necessity of the account of restrictions on protection the environment determined the necessity of modification of traditional methods of the estimation of operational reserves and resources of groundwater of the republic. As the calculated circuit, it is necessary to accept the whole hydrosphere in conditions of anthropogenous loading on the sphere. Hydro-geological conditions of the territory, technical means of water selection and size of allowable changes in the environment, which are inevitable when extracting and operating the groundwater, should define the size of operational reserves of groundwater. Based on the given circuit, in this article is given the technique and algorithm of the estimation of operational reserves of fresh groundwater in conditions of intensive economic activities.

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ON MODELLING OF GROUND AND SURFACE WATER INTERACTIONS

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Abstract: One of the largest native sulphur deposits in the world is located in Poland. Open-pits remain as the result of the completed exploitation in the Piaseczno and Machow mines. These pits after being filled with water will serve for recreation purposes. Complex interactions between reservoirs, rivers (Vistula and Trzesniowka) and two groundwater systems were analysed. Multivariant simulations were performed using MODFLOW and MT3D codes.

Keywords: groundwater and surface water interaction; flow and transport modelling; groundwater quality

1. Introduction

The interaction between groundwater and surface water is usually of great importance, because it influences quantity and quality of water in both systems. The interaction can be considered at the catchment scale as well as at the interface between groundwater and surface water. It occurs both in natural and anthropogenic conditions.

The aim of presented studies is to demonstrate the possibilities of groundwater flow and transport modelling for the evaluation of the relationship between groundwater and surface water. Presented two examples result from a

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research project related to the closing of sulphur mining in the Tarnobrzeg region, Poland. The first one describes the formation of water quality in the reservoir created in open pit of the Piaseczno mine. The second one shows the prediction of time and space changes in ground and surface water quality in the Trzesniowka River catchment in the case of ceasing of all pollution sources.

One of the largest native sulphur deposits in the world is located in the Tarnobrzeg region of Poland. The open-cast mining of the sulphur ore was completed in 1971 in the Piaseczno mine and in 1992 in the Machow mine. The exploitation of sulphur by Frash method at the Jeziorko was finished in 2001. At present, the exploitation of sulphur takes place only at the Osiek mine by that method. The open pits and waste-rock disposal sites remaining in Piaseczno and Machow interact with ground and surface waters (Figure 1).

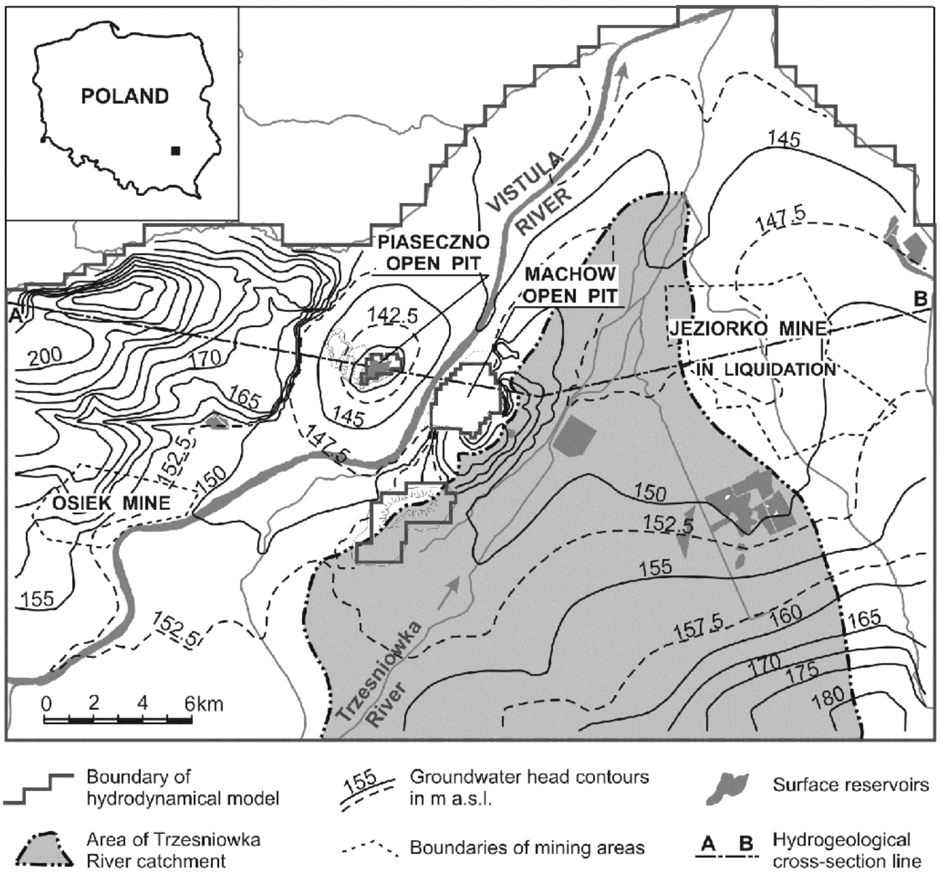

Figure 1. Hydrogeological map of the investigated area.

The conceptual model consists of two aquifers (Holocene-Pleistocene and Miocene) separated by an aquitard (Figure 2), with various types of boundary conditions including the artificial lakes and rivers. The MODFLOW (McDonald and Harbaugh, 1988; Harbaugh and McDonald, 1996) and MT3D (Zheng and Wang, 1998) codes with Processing Modflow interface (Chiang and Kinzelbach, 1998) were used for the 3-D numerical modelling (Kania, 2002). The total area of the model (about 900 km^2) was divided into the grid of 70 rows and 94 columns.

2. Hydrogeological Conditions

There are two main aquifers in the Tarnobrzeg sulphur ore region, the Holocene-Pleistocene and Miocene. The Holocene-Pleistocene aquifer consists of fluvial and fluvioglacial sandy gravel, forming a nearly continuous layer on impermeable Miocene clays (Figure 2). The aquifer thickness varies from 0 to 35 m, with the transmissivity of 300 to 600 m^2/d. The Holocene-Pleistocene aquifer is unconfined, and the atmospheric precipitation is the main source of aquifer recharge.The studied area is within the watersheds of the Trzesniowka, Leg and Koprzywianka Rivers which belong to the Upper Vistual River Basin.

The present (2002) distribution of hydraulic head in the Holocene-Pleistocene aquifer indicates that the regional flow of groundwater is in the direction of the Vistula River valley and is locally disrupted in the region of post-exploitation excavations at Piaseczno and Machow (Figure 1). Trzesniowka River strongly drains the eastern part of the area influencing the regional ground-water flow.

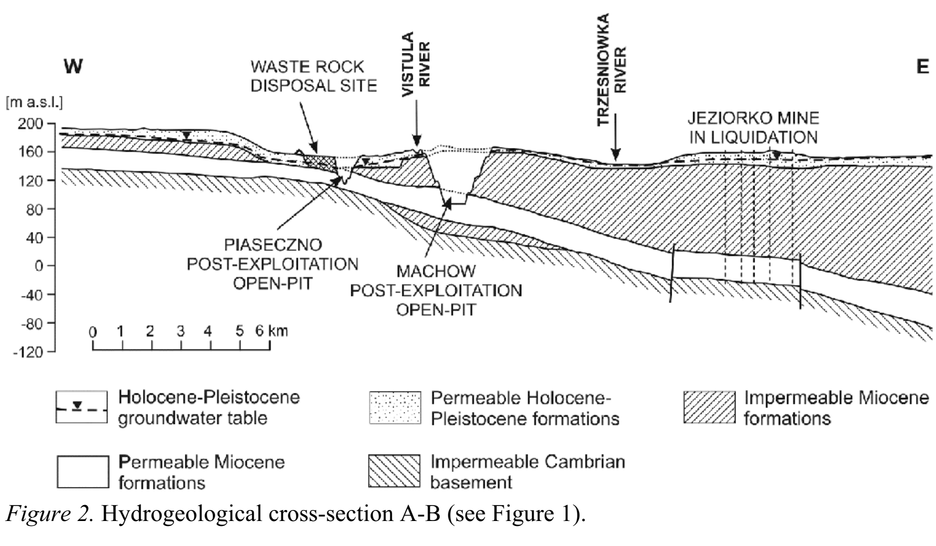

Figure 2. Hydrogeological cross-section A-B (see Figure 1).

The Miocene aquifer is related to the Chemical Series and Baranowskie Beds sediments resting on impermeable Cambrian basement. Both these layers form a confined aquifer isolated by impermeable Cracow clays. The recharge of the Miocene aquifer takes place at the outcrops in the western side of the Piaseczno excavation. The groundwater flow pattern in the aquifer is very variable, and depends on type, size, and arrangement of fissures. Additionally, it is locally influenced by changes resulting from sulphur exploitation by melting. Hydraulic conductivity values of sulphur-bearing limestone of the Chemical Series vary from 0.003 to 14.0 m/d, and of Baranowskie Beds from 0.07 to 4.7 m/d. The thickness of Chemical Series and Baranowskie Beds is within 5 to 15 and 20 to 70 m, respectively.

Local monitoring of groundwater in the Holocene-Pleistocene and Miocene aquifers has been conducted since 1997, in the area of post-exploitation excavations; its main aim is to control changes in hydrodynamic and hydrochemical conditions during, and after liquidation of mining activities.

3. Surface Water as Anthropogenic Systems

After closing mining operation, the pit lake was formed in the open pit at Piaseczno (Figure 3). The pit lake is fed mainly from the Holocene-Pleistocene and Miocene permeable formations. The water level of the reservoir is kept at 122 m a.s.l. by pumping.

The existing hydrodynamic model was used to identify the main sources o[0]f water supplying the pit lake from the Holocene-Pleistocene (Figure 3) and Miocene aquifers, resulting in:

1. The explanation of some processes occurring in the pit lake itself, including hydrogen sulphide oxidation coming from the Miocene aquifer inflow;
2. Description of the hydrochemical balance equation of the Piaseczno pit lake for present conditions;
3. Prediction of water quality changes in the outflow from the Piaseczno pit lake.

In the deeper part of the pit lake, water of higher salinity with hydrogen sulphide occurs due to the inflow from the Miocene aquifer. According to hydrological and hydrobiological studies, the reservoir is of meromictic type, where biological life exist (Dumnicka and Galas, 2005; Wilk-Wozniak and Zurek, 2005; Zurek, 2005b). The reason of that phenomenon is the water density stratification in the profile. Water density in the upper parts of the pit lake (γ_E) during cold period is still lower than in hypolimnion (γ_H) preventing mixing of the upper and lower layers (Figure 4).

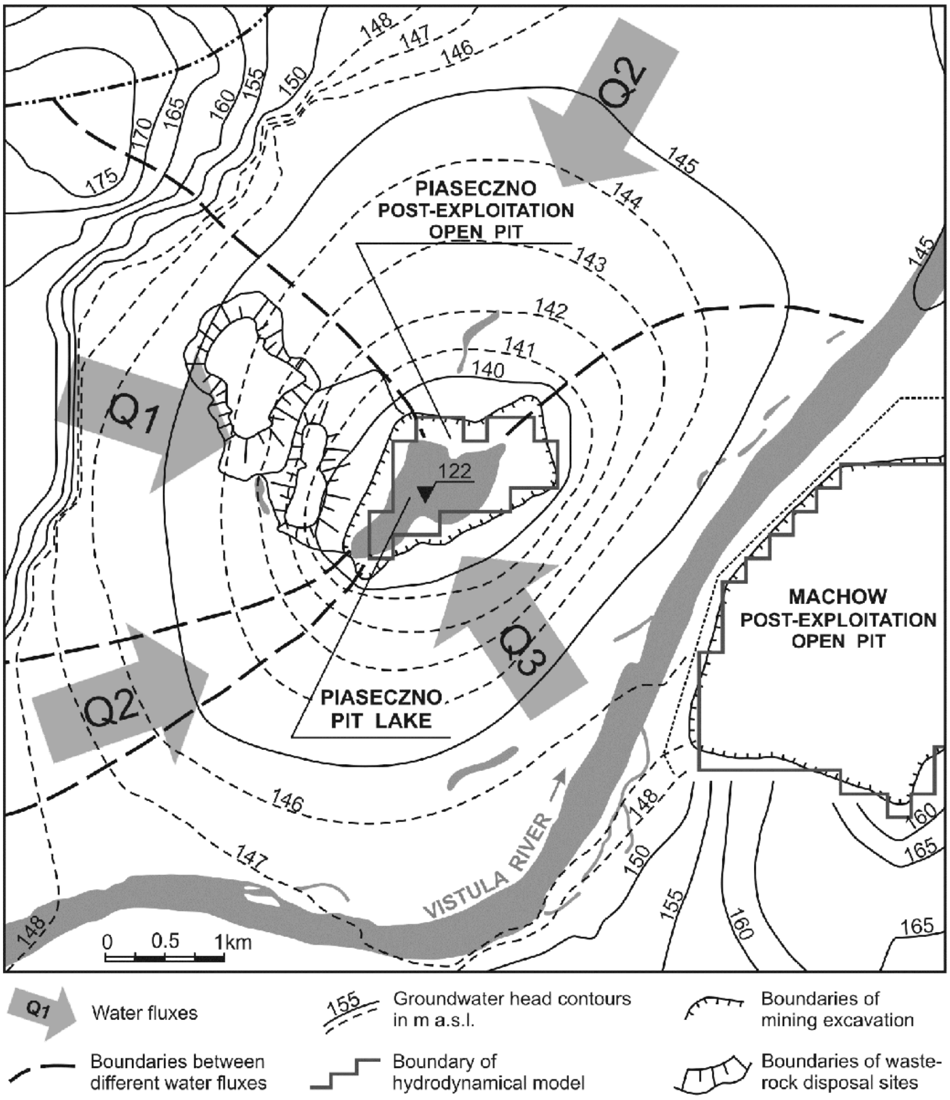

Figure 3. Hydrogeological map of the Holocene-Pleistocene aquifer.

Some processes occurring in the Piaseczno pit lake can be explained using simplified diagram presented in Figure 5. The inflow from the Miocene aquifer is of larger salinity with high hydrogen sulphide concentration up to 50 mg/dm^3. H_2S is next transported to the upper part of the reservoir, where is oxidized to sulphate as is shown with the simplified formula. Due to this process, H_2S is absent in the upper part of pit lake and development of biological life is possible there.

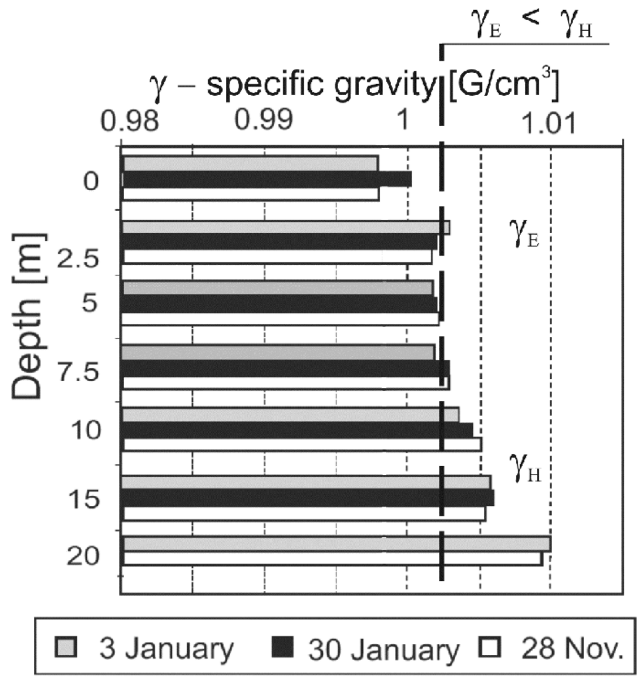

Figure 4. Water density stratification in the profile of the Piaseczno pit lake (from Zurek, 2005a, modified).

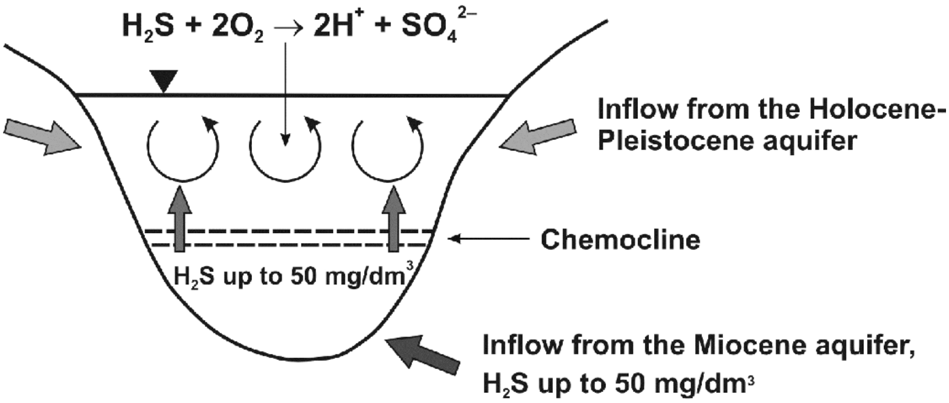

Figure 5. Simplified scheme of water circulation in the Piaseczno pit lake.

The calibrated flow model was used for the hydrochemical balance of the Piaseczno pit lake for present state (2002 with the water level of the reservoir at 122 m a.s.l.) and for prognosis after the final mining liquidation (water level at 146 m a.s.l). The hydrochemical balance of the reservoir is determined by water fluxes of different quality (Table 1):

- Q_P – precipitation on the reservoir,
- Q_E – evaporation from the reservoir,
- Q_S – runoff from the catchment area to the reservoir,
- Q_1 – inflow from the Holocene-Pleistocene aquifer passing through the waste rock disposal sites,
- Q_2 – inflow of non-contaminated water from the Holocene-Pleistocene aquifer,
- Q_3 – inflow from the Holocene-Pleistocene aquifer recharged mainly by the bank filtration from the Vistula River, contaminated by brines from the Upper Silesian coal mines dewatering in the upper part of river basin,
- Q_4 – inflow from the Miocene aquifer from the area of the hydrogeological window,
- Q_5 – local outflow to the Miocene aquifer, caused by drainage system of the Machow open pit dewatering – in the present state only,
- Q_6 – inflow from the Miocene aquifer, mainly from the eastern part of the studied area – in the prognosis state only.

Table 1. The primary components of the hydrochemical balance of the Piaseczno pit lake

Hydrochemical balance component[1]	Recharge/discharge – present state (2002), m^3/d	Recharge/discharge – prognosis state, m^3/d	Chloride – present state (2002), mg/dm^3	Chloride – prognosis state, mg/dm^3
$Q_P{\cdot}C_P$	1010	2630	10	10
$Q_E{\cdot}C_E$	1170	3040	0	0
$Q_S{\cdot}C_S$	200	60	35	35
$Q_1{\cdot}C_1$	4150	1430	90	90
$Q_2{\cdot}C_2$	3660	4360	35	35
$Q_3{\cdot}C_3$	4700	0	200	–
$Q_4{\cdot}C_4$	1840	190	500	500
$Q_5{\cdot}C_5$	370	0	167	–
$Q_6{\cdot}C_6$	0	410	–	3000

[1] explanations of symbols in the text

Furthermore, the indicator ion concentration in the individual fluxes was assumed (Table 1). The average concentration of indicator ion in the water of the Piaseczno pit lake is given by formula:

$$\overline{C} = \frac{\Sigma Q_i \cdot C_i}{\Sigma Q_i} \tag{1}$$

where Q_i is thedischarge of the i-th inflow/outflow flux to/from the pit lake, m^3/d and C_i is the indicator ion concentration at the i-th flux, mg/dm^3.

Chloride has been chosen as indicator ion, which is treated as conservative component. It does not arise due to processes occurring in the reservoir, and results as the average of chloride concentrations for different fluxes supplying the reservoir (obtained mainly from results of groundwater monitoring).

The quality of water in the reservoir is highly impacted by flux from the Holocene-Pleistocene aquifer which is recharged mainly by bank filtration from the polluted Vistula River (40.6%), and by inflow from the Miocene aquifer (39.7%). Under natural conditions, Vistula River was draining groundwater but now is recharging along the considerable distance (Figure 1 and 3).

For present situation (2002), chloride concentration measured in the discharge from the Piaseczno pit lake to the Vistula River (167.2 mg/dm^3) is in a good agreement with the result obtained from the calculation of the hydrochemical balance (165.3 mg/dm^3) which can be regarded as validation of the model (Table 2).

The chloride balance equation of the Piaseczno pit lake for prognosis is the same as for present state however, different quantities of recharging and discharging fluxes are used (Table 1). The calculations were performed under an assumption that chloride concentration in respective fluxes will not change.

Table 2. The hydrochemical chloride balance of the Piaseczno pit lake

	2002	PROGNOSIS
Water level in the reservoir [m a.s.l.]	122	146
Chloride concentration measured in the discharge to the Vistula River [mg/dm^3]	167.2	
Chloride concentration calculated from hydrochemical balance [mg/dm^3]	165.3	270.6
Chloride load in the discharge to the Vistula River calculated from hydrochemical balance [kg/d]	2344	1634

Although in the prognosis, the average chloride concentration in water supplying the pit lake will reach much higher value up to 270.6 mg/dm^3, the chloride load in the discharge to the Vistula River should be lower than present (from 2344 to 1634 kg Cl/d) due to lower outflow to the river.

4. Response of the River System after Changing the Contaminant Load in the Catchment Area

During mean low streamflow (MLQ) periods, rivers discharge mainly groundwater and wastewater. The quality of groundwater during base flow is the main factor responsible of the river water quality during MLQ.

In the case of river's catchments with shallow open groundwater systems (aspect ratio of the flow is $L/D > 10$, where L – the average length of the aquifer in the direction of subsurface flow, and D – the saturated thickness of the aquifer at the stream), the response of the system after changing the contaminant load has an exponential character, and is usually measured in tens of years. In such case, the flow geometry will have a small effect on base flow quality (Duffy and Lee, 1992; Duda et al., 1996).

Such typical response of the system was confirmed using modelling for the part of the Trzesniowka River catchment (Figure 1).

Among a number of modelling solutions, so called zero option has been chosen to demonstrate the prediction of time and space quality changes in ground and surface water in the case of ceasing of all pollution sources, including the nonpoint source contamination. The response of the system after changing the contaminant load becomes very nearly exponential, and is measured in tens of years. Chloride concentration (as an example of conservative component) decreases in time in the exponential way as is shown on the graphs for chosen sites (Figure 6). The same exponential type of the curve is obtained for the Trzesniowka River, which represents the mean for the whole catchment. It shows that results of groundwater monitoring can be used for prognosis surface water quality changes especially during MLQ.

Exponential character of the response of the system after changing the contaminant load let to the estimation of the half-time of attenuation for conservative components in the case of ceasing of all pollution sources. To find the half-time of attenuation for the Trzesniowka River, it is better to change the concentration scale from linear to logarithmic. Results of simulation indicate that for typical shallow river catchments, as Trzesniowka River basin, the process of contaminants attenuation will take tens of years after ceasing of all pollution sources (Figure 7).

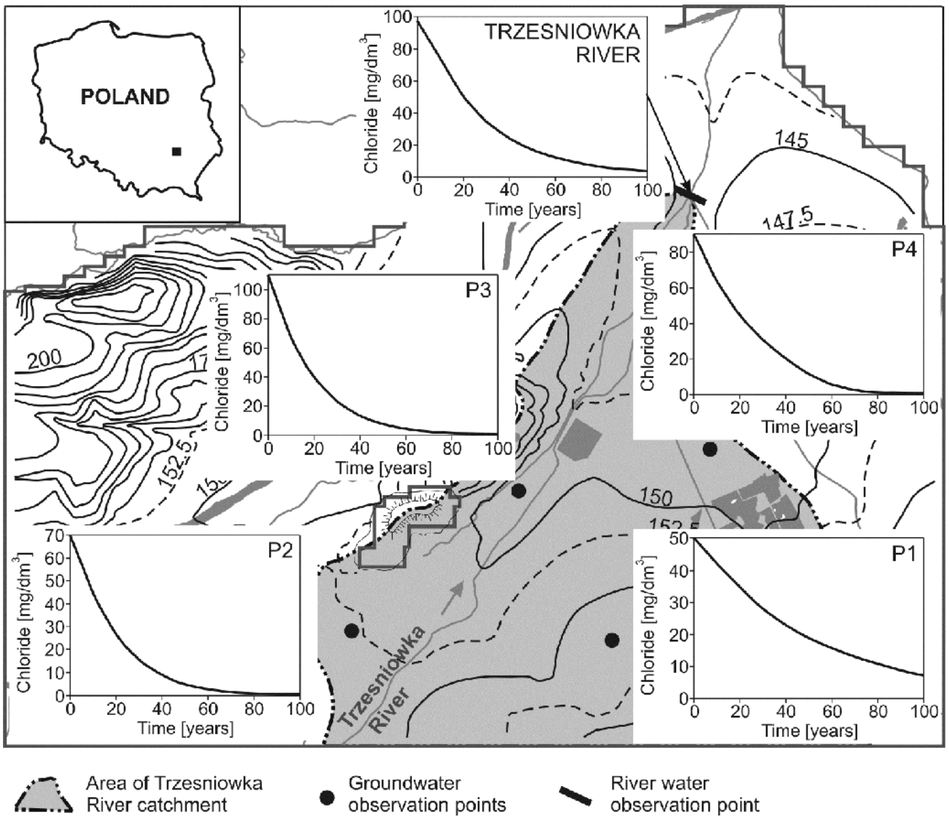

Figure 6. Response of the Trzesniowka River system after changing the contaminant load.

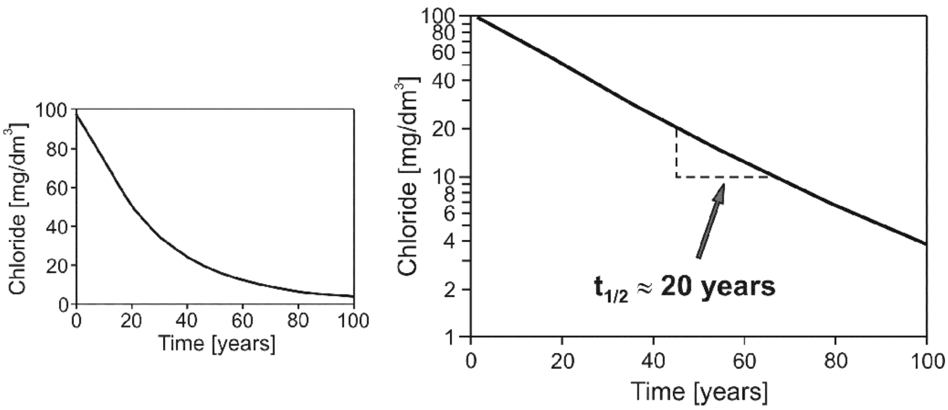

Figure 7. Half-time of attenuation for the Trzesniowka River for conservative components.

5. Conclusions

Two examples presented within this study indicate undoubtedly usefulness of groundwater flow and transport modelling for the evaluation of the interaction between ground and surface water systems. Such modelling gives the possibility to estimate the contributions of different water fluxes forming quality of water in the natural or artificial reservoirs. In turn, the observations of time and space changes of contaminants concentrations in groundwater seem to be a good indicator of base flow quality changes especially during MLQ. The response of the water system after changing the contaminant load, expressed by the half-time of attenuation for conservative components, is usually measured in tens of years.

ACKNOWLEDGEMENTS

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NITROGEN LEACHING IN AN AQUATIC TERRESTRIAL TRANSITION ZONE

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Abstract: Large parts of East Germany are characterised by sandy soils with a high hydraulic conductivity. The risk of nitrogen leaching and groundwater pollution may be minimised by organic farming, which has expanded in Germany during recent years. The study was conducted on an organically farmed rye field next to a lake in the state of Brandenburg between 2002 and 2004. In order to show how far organic farming may affect lake water quality, soil inorganic nitrogen ($CaCl_2$ extraction) and denitrification (acetylene inhibition method) were studied along an aquatic terrestrial transition zone (A = field site: 5 m above water level, B = field site: 1 m above water level, C = riparian zone with macrophytes: 0.5 m above water level). Although the field did not receive any organic and mineral fertiliser there was a nitrogen leaching from the field to the groundwater caused by the weather. Nitrogen loss during the winter was 29 kg N ha^{-1} y^{-1} and 12 kg N ha^{-1} y^{-1} in 2002/03 and 2003/04, respectively. Deviation between the two years seemed to be caused by great differences in precipitation. No nitrogen loss was observed from a control site. High denitrification was measured at sites B and C indicating an efficient nitrogen removal capacity within the riparian buffer zone.

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Keywords: organic farming; N_{min}; leaching; mineralisation; denitrification; groundwater; riparian zone

1. Introduction

Intensive farming in Germany leads to N surplus, which is up to more than 100 kg N ha^{-1} y^{-1}. Excessive N can be lost via two pathways. It is released as gas during microbiological transformation, or it undergoes leaching with the seepage water. N leaching is accelerated in sandy soils due to high rates of precipitation. Primarily lowland sites and aquatic terrestrial transition zones demand special attention because they are close to groundwater and surface water. Since large parts of the northeast of Germany belong to the subglacial lowland with sandy soils, there is a considerable risk of nitrogen translocation due to leaching (Wendland et al., 1994). On the other hand, long residence times of the groundwater and good hydrogeochemical conditions may support N removal mechanisms.

Organic and integrated farming are becoming more important in order to meet environmental requirements. Particularly organic farming has been well established in the study area with numerous lakes. With abdication of mineral fertiliser, organic farming is appropriate for sandy soils of the federal state of Brandenburg. In order to evaluate the close proximity of agricultural land and environmentally protected areas, a small lake with a size of 3 ha was selected. The lake is surrounded by a 5-40 m wide riparian macrophyte strip. Wetlands and ecosystems such as found in the study area are considered to be effective for nonpoint source pollution control (Hill, 1996; Blackwell et al., 1999; Hoffmann et al., 2000; Hefting, 2003). Next to the lake, the fields were organically farmed for rye production in the years 2002 and 2003. 2002 was a wet year in contrast to 2003, which had an extremely low precipitation rate. Such climatic distinctions might have affected both, the microbiological activity in the soil (Hellebrand et al., 2005) and the extent of nitrogen leached out from the soil (Shepherd, 1996; Kleinhenz et al., 1997).

The objective of this study is to estimate the output of nitrogen from the agricultural field and to compare it with the N output from nearby undisturbed grassland (control). It should be shown how far organic farming and climatic factors may affect groundwater and lake water quality and how far the littoral vegetation works as a buffer strip removing nitrogen.

2. Methods

Soils of one organically farmed field and one non-cultivated control area were studied during a two-year period from April 2002 to March 2004. Three sampling sites were located along two transects with a slope of 5% meeting the groundwater fed lake. Each transect had an upland site (A) 5 m above water level, a lowland site (B) 1 m above water level and a riparian site (C) 0.5 m above water level. Site A had a Kf value of 4.0 x 10^{-6} m s^{-1} and 2.5 x 10^{-7} m s^{-1} on the field transect and the control transect, respectively. Soil samples were taken with a corer (∅ = 2 cm) from 0-30 cm and 30-60 cm depths four times and were then mixed to obtain composite samples for the two layers. Soil samples were stored in a refrigerator before being analysed on the next day.

Usually the N_{min} method is applied to obtain the plant available nitrogen in the soil and to provide information about the N demand to the farmer. During the winter time, immobilisation and uptake of N by plants can be neglected. Therefore, the difference in N_{min} between October/November and in the following March can be considered as N loss. According to the agricultural monitoring programme of the state of Brandenburg, mineral nitrogen (N_{min}) is defined as the sum of ammonium and nitrate nitrogen that is determined after the extraction of wet soil samples with 0.0125 M $CaCl_2$ solution (shaken for 1 hour). NH_4-N was measured photometrically by the indophenol blue method and NO_3-N by ion chromatography. Total carbon (TC) and total nitrogen (TN) were measured by dry combustion method using an elemental analyser.

Gaseous nitrogen release from the soil was detected after incubation in a nitrogen atmosphere for 4 hours at 22°C. Two assays, each with 3 replicates, were applied to obtain the rate and the potential of denitrification. Denitrification was measured using the acetylene-blockage technique (15% C_2H_2) described by Yoshinari and Knowles (1976). To obtain the denitrifying potential 1 mmol NO_3 was amended to the incubation vessels. Gas samples of 1 ml were injected into a gas chromatograph (Fisons Instruments, GC 8340) equipped with a 3 m long packed column (Haye Sepe D 100/120 mesh). The oven temperature was programmed between 70°C and 120°C. N_2O is detected by a ^{63}Ni electron capture detector (ECD) operating at 320°C. The carrier gas was helium with a flow rate of 15 ml min^{-1}. All data for extractable nitrogen and denitrification rates were related to soil dry weight after drying at 105°C.

Climatic factors were monitored by a weather station in Potsdam (TOSS GmbH Potsdam) located 30 km north of the study area.

3. Results and Discussions

3.1. CLIMATIC CONDITIONS

During the study period, mean air temperature did not differ very much and measurement showed 10.9°C and 10.6°C in 2002 and 2003, respectively. However, there was a major difference in precipitation with a relatively high rate of 634 mm in 2002 and a very low rate of 272 mm in 2003. The extreme dryness in 2003 was observed in most regions of Germany, leading to heavy crop losses.

3.2. EXTRACTABLE NITROGEN

From the high elevated site (A) to the riparian site (C) total and extractable nitrogen (N_{min}) increase along both transects. On the field of winter rye an average of 0.2% of total N in the soil was N_{min} compared to 0.1% on the control transect (Table 1).

Table 1. Mean values of soil characteristics within 0-60 cm depths of the two transects at the lake. Upland sites (A), lowland sites (B) and the riparian sites (C) were sampled from April 2002 to March 2004

		A	B	C	A	B	C
		Winter rye		Littoral	Control		Littoral
pH		6.0	5.5	5.3	5.9	6.4	5.7
Electrical conductivity	$\mu S\ cm^{-1}$	30	58	83	41	48	70
Water content	%	4.5	18.5	34.6	4.9	5.3	36.9
Total C	$g\ kg^{-1}$	4.1	35.0	74.7	5.4	5.1	46.2
Total N	$mg\ kg^{-1}$	352	1,685	2,836	400	517	2,340
NH_4-N	$mg\ kg^{-1}$	0.20	0.77	4.09	0.24	0.38	1.96
NO_3-N	$mg\ kg^{-1}$	0.64	1.97	0.21	0.20	0.38	0.13
N_{min}	$mg\ kg^{-1}$	0.84	2.74	4.30	0.44	0.76	2.09
N removal (Oct. - March)	$kg\ ha^{-1}$	8.7	32.4	47.6	−0.5	−4.1	28.1

The low organic matter content, reflected by low total carbon and the low water content at the site A of both transects indicated a high hydraulic conductivity of the soil. With a rate of 70% sand on the control transect and 75% sand on the field transect, the soils have little water holding capacity implying a high risk of nutrient leaching.

On the field site A, the mean extractable nitrogen content was 0.84 mg kg^{-1} during the two-year study period. By contrast with this relatively low value, N_{min} content was 3 to 4 times higher at the low field site B where a sink of nitrogen can be assumed. Some of the nutrients may have been derived by leaching from the site A and subsequent groundwater runoff. By contrast with the control sites of A and B, where both NH_4-N and NO_3-N form similar amounts of N_{min}, NO_3-N predominates on the sites planted with rye. The situation at site C of both transects is quite different. The mean N_{min} content is 4.3 mg kg^{-1} and 2.1 mg kg^{-1}, respectively and consists primarily of NH_4-N.

Enhanced NH_4 concentrations at site C may be explained by accelerated mineralisation following warming and aeration of the organic soil. This is most obvious at the field site for 2003/2004, when the precipitation rate was very low.

Low NO_3 concentrations at site C can be explained by a suppression of nitrification in the waterlogged environment, plant uptake and/or a high denitrifying activity. This pattern can also be observed at the control site (Figure 1). By contrast with the impact of riparian forest buffers on agricultural non-point source pollution (Snyder et al., 1998; Addy et al., 1999), relatively little is known about the function of herbaceous buffer strips (Flite et al., 2001). Our results provide some information that non-cultivated herbaceous riparian zones can reduce the NO_3 concentrations of waters draining from upslope cultivated agricultural soils.

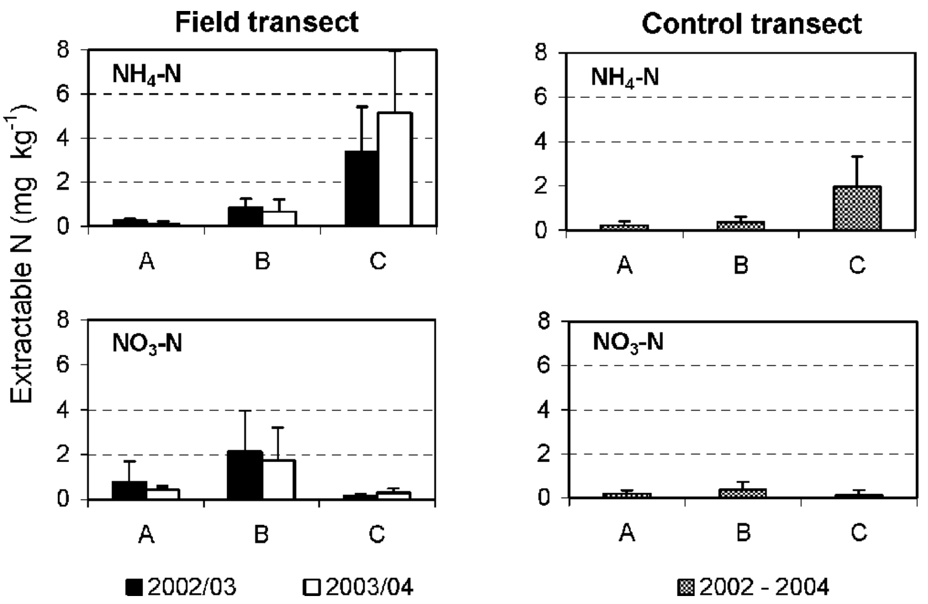

Figure 1. Extractable NH_4–N and NO_3–N in the 0-60 cm soil depths along the field and the control transect.

NO_3 concentrations were highest at the field site B in 2002/03 and 2003/04. NO_3 concentrations of this site were 5 times greater than those of control site B and it could be influenced by more elevated sites, such as site A, as a result of leaching and run-off. Since the field was not fertilised during the study period and total N of the field and control transect were in the same range, the enhanced NO_3 concentrations at the field sites A and B probably arise from the much more intensive mineralisation due to soil cultivation. Organic residues, which remain on the field after harvest, can explain the higher amount of easily available N on the field. Part of this enhanced soil N_{min} may be subject to leaching (Shepherd and Lord, 1996; Kavdir et al., 2005).

Nitrogen also becomes available after mineralisation of soil organic matter (SOM), which contains a considerable amount of N. Estimation of N mineralisation could be made using SOM content values. If 2% of the total organic N in the surface soil is mineralised annually, a soil with 1% SOM content can mineralise about 45 kg N ha^{-1} y^{-1} (Schepers and Mosier, 1991). Therefore if we calculate the potential N mineralisation of study soils using the data for Table 1 (soil bulk density of 1.5 g cm^{-3} and 0-30 cm depth), we find that N mineralisation potential from A and B (rye) and C is 32, 152 and 255 kg N ha^{-1} y^{-1}, respectively. N mineralisation from control sites A, B and C with 36, 47 and 211 kg N ha^{-1} y^{-1}, respectively is in the same range. Consequently tillage operations in planted areas seem to have increased NO_3 concentrations on the field sites compared with control sites. Such high amounts of N which may become available by minerlisation can explain both N removal by harvest (average of sites A and B = 39 kg N ha^{-1} y^{-1}, Kern, 2004) and N loss (site A = 8.7 kg N ha^{-1} y^{-1} and site B = 32.4 kg N ha^{-1} y^{-1}) during the winter. However, these are rough estimates, which will vary due to temperature, precipitation and tillage (Wienhold and Halvorson, 1999).

3.3. NITROGEN REMOVAL

The most important removal paths for nitrogen from the field are plant uptake and harvest, leaching, surface run-off and denitrification. The amount of N scavenged depends in part on plant type, plant growth, soil type, amount of fall soil inorganic N, and weather. Removal of scavenged nitrogen by harvesting was 64 kg N ha^{-1} in 2002 and only 13 kg N ha^{-1} in 2003 due to low water supply and poor plant growth (Kern, 2004). Similar differences in the loss of N during the winter are calculated by changes in N_{min} from October/November to March of the following year. Taking the means of field site A and B, there is an overwinter N loss of 29.1 kg N ha^{-1} in the first winter and 11.9 kg N ha^{-1} in the second winter of the study period. It becomes apparent that the dry weather reduced the N loss considerably. However, even under such extreme conditions, the N loss on the field transect was much higher compared with the control transect (Table 1). Negative values at site A and B indicate that there was no loss but a N input by mineralisation. In the same range as on our field transect, Wendland et al. (2004) estimated a leaching rate of 15 kg N ha^{-1} y^{-1} within the catchment area of the River Elbe. The authors stressed, however, that about

90% of the diffuse N input into the groundwater can be degraded in the aquifer system before entering the surface water. Contrasting results were presented by Brye et al. (2001) who found that less than 25% of the N leached below the root zone in an agro-ecosystem was subjected to denitrification. Limited denitrification was substantiated by insufficient lengths of saturated soil conditions and the supply of dissolved organic carbon.

Much lower N uptake by plants and lower amount of leaching/ denitrification in 2003/04 can be explained by the extreme dry weather in 2003, when the region of Potsdam received only 272 mm water by precipitation. Thus the total outflow of water and dissolved nutrients was very low.

Anthropogenic N input from the atmosphere is suggested to be about 30 kg N ha^{-1} in Germany (Dannowski, 1995). This amount of N should have been completely consumed in 2002. However, it is even possible that N is being enriched on the field sites due to atmospheric deposition in 2003 (dry year). The N_{min} removal data presented in Table 1 are integrated values, which do not allow us to distinguish between the removal paths such as leaching and gaseous release. It is rather difficult to present a real N balance, which at least should include precise data of one removal path.

A part of the biomass produced at the riparian site of the lake does not undergo a complete mineralisation due to the waterlogged environment. Carbon and nutrients may accumulate, resulting in a peat production. In the case of nitrogen, an immobilisation of nitrogen can be estimated for the last 60 years, which would explain all the N_{min} lost during the study period and extrapolated to a period of 60 years.

3.4. GASEOUS NITROGEN TRANSFORMATION

Although the objective of this study was not to quantify N leaching losses, it is possible to achieve a better understanding of the N dynamic along the aquatic terrestrial transition zone by studying microbiological processes, particularly denitrification.

Denitrification was restricted to the 0-30 cm uppermost layer at all sites indicating that there was no lateral flow of NO_3-rich water in deeper soil layers. Denitrification does not play an important role in N release from the control site, by contrast with the field site (Figure 2). Highest rates of denitrification were measured at field site B according to highest NO_3 concentrations in the soil. That means that N can be efficiently removed on this low field site before the mobile NO_3 pass through the riparian site C, reaching the lake afterwards. Alternative wetland buffer zones can be even more effective for NO_3 removal than riparian zones adjacent to the water body as reported by Blackwell et al.

(1999). Disadvantages of conventional riparian buffer zones may derive from by-passing due to pathways which follow drains and ditches.

Although the riparian site C is characterised by a high carbon content, which is favourable for denitrification (Maag et al., 1997; Burt et al., 1999), denitrifying activity is quite low by contrast with the field site B. This was caused by very low concentrations of NO_3, which seems to be limited for denitrification in the soil of site C. NO_3 supply should have been interrupted because nitrification is suppressed under anaerobic conditions in waterlogged environments (Phillips, 1999). The assay with the amendment of NO_3 showed that site C had the same ability to denitrify as site B, confirming the results from Davidsson et al. (2002) on flooded and drained peatland soils. Furthermore, the denitrifying potential of site B was nearly as same as the denitrification rate without addition of NO_3. Consequently, the denitrification capacity of site B fully used all NO_3 supply in both study years. Evidence for a high denitrifying potential of study soils supports the low amount of NO_3 supply to the riparian zone C.

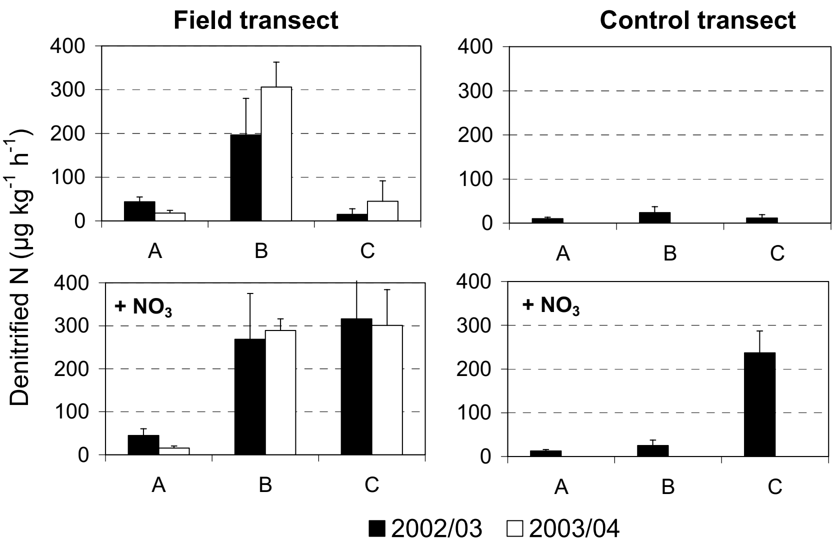

Figure 2. Denitrifying activity and its potential after NO_3 amendment in the soil layer 0-30 cm.

4. Conclusions

A good balance between the current organic farming N supply and N uptake by crop plants can be concluded for the study area. Both slight N leaching of the root zone and the low slope characterise the agro-ecosystem under study as an effective sink for N. Even if there is no mineral and organic fertilisation, N translocation from the field to the groundwater cannot be completely excluded. However, the nitrogen that is leached from the study field can be denitrified or fixed as organic matter in the riparian zone. Thus the lake under study seems to be well protected from N pollution.

The key factors controlling the subsurface NO_3 retention are residence time, low oxygen concentration, and high content of electron donors such as organic carbon. These factors are correlated to hydrogeological site conditions. With prevailing glacio-fluviatile sands and moraine deposits, the whole European Pleistocene Lowland can be considered as an NO_3 removing aquifer.

This study is an example of how far an aquatic transition zone may serve as a buffer against NO_3-loaded groundwater. The leaching of NO_3 into the groundwater must not necessarily be a problem if the pollution of drinking water resources can be excluded. NO_3 can be removed with high efficiency as long as groundwater-born NO_3 meets favourable conditions for denitrification. In this way, NO_3 inflow into surface water, particularly into river systems, with its low denitrifying capacity can be prevented.

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INTERACTIONS BETWEEN GROUNDWATER – SURFACE WATER AND TERRESTRIAL ECO-SYSTEMS

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Abstract: The effective management of a water resource requires a balance to be struck between the water requirements of humans and the natural ecology of a catchment. This task becomes especially difficult in catchments that are characterised by significant groundwater-surface interactions. This is largely because the vital role that groundwater plays in supporting riverine and wetland ecosystems is poorly understood and is difficult to quantify. Once understood, the implications of the interdependence of groundwater and eco-systems often present serious challenges for the future management of the water environment. This paper explores some of the challenges posed by the following elements: (i) identifying and characterising groundwater-surface water interactions - hydraulic connections, water quality and ecological dependence, (ii) determining environmental needs - ecological flows/levels and water quality requirements, (iii) assessing the impact of groundwater abstractions and pollutants on groundwater fed rivers, lakes and wetlands; and, (iv) some implications of the above to the future management of the water environment.

Keywords: surface-subsurface interactions, water management, terrestrial ecosystems

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1. Introduction

Sustainable groundwater management demands that a balance is struck between the water requirements of the natural ecology of a catchment and the impacts of human activity. Activities of concern are those that may reduce the quantity (flow or level) and/or the quality of the water upon which the ecology depends. This balance must be informed by both a scientific understanding of the catchment and broader socio-economic factors.

However, the role that groundwater plays in supporting the catchment's ecology is often difficult to quantify. This is partly because groundwater flows and groundwater chemical inputs to surface water and terrestrial ecosystems cannot usually be measured and are difficult to accurately quantify using standard hydro-geological approaches. In addition, the dependence of an ecosystem upon groundwater is not directly related to the amount of groundwater, but depends upon the specific needs of the ecology.

Groundwater dependent ecosystems (GDEs) may be found in groundwater fed rivers, estuaries, wetlands and lakes, at springs, and areas where the groundwater supports a root zone. To understand the phenomenon of GDEs it is necessary to develop an understanding of the scale and nature of the groundwater-surface water (groundwater-surface water) interactions (e.g. groundwater flow to a river or wetland), the occurrence of near-surface groundwater, and the dependence of the ecology upon the groundwater input. This presents a number of challenges both to the catchment scientist and the regulator.

This paper explores some of these challenges, drawing on practical experience of implementing Catchment Abstraction Management Strategies (CAMS) (Environment Agency, 2002i) and the early stages of the European Union Water Framework Directive (WFD, 2000) in England & Wales. CAMS currently address the issue of sustainable water resource management in rivers only, and are based upon the Environment Agency's Resource Assessment and Management (RAM) Framework (Environment Agency, 2002ii). The RAM Framework establishes river flow objectives and assesses the impacts of groundwater and surface water abstractions and impoundments on river flows. The WFD requires the identification of GDEs and the establishment of environmental standards in the form of water levels, flows, and chemical criteria, as necessary to protect groundwater and surface water supported ecosystems. It also requires an assessment of the current and predicted impact on GDEs from groundwater pressures (abstraction & pollutants).

2. Hydraulic Interactions, Water Quality and Ecological Dependence

GDEs are water dependent ecosystems that are either wholly or partly dependent upon groundwater. In general terms groundwater may be important to the ecology because:

1. Groundwater provides a significant proportion of the water requirements of a habitat (e.g. high base-flow rivers).
2. Groundwater continues to provide water throughout sustained dry periods.
3. Groundwater has a chemical composition or temperature upon which certain flora or forna are dependent for their survival.

If any of these criteria are met then the ecology can be classed as groundwater dependent. A good practical test of the degree of groundwater dependence is to ask the question 'How would the ecology fare if the groundwater flow or chemical input were reduced or ceased?'

Studying hydraulic interactions can therefore help us to identify GDEs. The hydrologist and hydrogeologist have a range of tools and approaches that can be used to explore the scale and nature of groundwater-surface water interactions, to estimate the amount of groundwater flow or chemical input to different habitats. These tools and approaches include hydrograph separation, piezometry, river flow accretion profiling, analytical and numerical models (water resource and pollutant transport models), and the use of natural or artificial tracers. However, an understanding of the 'dependency' of the ecology on the groundwater must also include an ecological assessment of the presence/abundance of the species that are regarded as ecologists as being groundwater dependent.

In some river habitats a high degree of groundwater dependence is obvious. For example in parts of Southern England, the flows in the groundwater fed 'chalk rivers' are heavily dominated by groundwater baseflow with a characteristic chemistry that sustains an ecology that is strongly associated with this habitat. These hydraulic and ecological features thus combine to strongly indicate a habitat that is heavily groundwater dependent.

In other cases the dependency may be far less obvious. For example, a 'headwater' section of a river that derives relatively little of its flow from groundwater. At face value this would seem to indicate a low dependence on groundwater. However, this small contribution from groundwater (providing a stable temperature) is in fact vital for the survival and development of salmonid eggs that live in the gravel beds through which the groundwater percolates (Crisp 1990; Dent et al., 2000). A similar picture emerges for wetlands whereby relatively small contributions of groundwater to wetlands have been found to support significant communities of groundwater dependent flora (Environment Agency and the Centre for Ecology and Hydrology, June 2004). Consequently, to accurately identify GDEs, it is necessary to study hydraulic interactions, water quality and the dependence of the ecology on the groundwater input. These examples illustrate the fact that ecological dependence is not directly related to the size of the groundwater contribution. A practical implication of this finding is that an ecological assessment to test for the presence of species that are considered to be groundwater dependent will be required even in habitats where the groundwater contribution may be minimal.

New approaches that combine hydraulic, water quality and ecological assessments to systematically identify and describe GDEs are actively being developed in England and Wales as part of the implementation of WFD. For example 'Impact Assessment on Wetlands: Focus on Hydrological and Hydrogeological Issues, R&D Technical Report W6-091/TR1', (Environment Agency and Centre for Ecology and Hydrology, June 2004).

3. Ecological Water Requirements

The ecology of a catchment has its own specific water requirements both in terms of quantity (level/flow) and quality. Establishing these needs is a prerequisite to managing and protecting the water resources of a catchment. Consequently, rivers, estuaries, lakes and wetlands may be assigned environmental standards that relate to minimum flow, level or quality regimes. These standards are generally derived from an understanding of ecological tolerances/ecological needs, many of which are habitat and species specific. The WFD is acting as a 'driver' for the review and amendment of the Environment Agency's approaches to setting ecologically based environmental standards.

As part of its existing CAMS, the Environment Agency currently uses the following steps in its RAM Framework in the preparation of an ecologically acceptable flow regime for rivers throughout England & Wales:

- Produce a conceptual model of the catchment that includes detailed consideration of groundwater-surface water interactions.
- Describe the river as a sequence of sectors and assign assessment points.

For each assessment point:

- Produce estimates of naturalised river flows.
- Estimate the ecology that should normally be sustained under naturalised (or baseline) flows and naturalised water quality conditions.

- Define river flow objectives based upon the flow/level requirements of the ecology defined in 4. and with reference to the naturalised river flows.
- Compare recent actual flows & full licensed uptake against river flow objectives to determine status of the surface water resource (consider impacts from surface water & groundwater pressures separately – links with groundwater procedure below).
- Carry out follow-up surveys of the ecology and flows/levels to validate or amend the ecological river flow objectives as necessary.

The RAM Framework groundwater procedure is carried out in parallel with the above RAM Framework river flow procedure. The groundwater procedure links directly to the river flow procedure such that the role of groundwater in supporting the river flow objective is fully recognised, as are the groundwater abstraction pressures that reduce its base-flow contribution to the river (Environment Agency, 2002 (ii)).

The RAM Framework is currently under review as the Environment Agency considers changes that may be required to better meet the requirements of the WFD with respect to river flows and related aquifer management. In addition, new approaches to assess the ecological flow & level requirements of lakes, estuaries and headwaters are being developed.

New requirements under the WFD to ascertain the impacts of groundwater abstraction and pollutants upon groundwater water dependent terrestrial ecosystems (GDTEs) are also driving a review of existing approaches to wetland protection. For example, Scottish & Northern Ireland Forum for Environmental Research, Research Project WFD62, Wetlands and Groundwater Interactions, 2005/2006, unpublished). Though much work has been carried out in recent years on understanding wetland water supply mechanism in an attempt to derive ecological water level and flow needs for wetlands, requirements for ecological quality criteria are less advanced. Further work on the tolerances of wetland ecology to pollutants is required to establish robust chemical thresholds that can be used to assess pollution risk and for future regulation.

Similarly, new ecologically based environmental quality standards (chemical criteria) are under development for rivers, lakes & estuaries.

Recent international surveys of approaches to setting ecological requirements at the catchment scale have identified more holistic approaches than have hitherto been adopted in England & Wales (Dunbar et al., 1997 and Scottish & Northern Ireland Forum for Environmental Research, Research Project WFD48. Development of Environmental Standards for groundwater abstractions, 2005, unpublished): In South Africa and Australia a multidisciplinary approach based around expert opinion has been developed called the holistic or building block approach. This considers the complete river ecosystem, including the source area, riparian zone, wetlands and groundwater system. It seeks to allocate an 'environmental reserve' that is designed to sustain a healthy river flow regime that takes into account nutrient cycling, community dynamics, animal movements the establishment of plants and a range of other factors. (Adams and Acreman., 1998). These approaches offer some attractive features and are currently subject to detailed review by the Environment Agency.

Another international survey, carried out as part of Scottish & Northern Ireland Forum for Environmental Research, Project WFD53, Framework for Determining Regulatory Standards for Groundwater Abstractions, 2005 (unpublished). This project has identified the wide use of traditional groundwater balance methods whereby a proportion of the groundwater recharge is 'reserved' for the environment (or other legitimate uses). The proportions reserved for the environment vary widely both between countries and sometimes within them and probably reflect socio-economic pressures as well as ecological considerations. The research concluded that groundwater balance is a relatively crude instrument that is best used in conjunction with additional consideration of the impacts that groundwater abstractions may have on specific receptors i.e. receiving waters and GDTEs.

The same recommendation is also valid with respect to catchment or aquifer wide groundwater quality standards. Hence aquifer wide quality standards should also be used in conjunction with additional quality standards that are specifically designed to protect the ecology of the receiving waters and GDTEs.

Despite recent improvements, defining the fundamental water requirements (quantity and quality) of the ecology of a catchment remains an inexact science. In general it still remains difficult to provide the necessary scientific evidence to support unequivocal assertions about the ecological flow, level & quality requirements of a habitat. Furthermore, the existing approaches generally fail to quantify separately the ecological requirements for groundwater. This would require consideration of the special attributes of groundwater: its quality and its role in supporting GDEs during prolonged dry periods. Though scientifically challenging, a separate estimate of the ecological needs for groundwater (as a subset of ecological water need) could lead to better recognition of the role of groundwater and a better informed management of groundwater.

4. Impacts of Groundwater Abstraction and Groundwater Pollutants

Quantifying the impact that groundwater pressures (abstraction and/or pollutants) are having (or may have) upon the groundwater dependent ecology also presents significant technical challenges (Figure 1). In addition to the usual hydrogeological uncertainties (e.g. aquifer parameters), the degree and nature of the hydraulic connection with the habitat (e.g. a wetland or river) is often uncertain. Hydraulic connection can have a very large influence on the spatial and temporal impacts of an abstraction (Kirk and Herbert., 2002) or a pollutant.

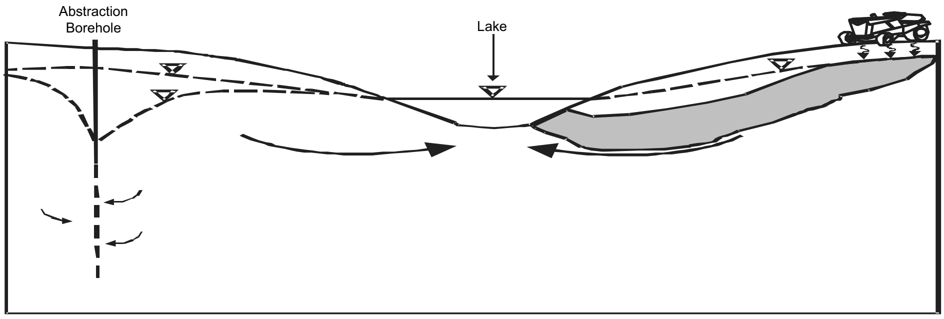

Figure 1. Groundwater-Surface Water Interactions and Potential Impacts of Groundwater Pressures (Pollutants and Abstractions) on a Surface Water System.

The Environment Agency has produced a tiered decision support system for the estimation of the impact of groundwater abstractions on river flows (which is supported by software tools) (Environment Agency, 1999). But even with detailed guidance and supporting analytical & numerical modelling, uncertainties remain, even in the best estimates. These uncertainties can of course be reduced by testing and monitoring; though short term pumping tests can often produce results that are poor indicators of the long term or steady state impacts from groundwater abstractions. (Op cit).

A key lesson that has been learned from the Environment Agency's work on the impact of groundwater abstractions on river flows is the importance of recognising that groundwater abstractions will ultimately result in an equal reduction of groundwater discharge somewhere in the catchment. Consequently, the Environment Agency's RAM Framework requires the user to distribute steady-state abstraction impacts across the catchment model as reductions in natural groundwater discharges (e.g. rivers or spring flows), thus properly accounting for net losses from the system. The Environment Agency's guidance and supporting analytical tools allows the user to explore the time lags between starting or ceasing pumping and the predicted impact on river flows. This is especially useful in exploring control measures. For example, it can be demonstrated that ceasing groundwater abstractions during low flows may not benefit river flows for several weeks or months. Hence in this instance, a temporary cessation of abstraction in response to the onset of low river flows would not be a viable management option for the rapid restoration of the current, depleted flows.

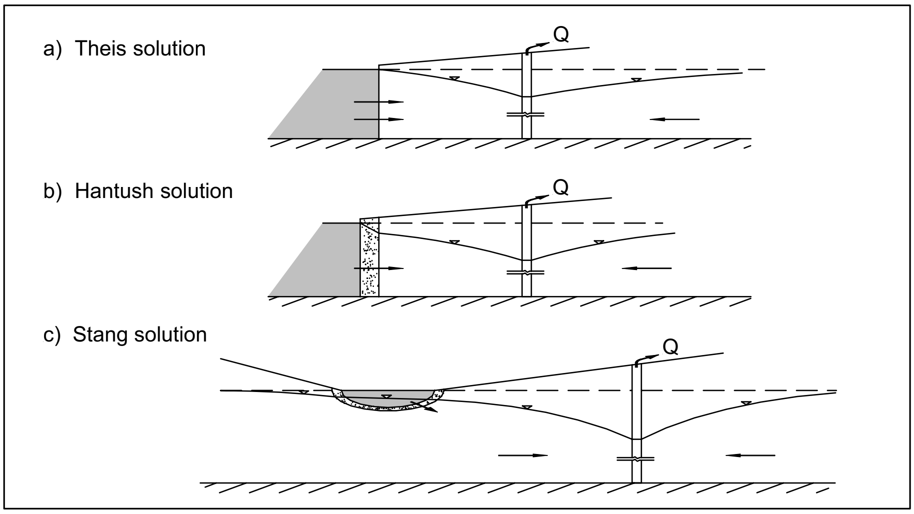

Figure 2. Conceptual Models used in modelling the impact of groundwater abstractions upon river flows.

The Environment Agency is also developing a new decision support systems to help assess the impact of groundwater abstractions on GDTEs (Environment Agency and Centre for Ecology and Hydrology, 2004) in line with requirements for the WFD. These draft procedures recognise the uniqueness of each GDTE and require a conceptual model to be developed for each site where a risk from groundwater pressure has been identified (Figure 2). It comprises a tiered, risk-based approach that commences with a GIS based risk assessment and progresses to site surveys and detailed modelling in accordance with the scale of the risk identified in the preceding tier.

Quantifying the impacts of groundwater pollutants on GDEs in surface waters or terrestrial sites involves additional complications. Flow paths, dilution and attenuation are key factors that introduce uncertainties. Transport and attenuation in the unsaturated zone and hyporheic zone are generally not monitored and therefore are not well quantified. Our conceptual model must also often include extensive 'time lags' between the time that a pollutant enters an aquifer until the time it is discharged from the aquifer. For example, diffuse agricultural fertiliser applied to the land may take several decades for the resulting diffuse pollutants to emerge in a receiving surface water or in a GDTE.

The Environment Agency has recently embarked upon a new nitrate modelling project that seeks to predict trends in nitrate in groundwater and rivers to 2027 (Environment Agency, Nitrate Framework Project, Science Ref. SC050029). It is comprised of both statistical and numerical modelling elements. The results will support on-going work on the WFD which has identified nitrate in groundwater as a major issue. The modelling work will be based upon a conceptual model of each catchment and will maximise the use of available monitoring data in validating the model results. Quantifying the nitrate flux from groundwater into surface water is considered to be one of the most challenging aspects of the project, with a lack of knowledge about the hyporheic zone being a significant contributing factor.

The effect of the hyporheic zone at the catchment scale on the groundwater chemical flux to a river is currently not well understood in the UK. It is hope that new research in the UK will soon improve our knowledge of the pollution attenuation processes that occur in the hyporheic zone and allow these findings to be ‘scaled up’ to help inform the Environment Agency’s response to the WFD. For example, Environment Agency Science Project.

‘Groundwater – surface water interactions in the hyporheic zone’, Ref. SC030155, (Smith, 2005). In particular, understanding hyporheic zone processes may prove important to the modelling of the fate and transport of nitrate in groundwater as it enters surface waters.

In England and Wales, often the challenge is to determine the impact that groundwater pollutants or abstractions are having or will have upon the ecology, *relative* to other existing pressures such as sewage treatment works discharges or direct surface water abstractions to rivers, or land drainage activities around wetlands. In these circumstances the groundwater pressure ‘signal’ is often difficult to differentiate from other pressures. Modelling can play a role here in testing the possible effects of various pressures.

Whilst many approaches and modelling techniques are available to estimate the impact of abstraction or pollutants on receiving waters or terrestrial ecosystems, there exists considerable scope for validation and improvement of approaches. Issues that need further attention include: Practical methods to ascertain the degree of hydraulic connection between groundwater and surface waters and groundwater and GDTEs; the effect of the hyporheic zone on groundwater flux into surface water systems (particularly with respect to nitrate); and the response of GDEs to changes in pressures (Figure 3).

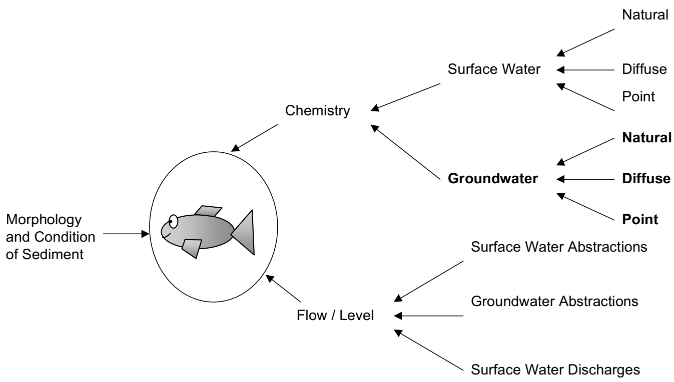

Figure 3. Multiple pressures acting upon an ecosystem.

5. Implications for Water Management

In general it still remains difficult to provide the necessary scientific evidence to support unequivocal assertions about the ecological flow, level & quality requirements of a habitat or to discern impacts that can be attributed to specific pressures. This presents a significant challenge for the effective management of groundwater that seeks to protect GDEs.

Consequently there exists a need for a systematic framework to firstly quantify the interactions between groundwater and surface water, and between groundwater and terrestrial habitats, and secondly to explore the dependence of the ecology on these interactions. The framework should offer a tiered approach that can be applied at the catchment scale but must be able of capturing site-specific knowledge where appropriate.

Key components of the proposed framework that would benefit from improved technical approaches or validation include:

- Estimation of the degree of hydraulic connection between groundwater, surface water and between groundwater and terrestrial ecosystems.
- Ecological (species) indicators to help assess groundwater dependence.
- The setting of ‘ecological thresholds’ from which ‘environmental standards’ can be produced in order to protect GDEs.

- Quantifying ecological responses to changes in groundwater pressures.
- Catchment-wide estimation of the attenuation capacity of the hyporheic zone especially with respect to diffuse groundwater pollutants.

Given the inherent uncertainties in determining the groundwater needs of GDEs and their vulnerability to groundwater pressures, it is essential that risk based decision support systems are developed that can accommodate these uncertainties and still guide management decisions to protect or restore GDEs. This will require the regulator to embrace the principles of adaptive management, accepting expert opinion in areas where quantitative assessment is weak or lacking. Measures taken in response to perceived risk to a GDE should be closely monitored to judge their efficacy and to improve conceptual understanding of the habitat and its relationship to groundwater pressures.

It is hoped that the growing recognition of the importance of groundwater in supporting catchment ecosystems will allow groundwater management to assume its full role in integrated river basin management. This should help to produce more holistic, well-informed approaches to integrated catchment management based upon sound conceptual models that are fully cognisant of the role of groundwater in the catchment.

DISCLAIMER

Any opinions expressed herein are those of the author and do not necessarily reflect the views of the Environment Agency.

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NATURAL WATER SUPPLY AND FERTILIZATION INTERACTIONS ON CROPS YIELD IN FRAGILE AGROECOSYSTEM

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Abstract: Drought and excess rainfall conditions result in the breakdown between man and his environment. The effects of drought and excess rainfall situations on a community require extraordinary efforts to cope, oftentimes with outside international aid. The effects of global climate change on water resources may be hidden by natural climate variability. With a warmer climate, drought and excess rainfall cases could become more frequent, severe, and longer-lasting. The potential increase in these natural hazards is of concern given the stresses they place on water resources and agricultural production, and high costs that result from these hazards. For these reasons, the effects of rainfall variation (quantity, distribution) and fertilization (N, P, K, Ca, Mg) on soil (Haplic Luvisol-acidic sandy brown forest soil) system was evaluated for crop yield (rye, potato, winter wheat, and triticale) as part of a 43-year field experiment that began in 1962 at Nyírlugos (Nyírség, a fragile eco-region of Eastern Hungary). The ploughed soil (0-25 cm) had the following agrochemical characteristics: pH (H_2O) 5.9, pH (KCl) 4.7, hydrolytic acidity 8.4, hy_1 0.3, humus 0.7%, total N 34 mg · kg^{-1}, ammonlactate (AL) soluble-P_2O_5 43 mg · kg^{-1}, AL-K_2O 60 mg · kg^{-1}. From 1962 to 1980 the experiment consisted of 2 x 16 x 4 x 4 = 512 plots and from 1980 of 32 x 4 = 128 plots in split-split-plot and factorial random block designs. The gross plot size was 50 m^2. The average fertilizer rates in kg · ha^{-1} $year^{-1}$ were nitrogen 45, phosphorus 24 (P_2O_5), potassium 40 (K_2O), magnesium 7.5 (MgO) until 1980, and nitrogen 75, phosphorus 90 (P_2O_5), potassium 90 (K_2O), calcium 437.5 ($CaCO_3$)

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magnesium 140 ($MgCO_3$) after 1980. Averaged rainfall quantities over many years, in the experimental years, during phenological phases in the many years, and in the experimental years for rye were 567, 497, 509, 452 mm, and for winter wheat 586, 509, 518 and 467 mm. Rainfall deviations from the many years's average in the experimental years and during the phenological phases of potato were –3%, –13% and of triticale 2% and –3%. During the vegetation period, the relationships between rainfall quantity, nutrition (N, P, K, Ca, Mg), and yield were characterized by polynomial correlations (Rye, control: R = 0.99; N: R = 0.84; NP: R = 0.84; NK: R = 0.91; NPK: R = 0.85; NPKMg: R = 0.65. Potato, control: R = 0.98; N: R = 0.95; NP: R = 0.96; NK: R = 0.95; NPK: R = 0.98; NPKMg: R = 0.96. Winter wheat, control: R = 0.59; N: R = 0.57; NP: R = 0.76; NK: R = 0.54; NPK: R = 0.67; NPKMg: R = 0.71. Triticale, control: R = 0.35; N: R = 0.28; NP: R = 0.47; NK: R = 0.37; NPK: R = 0.63; NPKCa: R = 0.67; NPKMg: R = 0.67; NPKCaMg: R = 0.62). Maximum yields for rye: 4.0 $t \cdot ha^{-1}$, potato: 21.0 $t \cdot ha^{-1}$, winter wheat: 3.4 $t \cdot ha^{-1}$, and triticale: 5.0-6.0 $t \cdot ha^{-1}$ were observed when the respective natural rainfall amount was in the range of 430-500, 280-330, 449-495 and 550-600 mm. At rainfall amounts above and below these ranges, there was a corresponding quadratic reduction in the yield.

Keywords: drought; flood; nutrient, crop; yield

1. Introduction

Climate change is recognized as a serious environmental issue (Easterling et al., 1999; Johnston, 2000). Presently, the build-up of greenhouse gases in the atmosphere and trends in emissions suggest that we can expect significant climate changes probably into the 21th century (Hulme et al., 2002; Márton, 2002a; Rajendra, 2004; Barrow et al., 2000). A decade ago, researchers asked what effects climate change may have on the ecology. Today, researchers are asking how to respond to, and take advantage of, the effects of climate change (Márton, 2002b). Answers to this new question require information regarding the anticipated effects and associated adaptive measures required at local and regional scales. Important information should be gathered on whether yields can be maintained, if and where new crops should be grown, if new processing plants will be required, and degree of competition for water. Information on methods of adaptation is required for government officials, landscape planners, stakeholders, farmers, producers, processors, supermarkets, and consumers.

Many agricultural investigations focused on understanding the relation between mean climate change and crop production (Várallyay, 1992; Rajendra, 2004). Few investigations, however, studied the effects of climate variability on agriculture crop yields (Németh, 2004). The response of agricultural crop yield to changes in climate variability was attributed primarily to changes in the frequency of extreme climatic events (EU, 2003). Recent studies demonstrated a greater effect on the frequency of extreme climatic events than changes in the mean climatic response (EM, 2004). Hence, in studying the effects of climatic change on crop production, the changes in the climatic variability and associated weather patterns should be included (Barrow et al., 2000).

Changes in weather patterns were observed throughout Europe (including Hungary) as early as 1850. Among the natural consequences of changing weather patterns, years of drought (rainfall deficit) and wet (rainfall excess) conditions, resulted in problems among plant nutrition and field crop production (European Union, 2003). Whereas rye (*Secale cereale* L.), potato (*Solanum tuberosum* L.), winter wheat (*Triticum aestivum* L.), and triticale (Kádár et al., 2000; Márton, 2002abcd) are crops of worldwide importance, limited research exists about the effects of climate change on these crops. All four crops are sensitive to the prevailing weather conditions (such as rainfall) and, for this reason, understanding the effects of anthropogenic climate change on their production is important. In addition to rainfall, these crops require a high level of soil macronutrients: nitrogen (N), phosphorus (P), potassium (K), calcium (Ca) and magnesium (Mg). This paper describes findings related to climate-change and fertilisation effects on crop yield at an experimental site in Hungary.

2. Materials and Methods

The effect of rainfall (quantity and distribution) on crop fertilisation factors, such as macronutrients and yield, were studied during a long-term (1962 to 2001) field experiment on a Haplic Luvisol (acidic sandy brown forest soil) in northeastern Hungary-Nyírlugos. The ploughed soil (0-25 cm) had the following agrochemical characteristics: pH (H_2O) 5.9, pH (KCl) 4.7, hydrolytic acidity 8.4, hy_1 0.3, humus 0.7%, total N 34 mg $\cdot$ kg^{-1}, ammonlactate (AL) soluble-P_2O_5 43 mg $\cdot$ kg^{-1}, AL-K_2O 60 mg $\cdot$ kg^{-1}. From 1962 to 1980 the experiment consisted of 2 x 16 x 4 x 4 = 512 plots and from 1980 of 32 x 4 = 128 plots in split-split-plot and factorial random block designs. The gross plot size was 50 m^2. The fertilization treatments were for rye N: 45, P_2O_5: 24, K_2O: 40, MgO: 7.5 kg $\cdot$ ha^{-1} $\cdot$ year^{-1}; potato N: 75, P_2O_5: 24, K_2O: 75, MgO: 15 kg $\cdot$ ha^{-1} $\cdot$ year^{-1}; winter wheat N: 45, P_2O_5: 24, K_2O: 40, MgO: 7.5 kg $\cdot$ ha^{-1} $\cdot$ year^{-1} from 1962 to 1980 and N: 75, P_2O_5: 90, K_2O: 90, $MgCO_3$: 140 kg $\cdot$ ha^{-1} $\cdot$ year^{-1} from 1981 to 1990; and triticale N: 75, P_2O_5: 90, K_2O: 90, $CaCO_3$: 437.5, $MgCO_3$: 140 kg $\cdot$ ha^{-1} $\cdot$ year^{-1} from 1991 to 2001 in the form of 25% calcium ammonium nitrate, 18% superphosphate, 40% potassium chloride, calcium carbonate and magnesium sulphate. The groundwater table was at a depth of 2-3 m below the surface. Rainfall amounts (deviation in rainfall from the average over many years: dry year –10 - –20%, drought year –20% over, wet year +10 - +20%, year with excess rainfall +20% over) and other related data were determined based on traditional Hungarian (Harnos, 1993) and Research Institute for Soil Science and Agricultural Chemistry of the Hungarian Academy of Sciences (Márton, 2002abcd) standards, and MANOVA (Multivariate Analysis of Variance) by SPSS test (SPSS Inc., 1988).

3. Results and Discusions

Relationships between crop (rye, potato, winter wheat, triticale) x climate (rainfall quantity and distribution) x mineral nutrition (N, P, K, Ca, Mg) system changes with respect to agricultural sustainability at a long-term Hungarian experimental field site. The most important results are given below.

3.1. CLIMATE-RAINFALL-CHANGE AND ARTIFICIAL FERTILIZATION EFFECTS ON RYE YIELD

i. Certain experimental years were characterized by extremes in rainfall variability. For example, there was one average year with 450 mm of rainfall (1966), one wet year with 721 mm of rainfall (1970), and three dry years with 353, 369, 378 mm of rainfall (1964, 1968, 1972). ii. Rainfall extremes characterized by drought or wet years did not cause significant differences on the rye yield without fertilization (average year: 1.66 $t \cdot ha^{-1}$, drought year: 1.51 $t \cdot ha^{-1}$, over rainfall year: 1.47 $t \cdot ha^{-1}$). iii. Yields varied from 2.01 to 3.04 $t \cdot ha^{-1}$ under low (N: 30 $kg \cdot ha^{-1}$ and NP, NK, NPK, NPKMg combinations) fertilization input. During drought and wet years, the respective yields decreased by 14% and 10%. iv. At mean fertilization (N: 60 $kg \cdot ha^{-1}$ and NP, NK, NPK, NPKMg combinations) levels, the maximum yield reached 3.6 $t \cdot ha^{-1}$ during average rainfall year. In years with excess rainfall, however, the rye yields decreased with average fertilization treatments by 20%. v. During an average rainfall year with typical fertilization (N: 90 $kg \cdot ha^{-1}$ and NP, NK, NPK, NPKMg combinations), the maximum yield reached 3.8 $t \cdot ha^{-1}$; the maximum yields decreased by 17% and 52% during the respective conditions of drought and excess rainfall. The negative effects of excess rainfall conditions, however, decreased by 20-25% with the use of Mg treatments. vi. Polynomial correlations between rye yields and rainfall during the vegetation period (control: R = 0.99, N: R = 0.84, NP: R = 0.84, NK: R = 0.91, NPK: R = 0.85, NPKMg: R = 0.65) indicated that optimum yields develop in response to rainfall amounts in the 430-470 mm range. Under and above these rainfall ranges, the yields decrease according to a quadratic relation.

3.2. CLIMATE-RAINFALL-CHANGE AND ARTIFICIAL FERTILIZATION EFFECTS ON POTATO YIELD

i. The trial years (1963, 1965, 1967, 1969, 1971) were characterized by recurrent rainfall extremes during the vegetation seasons for potato. Three periods had average rainfall, while two periods were dry. ii. Droughts in the winter or summer half-year had similar effects on the yield. Precipitation deficiency in the winter could not be counterbalanced by average rainfall during the vegetation period, and the effect on yield was similar to that of summer drought. iii. Yield and quality were influenced by rainfall to a greater extent than by fertilization. iv. In vegetation periods subject to drought conditions, the yield of potato could not be maintained by fertilization alone, as the yield decreased by 35%. Also, economic yields could not be achieved with poor nutrient supply even with a normal quantity and distribution of rainfall. v. The unfavorable effects of climate anomalies (drought or rainfall excess) on the yield formation, yield quantity of potato depended on the time of year. vi. Using regression analysis, the correlation between rainfall and yield were determined for the control nutrition system: R = 0.98, N: R = 0.95, NP: R = 0.96, NK: R = 0.95, NPK: R = 0.98, NPKMg: R = 0.96. Optimum yields of 17-20 $t \cdot ha^{-1}$ developed in response to rainfall in the 280-350 mm range.

3.3. CLIMATE-RAINFALL-CHANGE AND ARTIFICIAL FERTILIZATION EFFECTS ON WINTER WHEAT YIELD

i. Climate-rainfall-conditions during winter wheat years were determined primarily by precipitation during average (1982 and 1989), drought (1976 and 1990), dry (1974) and wet (1978 and 1980) years. ii. The experimental climate-rainfall character were formed by winter half-years (October-March), months (October-September), pre-months of sowing (august), critical sequential month number in vegetation seasons (September-July) and critical sequential month number in experimental years (September-August). iii. In average rainfall years without any mineral fertilization, the wheat yield stabilized at the level of 1.8 $t \cdot ha^{-1}$. With N, P, K and Mg fertilizer input, the minimum and maximum yields were 2.7 and 4.1 $t \cdot ha^{-1}$. The yield only increased with a whole NPK and Mg completed NPKMg treatment. iv., Without mineral fertilization on the control plots, the yield decreased 39% during a drought year compared to average year. On N, NP and NK combinations yields were diminished to 48%.

Drought damage on yield production increased to 51% with NPK and NPKMg applications. v. In drought and average years, yields were similar on the control plots. Yields were decreased for an average year by 20% and 16% with N, NP, NK and NPK, NPKMg treatments. vi. During excess rain conditions and without fertilizer application, the yields decreased more dramatically (56%) as compared to drought conditions (39%). The yield was reduced by 47% with unfavorable (N, NP, NK) nutrition. But the negative effect of excess rainfall was diminished on NPK and NPKMg treatments to 41%. vii. Correlations between yield and precipitation during vegetation seasons (control: R = 0.59, N: R = 0.57, NP: R = 0.76, NK: R = 0.54, NPK: R = 0.67, NPKMg: R = 0.71) indicated that optimum yields developed in response to rainfall in the 450-500 mm range. Above or below this rainfall range yields decreased quadratically.

3.4. CLIMATE RAINFALL-CHANGE AND ARTIFICIAL FERTILIZATION EFFECTS ON TRITICALE YIELD

i. During dry and drought conditions, the respective yield of the control areas was 14% and 36% lessthan for average years. The application of N alone, or of NP and NK treatments, led to yield losses of 45% and 24%, respectively, while that of NPK, NPKCa, NPKMg or NPKCaMg caused a further 22% drop during both types of years. ii. In the wet years, the yield decreased by 14% in the unfertilised plots; remained unchanged in the case of N, NP, or NK nutrition; and increased by 31% with NPK, NPKCa, NPKMg and NPKCaMg treatments. In the very wettest year, the yields were similar to those in the average year. iii. The relationships between rainfall quantitiy during the vegetation period N, P, K, Ca and Mg nutrition and yield were characterised by polynomial correlations (control: R = 0.35, N: R = 0.28, NP: R = 0.47, NK: R = 0.37, NPK: R = 0.63, NPKCa: R = 0.67, NPKMg: R = 0.67, NPKCaMg: R = 0.62). iv. Maximum yields of 5.0-6.0 $t \cdot ha^{-1}$ were achieved in the rainfall range of 550-600 mm. At values above and below this range, the grain yield reduced quadratically.

4. Conclusions

We can state that both, drought and excess rainfall conditions resulted dramatically significant negative effects between fertilization (N, P, K, Ca, Mg) and crop (rye, potato, winter wheat, triticale) yield on Haplic Luvisol in the Nyírlugos long-term field fertilization experiment in the fragile Hungarian (Nyírség) agro-ecosystem under forty years from 1961 to 2001.

During drought years yield of rye, potato, winter wheat and triticale was decreased with an average of 14%, 35%, 46% and 28%, and in the wet years yield's drop was in the case of rye 10%, winter wheat 56% and triticale 9%.

The relationships between rainfall quantitiy during the vegetation period and N, P, K, Ca, Mg nutrition and crop yield can be characterised by polynomial correlations. The total regression coefficients renged in the case of rye from 0.65 to 0.99, potato: 0.95-0.98, winter wheat: 0.54-0.76, triticale: 0.28-0.67 in depence on the different nutrient application. At values above and below this range, the grain yield reduced quadratically. Behind this scientific concrete results we draw the farmers attention to that the drought and excess rainfall extreme effects are significantly negative on crop yields, generally.

So, this paper has a basic importance for describes findings related to climate-change and fertilisation effects on the crop yields for farmers to their crop production optimalization under changing climate in the nearest future.

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GROUNDWATER FLUXES IN ARID AND SEMI-ARID ENVIRONMENTS

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Abstract: In arid and semi-arid areas groundwater fluxes such as recharge and groundwater evapotranspiration are substantially different than in moderate climates. Rainfall, if present is intensive and occurs in short, spatio-temporally variable events. This results in substantially more spatio-temporally variable recharge pattern than in moderate climates. In arid and semi, in contrast to moderate climates also the importance of groundwater evapotranspiration (ET_g) is significantly higher. This is because: a) recharge and therefore entire groundwater flux input is low so then even low contributions of (ET_g) are significant; b) large water deficit at the surface and shallow subsurface, results in common groundwater uptake (groundwater transpiration) by plant (mostly tree) roots and by groundwater evaporation through vapor and capillary upward water movement.

Keywords: groundwater; fluxes; evapotranspiration; recharge

1. Introduction

In many arid and semi-arid regions groundwater is the main and therefore critical source of water supply. Moreover, there are countries in the world such as e.g. Botswana, where groundwater resources are the only source of water supply. In all such countries groundwater is strategic receiving a lot of attention particularly with respect to its sustainability.

A. Baba et al. (eds.), Groundwater and Ecosystems, 225–236.

The sustainability of groundwater resources depends on hydrogeological constraints such as net recharge to the aquifers, aquifer transmissivity, aquifer storage, groundwater quality and on the anthropogenic constraints related to the human impact upon groundwater (Lubczynski, 2005). Aquifer transmissivity and aquifer storage are naturally constrained and rather not to be changed. Groundwater quality depends on the natural state of an aquifers but also upon the human impact that largely constraints groundwater sustainability in qualitative as well as in quantitative manner. Regarding quantitative aspect of sustainability of groundwater resources however, the most influencing is the net recharge flux (R_n) because it constrains aquifer replenishment.

2. Groundwater Fluxes

For any part of an aquifer (also for any individual element of an aquifer or a cell of a groundwater model) the following groundwater flux balance equation can be applied (Figure 1):

$$q_{\mathrm{Gin}} + R = q_{Gout} + ET_g \pm \Delta S \pm q_{ex} \qquad [L/T] \qquad (1)$$

where q_{Gin} is the internal aquifer inflow to the selected aquifer element per unit area of this element; R is the external groundwater recharge into the selected aquifer element; q_{Gout} is the internal aquifer outflow from an individual aquifer element per unit area of this element; ET_g is the groundwater evapotranspiration from an individual aquifer element; ΔS is the change of storage of an individual aquifer element and q_{ext} is the external sinks/sources from/to an individual aquifer element per unit area of this element.

q_{Gin} and q_{Gout} terms in Equation 1 are the fluxes representing the internal (means that q_{Gin} and q_{Gout} are from/to the similar aquifer elements of the same flow system) groundwater inflows and outflows (flows per unit area of the aquifer element). Both, q_{Gin} and q_{Gout}, according to Darcy law depend on the aquifer transmissivity (T) and hydraulic gradient (I) interrelated with each other. In general, large hydraulic gradients are associated with low aquifer transmissivity and opposite. In principle then, the higher the transmissivity the better for the well productivity is. However, it happens also (e.g. hill slope aquifers), that deep incised drainage structures with large transmissivities, may enforce substantial gradients resulting in quick, unwanted dewatering of aquifers. In arid and semi-arid aquifers q_{Gin} and q_{Gout} are typically low as all other fluxes.

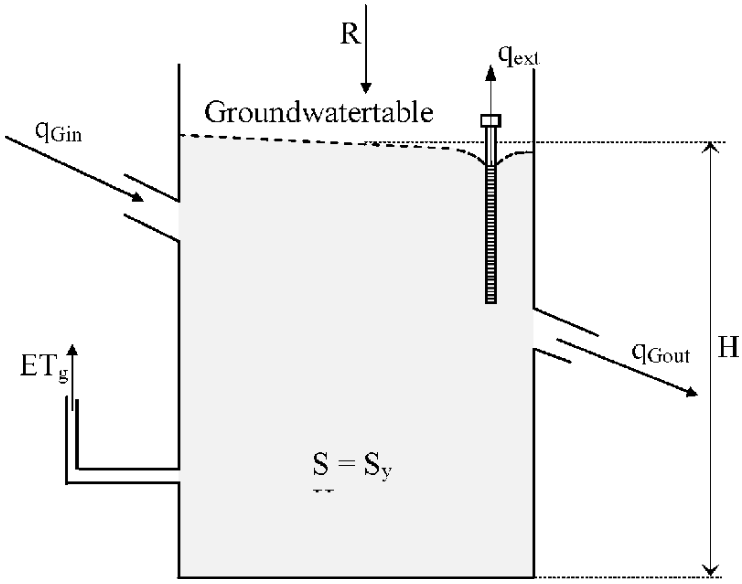

Figure 1. Schematic reservoir diagram representing an individual aquifer element.

q_{ext} represents the external, human related sink (negative) and source (positive) fluxes, representing external with respect to the aquifer system analyzed inputs and outputs of water. The most common example of q_{ext} is well abstraction.

ΔS change of storage depends on the specific yield (S_y) and on the change of the hydraulic head (ΔH) in the analyzed period; in case of unconfined aquifers it represents the change of groundwater table informing about changes in gravitational storage and in case of confined aquifers the change of potentiometric surface informing about changes in the elastic storage.

R is a groundwater recharge flux [L/T] which is the amount of water that reaches a saturated part of an aquifer and/or its capillary fringe per unit area. The spatio-temporal variability of recharge directly depends on the rainfall rate and rainfall distribution, therefore is largely constrained by the climatic zone. In general, the more arid the climate, the less rain and the larger is the spatial and particularly temporal variability of recharge. This large variability is characterized by long dry seasons and short or very short wet seasons which typically generate all the recharge of the year. The yearly average recharge (*R*) in arid, and in semiarid climates, is typically low. In cases of aquifers overlain by thick unsaturated zone, it can be even as low as only ~5 mm/y as for example in case of the Kalahari in Botswana (de Vries et al., 2000).

Not only seasonal but also long term temporal distribution of recharge is irregular in arid and semi-arid areas. It varies from no recharge in “dry” years characterized by low rain to largely exceeding yearly averages in “wet” years (Figure 2).

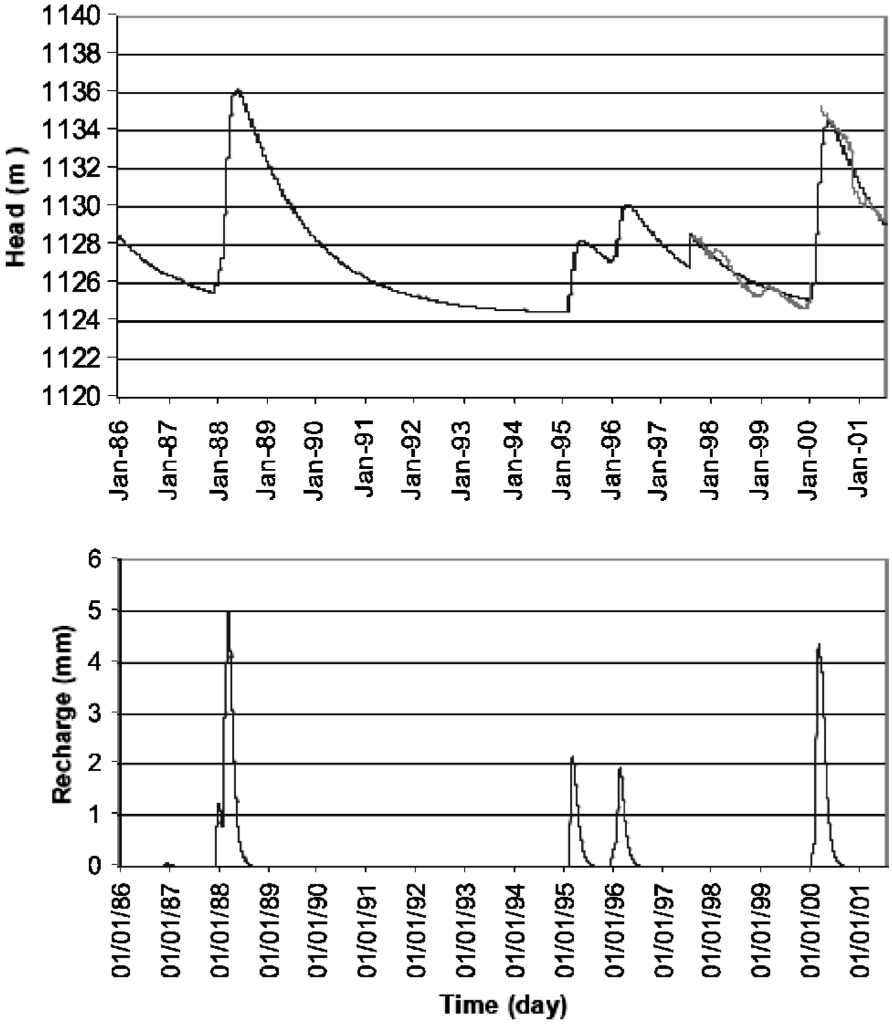

Figure 2. Example of recharge simulation using 1-D EARTH model (van der Lee and Gehrels, 1990) from (Lubczynski and Obakeng, 2006); the upper graph shows head calibration while the lower the corresponding model solution.

The availability of groundwater resources is constrained by the wet years that replenish aquifers such as the four hydrological years in Figure 2. The unsolved problem in groundwater management and sustainable planning is that as yet rainfall cannot be reliably predicted so the recharge as well. This is why in countries fully dependent on groundwater resources, such as Botswana, mining of groundwater storage reserves, is still common and of strategic importance in case of sequential dry years. Relatively recent overview of recharge advances, also in arid and semi-arid areas, can be found in the special issue of the Hydrogeology Journal vol. 10 no. 1 from 2002.

ET_g is groundwater evapotranspiration as discussed below.

3. Groundwater Evapotranspiration

Groundwater evapotranspiration (ET_g) is the amount of water that is lost from an aquifer or its capillary fringe by plant root water uptake (T_g) and by direct evaporation from aquifer's groundwater table (E_g).

3.1. IMPORTANCE OF GROUNDWATER EVAPOTRANSPIRATION

ET_g is typically underestimated (Lubczynski, 2000) despite it can be a significant component of groundwater balance (Eq.1) particularly in arid and semi-arid environments because it decreases the effective recharge known as the net recharge (R_n) where:

$$Rn = R - ET_g \tag{2}$$

In groundwater resources evaluations, management and planning, of critical importance is the net groundwater recharge flux (R_n) representing net water input to an aquifer. Unfortunately, it is a common practice to use recharge without specifying whether the intention was to use R_n or R. Also the recharge evaluation methods do not specify whether the resultant output provide R_n or R. Use of R instead of R_n (or opposite) while disregarding ET_g, for example in groundwater modeling, leads to substantial errors that can mislead groundwater resources evaluation particularly in arid and semi-arid environments where ET_g is usually significant so the difference between R_n and R as well.

3.2. GROUNDWATER EVAPOTRANSPIRATION AS PART OF TOTAL EVAPOTRANSPIRATION

The groundwater evapotranspiration (ET_g) is a groundwater component of total evapotranspiration (ET) as shown in Equation 1.

$$ET = ET_s + ET_u + ET_g = ET_s + \left(E_u + T_u\right) + \left(E_g + T_g\right) \quad (3)$$

$$ET_u = E_u + T_u \quad (4)$$

$$ET_g = E_g + T_g \quad (5)$$

where ET_s is the surface evapotranspiration (direct evaporation from terrain surface, evaporation from water bodies etc.); ET_u is the unsaturated zone evapotranspiration; ET_g is the groundwater evapotranspiration; T_u is the unsaturated zone transpiration - transpiration by plant root water uptake from unsaturated zone (excluding capillary fringe); E_u is the unsaturated zone evaporation - evaporation by upward water movement originated from soil moisture of an unsaturated zone (excluding capillary fringe); T_g is the groundwater transpiration - transpiration by plant root water uptake from an aquifer and/or its capillary fringe; E_g is the groundwater evaporation - evaporation by upward water movement originated from an aquifer and/or its capillary fringe. All the components of the Eq. 3 are illustrated in Figure 3.

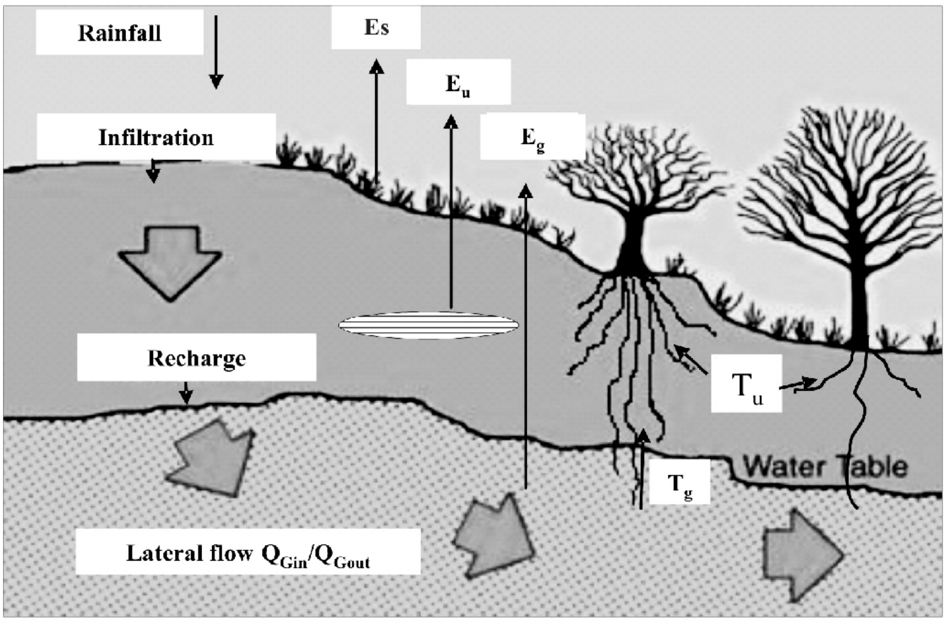

Figure 3. Schematic diagram showing components of total evapotranspiration and other water fluxes.

The ET_g was first investigated and defined using tank experiments and groundwater table measurements by White (1932) in the Escalante Valley (Utah) and by Robinson (1970) in Humbold River Valley (Nevada). The relatively recent development of micrometeorology, allowed Nichols (1994) to estimate dry season ET_g for the shrubs of the Great Basin Desert (Nevada) using correlation between ET defined by Bowen ratio method and corresponding groundwater table depth in the dry season when $ET_s = 0$. However no guidelines were found in this study on how generic the proposed correlation formula was and on how good was the assumption neglecting T_u in the dry season and T_g in the other months than those specified as dry (mid-July to early September). The validity of such assumption in arid and semi-arid areas is however of critical importance, as confirmed in the Serowe study case on the Kalahari (Botswana), where even with a deeper groundwater table than 10 m b.g.s., the measured dry season tree transpiration consisted not only of the root groundwater uptake (T_g) but also of T_u (Lubczynski, 2000). The presence of ET_u ($ET_u = T_u + E_u$) even in the peak dry season, forbade the use of the convenient groundwater modeling assumption that dry season $ET = ET_g$. Therefore, in this approach, instead of direct determination of ET_g, the spatially variable dry season ET ($ET = ET_u + ET_g$) obtained from the remote sensing (RS) solution of energy balance (Bastiaansen et al., 1998, Timmermans and Meijerink, 2000) was scaled down in the numerical model calibration (Lubczynski, 2000). This procedure was further improved in another case study in Spain (Lubczynski and Gurwin, 2005), where the remote sensing solution of energy balance was scaled down using sap flow measurements and finally the ET_g distribution was calibrated in the fully transient model i.e. with spatio-temporally variable fluxes.

3.3. COMPONENTS OF GROUNDWATER EVAPOTRANSPIRATION

As mentioned above, groundwater evapotranspiration (ET_g) consists of groundwater evaporation (E_g) and groundwater transpiration (T_g).

3.3.1. *Groundwater evaporation*

Until recently, based on the experience from hydrogeological studies in moderate climates such as in Europe and North America it was used to be assumed that if groundwater was deeper than few meters, then $E_g = 0$. This understanding has recently changed. E_g is important, particularly in arid and semi arid environments characterized by large diurnal and seasonal temperature differences. In such climates there are substantial soil temperature gradients with depth influencing also the presence of large water potential gradient. The two are the main driving forces of the vertical water movement. Recent research by Walwoorth (2002a, b) and Scanlon et al. (2003) present modeling studies (using HYDRUS code) of groundwater evaporation from >20 m b.g.s. in vapor and liquid form, driven by water potential and the soil temperature gradients. According to Coudrain-Ribstein et al. (2003), the vapor movement is dominated by thermal gradient and it condenses in the shallow subsurface of a few meters depth below the ground surface depending on the soil type. The condensed as well as the capillary-driven water evaporates then from the shallow subsurface to the atmosphere.

Despite very valuable attempts made, the process of groundwater evaporation from large depth is still not well understood; this is mainly because:

- in unsaturated zone the water moves up in liquid and vapor form; the partitioning of the two forms in different soil types and at different depth below the surface is not well understood yet;
- the partitioning between evaporation originated from groundwater (E_g) and the other originated from unsaturated zone (E_u) is not defined yet;
- the vapor form of water movement cannot be directly measured, for example by sensors; it can only be deduced indirectly for example from profile temperature and matric potential measurements;
- the interactions between soil/rock, water and plant root systems at large depth (of several tenth of meters) are not well understood yet;

3.3.2. *Groundwater transpiration*

Water uptake from subsurface is a process that can be estimated either at the canopy level using for example two source RS based energy balance models (Kustas and Norman, 1999; French et al., 2000) or at the stem level by using sap flow measurement techniques (Smith and Allen, 1996; Kostner et al., 1998) combined eventually with remote sensing when spatial aspect of transpiration is required. In arid and semi-arid conditions where subsurface fluxes are usually very low so the high accuracy needed, and where energy balance methods experienced problem of overestimating evapotranspiration, the stem based methods are preferred. This is because they provide direct, stem-restricted measurement of the overall water uptake from subsurface.

Transpiration measurement is complicated because of the complexity of that process and its diversity with regard to species type and subsurface soil condition of root water uptake. These complications are particularly common in arid and semi-arid conditions and are related mainly to:

- Large rooting depth of some species (more than 60m depth) like for example *Boscia albitrunca* or *Acacia erioloba* (Canadell et al., 1996; Le Maitre et al., 2000) that can reach groundwater table;
- Hydralic lift (Horton and Hart, 1998), which is a process by which some deep-rooted plants take up groundwater from large depth and exude it into the lower depth root system;
- Phenomenon's such as reverse direction of sap flow or presence of sap flow in the night that are characteristic for some tree species in arid and semi-arid areas; these phenomenon's complicate measurements, particularly when using methods such as constant temperature heat balance, compensation heat pulse and thermal dissipation methods (Burgess et al., 2000);
- Natural thermal gradient which can be relevant in some of the tree species measurements (Do and Rocheteau, 2002a, 2002b); while using thermal based methods the natural thermal gradient is considered as a measurement noise so it has to be removed.

The above mentioned sap flow measurement complications, if present however, typically result in less than few percents of error that unlikely would exceed 20%.

The sap flow measurement can provide species-specific, temporal pattern of transpiration. It can also provide a rough estimate of T_g but only if T_u can be neglected. If not, the partitioning between T_g and T_u is required but it is a very difficult and unsolved matter as yet. The approximate solutions to this problem using various combinations of methods were presented by Thornburn et al. (1993), Cook et al. (1998) and Lubczynski and Gurwin (2005).

4. Discussion and Conclusions

Specific environments, like areas characterized by arid and semi-arid climates, enforce different hydrological regimes than moderate climates such as in Europe or North America and therefore also different solutions of groundwater balances (Eq.1). The largest differences are in the recharge and groundwater evapotranspiration fluxes.

Recharge in arid and semi-arid countries is affected by characteristic for this climate, short, intensive and spatio-temporally variable rainfall events. Because of that, recharge is by far more spatio-temporally variable than in moderate climates. Concerning temporal recharge pattern, of particular importance is the large spatio-temporal variability and large temporal concentration of intensive rainfall events, typically restricted to 1-3 month period only. Such events if sufficiently productive result in recharge that is very different from one year to another. In extreme cases like in Botswana Kalahari environment it results in the erratic recharge, occurring only once per several years (Figure 2). With such temporally concentrated distribution of recharge, only fully transient models with spatio-temporally variable input fluxes can provide reliable model calibration that can further be used in groundwater management (Lubczynski, 2005).

Groundwater evapotranspiration is usually underestimated. This is probably because in moderate climates of Europe and North America from where most of the hydrological research originates, the ET_g is not always critical. However in arid and semi-arid countries ET_g represents a significant component of groundwater balance. The ET_g can occur in two different ways, either as groundwater evaporation (E_g) or as groundwater transpiration (T_g). Large seasonal water deficits and large diurnal and seasonal temperature differences in arid and semi-arid environments result in large temperature and tension pressure gradients resulting in significant E_g. The rates and depth restricting such process, are not well defined yet despite number of significant scientific contributions already made (Wallvoorth et al., 2002a, 2002b; Scanlon et al., 2003, Coudrain-Ribstein et al., 2003; Lubczynski and Gurwin, 2005). Moreover, since the process of upward water movement is not well understood and cannot be directly measured, the partitioning between E_g and E_u is not feasible yet. In contrast, the transpiration (T_g+T_u) can be directly measured at the stem or even at the shallow root level. The most widely used thermal-based sap flow measurement methods, involve number of complications resulting in various measurement errors which however can be avoided or at least reduced to insignificant if the measurements are carried out properly. The advantage of such measurements is that by using them, species-specific transpiration patterns (spatial and temporal) can be determined. Unfortunately however, like in case of E_g, the partitioning of T_g from T_u is not developed yet, despite some significant contributions have already been made (Thornburn et al., 1993, Cook et al., 1998 and Lubczynski and Gurwin, 2005).

More research is needed to better define ET_g particularly in arid and semi-arid environments. Neglecting significant ET_g, leads to erroneous groundwater flux balances (also in groundwater modeling calibrations) and in consequence to groundwater mismanagement.

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WATER MANAGEMENT IN THESSALY, CENTRAL GREECE

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Abstract: Infrequent reports on the water reserves of Greece paint a very bleak picture of the future, especially on the Thessaly plain which has reached unnerving proportions. For decades, the work of draining lakes went on in Thessaly without moderation. Thus, lakes such as Nezeros, Xynias, Nessonida, Karla and many marshy areas were drained with the double excuse of obtaining more land for farmers but also of getting rid of malaria. These immoderate interventions caused the disappearance of large areas of surface water which would have been useful, not only for the irrigation of large areas but also for the enrichment of the subterranean water level. As a result, where boring used to find water at 30 metres, it is now necessary to get to a great depth. Especially as far as the draining of Karla is concerned, an area of 8,500 to 10,000 hectares of complete drainage was decided. Recent data on water concluded that 86 percent of Greece's total water consumption is used for irrigation purposes, while the area of Thessaly alone accounts for 21.7 percent of national consumption. The greater Acheloos diversion scheme is a large and controversial project intended to provide irrigation water for between 240,000ha and 380,000ha of farm land in the plain of Thessaly. Farmers' excessive consumption of water is evident in the Thessaly region and cotton production absorbs over one-fifth of the country's water. Mismanagement has also affected the quality of fresh water, the pollution of resources by pesticides and their residues, the intrusion of seawater into coastal aquifers, and the gradual desertification of land.

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Keywords: Acheloos river; groundwater; Lake Karla; Thessaly; water reserves.

1. Introduction

1.1. WATER SUPPLY IN THE REGION OF THESSALY

The plain of Thessaly covers an area of 13,377 km^2 that occupies the central section of mainland Greece. It is surrounded by high mountain ranges (Pindus, Olympus, Pelion, Othrys, Ossa and Agrapha), encircling a low plain. Thessaly borders Macedonia to the north, Sterea Ellada to the south, Epirus to the west, and its eastern shoreline lies on the Aegean Sea. It has the highest percentage of flat land in Greece.

Thessaly has about 800,000 inhabitants. The biggest cities in the area are Larissa and Volos (total population 300,000). The main economic activities are agriculture, industry and tourism. The region of Thessaly produces 6.3% of the GNP, while the per capita product is 13,000 €. The unemployment rate in the region is 12.2%. Total annual water consumption is 1,171 hm^3, consisting of 65 hm^3 for domestic use, 1,060 hm^3 for agricultural use and 46 hm^3 for industrial use. The consumption index is estimated at 38% and the population to water resources index is equal to 204. The exploitation index is 31%.

Water shortage problems are frequent during the irrigation period, while in the winter floods occur in large areas. The coastal zone is a favourite destination for many tourists during the summer, increasing water supply requirements during the tourist period.

The drainage basin of Pinios River is 9,500 km^2 and the main tributaries are the rivers Titarisios, Enipeas, Kalentzis, Litheos and Asmaki. The Region of Thessaly also has two more water basins: the drainage basin of (ex) Lake Karla (1,050 km^2), rising at the eastern side of the region, and Lake Plastira at the western side (Table 1). Lake Plastira is a part of the waterhed area of Achelloos River which belongs to the West Sterea Ellada water region. Lake Plastira, with a storage capacity of 400 hm^3, is regulated for hydropower production. The installed hydropower capacity is 141 MW, and the power plant produces a total of 250 GWh per year.

Table 1. Surface of the drainage basins in Thessaly

Drainage Basin	Surface (km^2)
Pinios River	9,500
Lake Karla	1,050
Other Basins	2,812
Total	13,362

Annual "needs" in water according to existing composition of cultures in Thessaly are about 1.8 billion cubic meters of water, of which 136 million m^3 are for house hold water supply, 100 million m^3 for environmental maintenance and 1.6 billion m^3 for irrigation.

The real annual uptake of irrigatory water today, with defective irrigation of 2.600.000 ha in Thessaly, is 750 millions m^3 of water, of which 200 million m^2 are surface water and 550 million m^2 underground water (Table 2). From the total amount of water used, 26% (200 millions m^3) comes from surface water (Lake Plastira, Pinios river, Minor dams) and 74% (550 millions m^3) comes from roughly 30,000 drillings (1/3 of which were collective and 2/3 private).

Table 2. Water demand versus real situation in Thessaly

Water Demand in Thessaly	1.2 billion m^3
	136 million m^3 household needs 100 million m^3 environmental needs 1600 million m^3 irrigation
Real Situation	750 million m^3
	200 million m^3 surface water (lake Plastiras, Pinios river etc) 500 million m^3 underground water (30 000 drillings) (400 million m^3 recharge + 100 million m^3 subtract)

By analyzing the piesometric data from the Ministry of Agriculture (1974), it is observed that the level of underground water have a systematic fall (Figure 1) mainly in the Central and South-eastern part of Thessaly. Same analysis between the years 1984-1996, shows that the total of water that was removed from underground water of the whole Thessalian plain was of the order of 1000 million m^3 of water. This caused the fall of the water table from 0-11 meters in the Prefecture of Karditsa, 1-11 meters in the Prefecture of Trikala, 5-25 meters in the Prefecture of Larissa, 10-15 meters in the Prefecture of Magnesia (Figure 2). Measurements during the years 1993-1994, showed that 550 millions m^3 of underground water are drawn annually, i.e. more than 100 millions m^3 of underground water without renewal.

1.2. IRRIGATION FOR COTTON CULTIVATION

Although Thessaly is a region that is naturally rich in water, historic mismanagement of its resources, coupled with the widespread cultivation of cotton - a crop with a particularly high water demand, have led to serious supply problems. Unregulated bore drilling for irrigation has caused depletion and increased salinity of the groundwater, a situation further exacerbated by the wasteful irrigation methods used.

Farmers' excessive consumption of water is evident in the Thessaly region and the public was stunned by reports that the farmers of the Thessaly plain break EU production quotas (no more than 1,131,000 tons/year is the EU Quota) on a product that is subsidized at four times its price (Table 3).

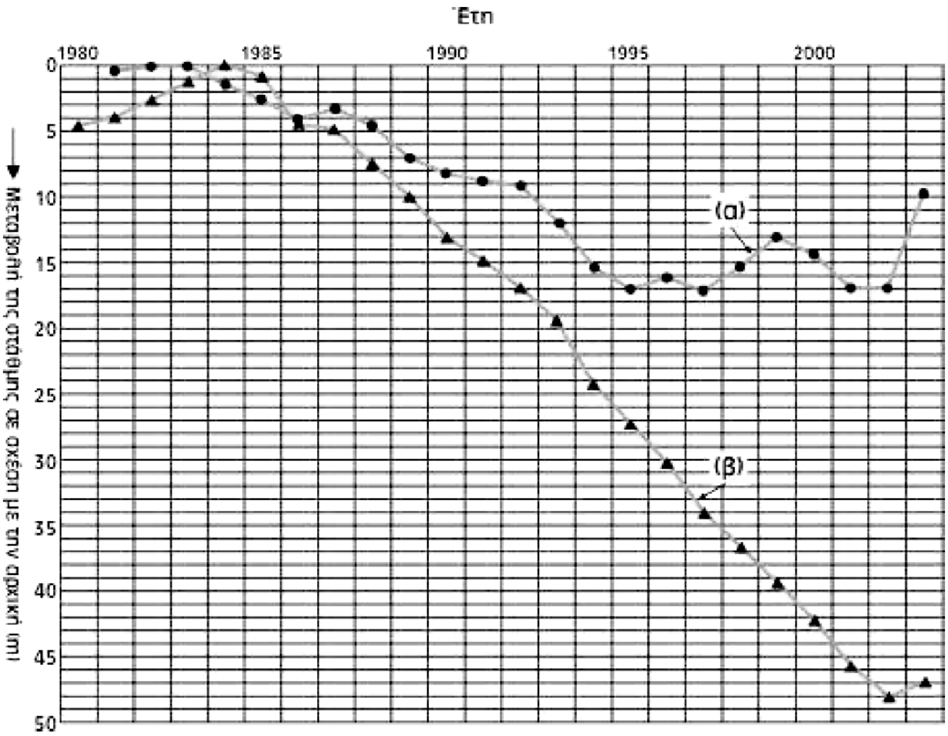

Figure 1. Variations of the water table depth (in meters) through time.

Table 3. Production of cotton in Greece

Year	Area (hectars)	Production (tons/year)
1961	208,300	277,000
1971	130,200	330,000
1981	126,300	358,000
1991	233,000	680,000
2001	378,700	1,246,000

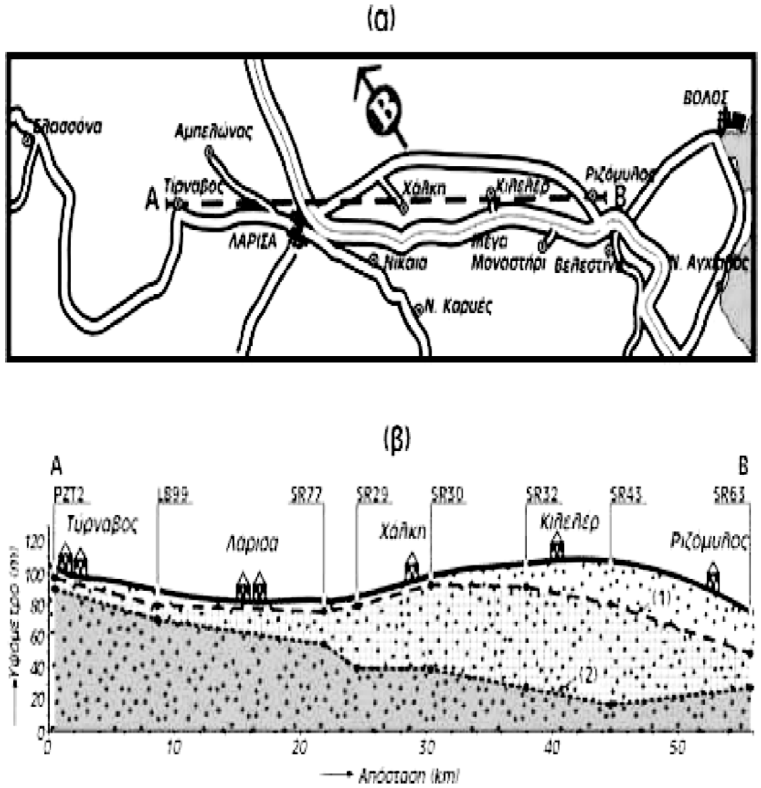

Figure 2. Section AB in Eastern Thessaly, showing the water table depth in (1) 1973 and (2) 2003 (Euagelopoulos, A., 2004).

2. Acheloos River Diversion Scheme, Greece

The greater Acheloos diversion scheme is a large and controversial project intended to provide irrigation water for between 250,000 ha and 400,000 ha of farm land in Thessaly. The scheme involves the construction of a major diversion channel, two tunnels, a water intake system, sluice gates and surge shafts together with additional service tunnels and access roads. In addition, a series of large dams are also to be incorporated for hydroelectric generation.

Acheloos River flows from the Pindos Mountains westwards to the Ionian Sea. The plan to divert the river eastwards to irrigate the growing crops, was first conceived in the 1930s, but lack of funding halted its implementation. Although a number of dams were built along Acheloos in the intervening decades the diversion project itself remained stalled until 1984, when the government expressed its renewed intention to proceed.

The final plan devised for the Acheloos diversion project, includes the construction of major dams and associated reservoirs at Mesochora, Sykia, Mouzaki and Pyli, together with a diversion channel to Thessaly, the Mesochora-Glystra tunnel and the long Pyli-Mouzaki tunnel. This system of dams, reservoirs and tunnels is designed to supply an estimated 600 million m^3 of water from the Acheloos rivers to Thessaly. However, a number of things have changed since the diversion scheme had first been envisaged. Moreover, greater environmental awareness made the whole issue far more contentious than half a century earlier.

3. The Story of Lake Karla

Up to 1962, a big lake existed at the southeast part of the Thessaly plain. It was Lake Karla (the Voiviis Lake of ancient Greeks), with a maximum size of 20 km^2. The lake gave a production of about 1,000 tons of fish annually. It was also a very important wetland on a European level with more than 1,000,000 waterfowl hibernating in the lake's greater area. In 1962, mainly for agricultural purposes, Lake Karla was drained through an ambitious reclamation project.

Following its drainage, a part of the old lake-bed was cultivated, climate changed for the worse, the greater area's hydrological regime was severely disturbed with groundwater remarkably lowering, big cracks occurred at the soil and chemical degradation of the soil was detected. Waterfowl almost disappeared and after the completion of the land reclamation project, it became evident that it was a failure. Currently, a project for the partial restoration of the Karla Lake is under progress and a water reservoir with a size of 3,800 ha will be created. Environmental impact assessments have finished and the project will be completed in three years.

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MODELING OF HEAVY METAL CONTAMINATION WITHIN AN IRRIGATED AREA

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Abstract: Leaching of exposed rocks associated with the Madneuli complex ore deposit and direct discharge of mine waters to nearby watercourses are attributed to the heavy metal contamination of ground and surface water. Human-health issues may exist for two reasons. First, contaminated surface water, such as the Kazretula, Poladauri and Mashavera Rivers, is diverted to agricultural fields through a network of irrigation channels. Second, the Bolnisi Kvemo and Bolnisi villages derive their water supply from wells drilled into alluvial deposits of the Poladauri River. Field and laboratory investigations were carried out to understand the water quality of this area. Geophysical prospecting by electrical and seismic methods was used in the Poladauri valley to establish the intensity and direction of ground water flow. The results of sampling and analysis of surface and ground water, soil, and vegetables were used to assess the extent and character of metal contamination. Using the GIS-technology, thematic maps were used to create 3-dimensional models to

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visualize the geological-hydrogeological and geochemical structure of the region and identify likely pathways of contamination. Geochemical and transport modeling indicates that the present-day maximum contaminant levels will eventually reach the total investigation area posing health risks to the local population.

Keywords: heavy metal, pollution, modeling

1. Introduction

Acid mine drainage from waste rock piles (150 million m^3) and sulfide ore tailings of the Madneuli Cu-Au open-pit mine resulted in an environmental contamination problem in Bolnisi mining district, Georgia (Figure 1). Intensive leaching of exposed rocks and direct discharge of mine waters to nearby watercourses led to heavy metal contamination of groundwater and surface waters, such as the Kazretula, Poladauri, and Mashavera Rivers.

Of these three contaminated rivers, the concentrations are greatest in the Kazretula and Poladauri Rivers, probably because they receive discharge water directly from the quarry. Whereas the concentration of toxic metals commonly exceed the maximum permissible concentration (MPC) by 50 to 100 times, one inlet had Cd concentrations (3.8 mg/l) that were about 2,000 times larger than the ambient groundwater concentration (0.002 mg/l). In the area adjoining the quarry and mill, 18 chemical elements were present in concentrations that exceeded the existing MPC Georgian norms. The analyses of surface water almost everywhere in this region indicated that concentrations of Cu, Zn, Pb, Ni, Mn, Cr, Ti, Mg, Cd, Hg exceeded ambient concentrations.

There is intensive agricultural activity in the study area including gardens, cornfields, and vineyards. The study region is the main source of vegetables and wine production by the Bolnisi winery is significant. The breeding of cattle is also intensively developed in this area. In all these agricultural activities, highly developed irrigation systems are used that derive water using one of two approaches. One approach is to divert surface water, such as the Kazretula, Poladauri and Mashavera Rivers, to agricultural fields by the network of irrigation channels. In the second approach, water is produced from wells drilled into alluvial deposits of River Poladauri. In addition to agricultural use, the Bolnisi Kvemo and Bolnisi villages use water from these wells to supply drinking water to people whose population was about 56,651 in 2005.

2. Data Analysis

The investigation of water-resource quality in the study area was conducted over the period 2000-2002, and as part of the TOXICAL project (4); this work was then continued over the period 2002-2004 as part of the ENRISK project (5). Findings from these studies revealed that contamination of water and soil exceeds, by many times, the above mentioned MPC norms. This finding is particularly true in the Poladauri River valley which is the main source of vegetables for Tbilisi.

2.1. GEOPHYSICAL PROSPECTING

Quantitative modeling of chemical transport requires detailed knowledge of geological structure of the area, as well as the principle directions of groundwater flow. To determine the directions of groundwater flow, several geophysical surveys were carried out using the self-potential electrical prospecting method along the valley between the slope and river. Using the self-potential electrical prospecting method, the northeastern direction of underground water flow was determined (Figure 1).

In the Poladauri River valley, the piezometric head varies according to the site elevation; for example, on the slopes of the valley the head is about 17 m, in the central part of valley (village Lower Bolnisi or Kapanakhchi) the head is about 12 m, and close to the river bed the head is between 1-2 m (1). The above distribution is typical and reflects the general pattern of groundwater flow.

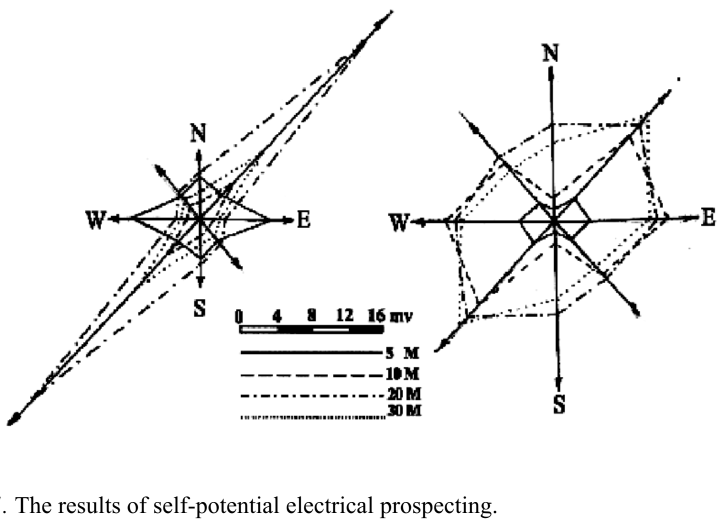

Figure 1. The results of self-potential electrical prospecting.

Electrical prospecting provides information about the water levels in quaternary sediments, as well as the dominant (North-East) direction of the groundwater flow in the Poladauri River basin. Following the identification of areas and depths of water-bearing strata, several slug-tests were performed. Based on findings from the slug tests, the quaternary sediments were determined to have comparatively low hydraulic conductivity and high absorption capacity resulting in the accumulation of contaminants in sediments.

2.2. CHEMICAL DATA

Analysis of solid (rock, soil, vegetable) and liquid phase (surface and ground water) chemistry was carried out to assess the extent and character of contamination by metals. At selected sampling sites, the analyses were repeated in regular time intervals (Figure 2).

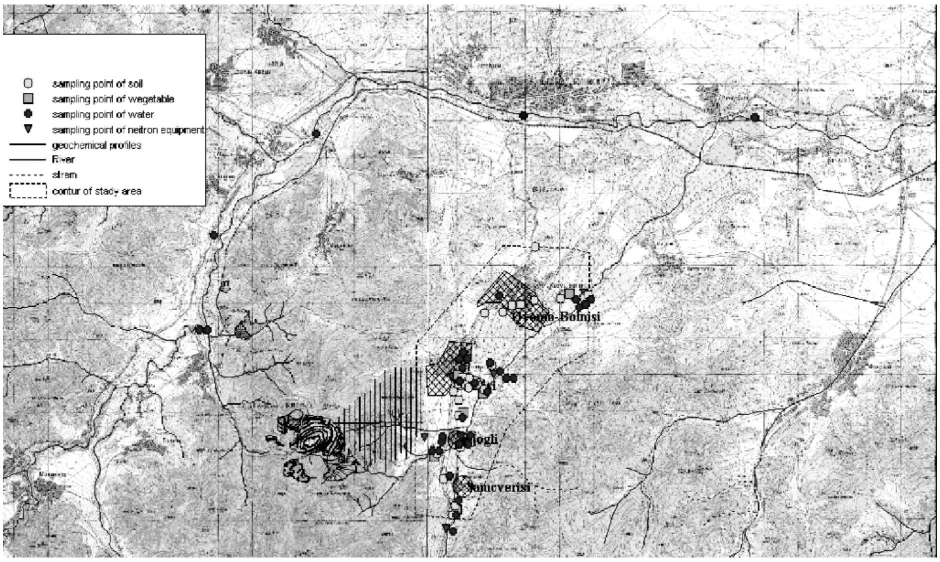

Figure 2. Schematic of sample locations.

On the East flank of the Madneuli gold-copper-polymetal deposits, geologists carried out a geochemical investigation on the secondary aureole scattering. The samples were selected from tailings subsoil horizon B. The analyses of 50 elements were conducted at a laboratory in Vancouver, Canada. The ultra-trace level method used conventional ICP-AES analysis combined with ICP-MS (mass spectroscopy) to provide the best possible geochemical detection limits. Major rock forming elements and more resistive metals are only partially dissolved. Concentrations of heavy metals in parts per million (ppm) remained high in rocks. For example, the maximum concentration of Cd in rock is 2 ppm, while its MPC in rock is 0.5 ppm. Therefore, for Pb-263 and 10; Sr-260 and 10; Zn-750 and 50; Cu-630 and 20. It approximately exceeds the MPC by 4-30 times. These findings confirmed the hypothesis that leaching of waste pile by groundwater was the principle source for contamination to the Poladauri River. Accumulation of toxic metals in the Poladaui River, over 25 years from surface leaching, resulted in respective sediment concentrations of Cu and Zn at levels as high as 2,100 ppm and 10,000 ppm (Figure 3).

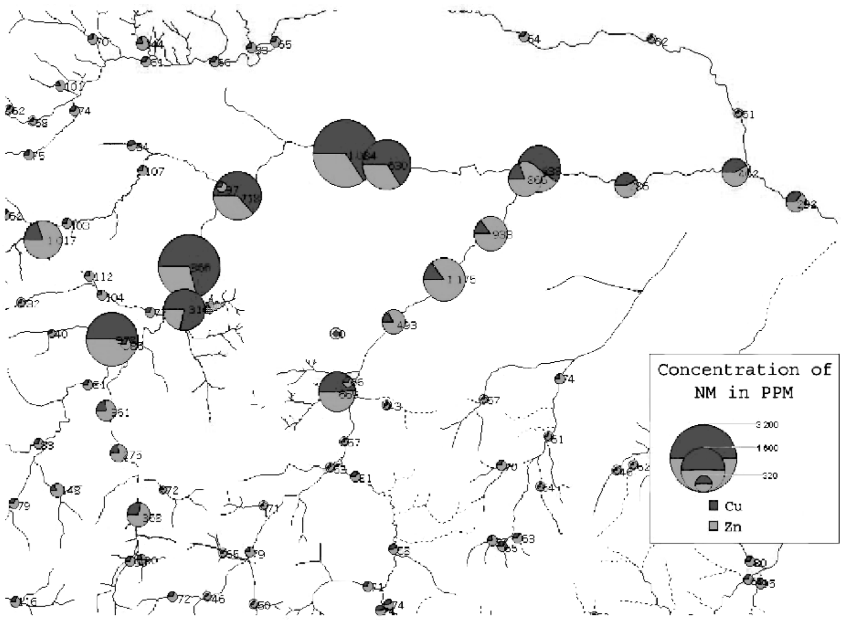

Figure 3. The concentration of heavy metals in river sediment.

Monitoring of the study region included sampling of the soil and surface and ground water at specific sites and at regularly intervals. Whereas the soil was sampled only once, the respective sample frequency for surface and ground water was every 3-4 months. All samples were analysed using standard chemical analysis techniques and protocols for heavy metal (HM) content and concentration of other components (4, 5). The reliability of analyses was controlled by analysis of the same samples using various atomic-absorption spectrometers and independent chromatographic analyses. A summary of typical results for 30 ground water parameters determined at sampling sites 7-14 are presented in Table 1.

Table 1. Summary of various parameters in ground water

Hydrochemical	Unit	MPC	Sampling sites							
parameters			7	8	9	10	11	12	13	14
pH		6.5-8.5	3.92	5.45	7.08		2.87	5.15	6.14	6.97
Cl	mg/L	350	17.8	19.9	10.0		16.8	12.0	447	11.9
SO_4	mg/L	250-500	1070	1070	24.0		2460	416	1150	336
HCO_3	mg/L	400	0	439	146		0	0	3050	122
Na	mg/L	120-175	207	70	10.2		531	78.8	1152	65.5
K	mg/L	12	4.7	6	1.0		6.8	2.6	14.0	1.6
Mg	mg/L	50	48.8	95.2	13.4		64.7	25.6	165	35.4
Ca	mg/L	100	120	424	40.0		130	60.0	6358	70.0
NH_4	mg/L	0,5	0.60	0.50	<0.05		0.50	0.05	0.0	0.05
NO_2	mg/L	0,1	0.00	0.00	<0.01		0.00	0.015	<0.01	0.010
NO_3	mg/L	40-50	0.8	0.3	0.7		5.7	1.2	0.3	1.0
PO_4	mg/L	3,5	0.00	0.01	<0.01		0.00	0.01	0.02	0.01
Fe	mg/L	0,2-0,5	240	560	57.6	76.8	528	160	2.80	0.2
Cu	mg/L	0,01	84	1.68	2	2.72	39.5	66	0.10	0.54
Pb	mg/L	0,01	0.88	0.58	0.28	0.3	0.52	0.6	0.00	0.03
Zn	mg/L	0.01-0.1	5.12	<0.002	0.15	2.48	5.68	6.08	<0.002	3
As	mg/L	0,03	0.021	0.021	0.010		0.010	0.015	0.010	0.03
Mn	mg/L	0,01	0.7	0.15	0.31	0.34	1.08	0.16		0.1
Ni	mg/L	0,1	0.16	0.2	0.18	0.6	0.22	0.22		
Co	mg/L	1	1.2	1.28	1.28	2.48	0.8	0.72		0.72
Li	mg/L		0.02	0.04	0.04	0.06	0.03	0.03		
Cd	mg/L	0,005	0.54			0.16				
F	mg/L	1,5	0.13	0.04	0.13		0.25	0.12	0.35	0.12
Water hardness	mg-eq/l	7	11.6	30.3	3.1		14.1	5.6	35.8	6.5
Turbidity	mg/L	2	1670	0	10		3036	241	0	323
Dry remainder	mg/L	1000	1520	2204	268		3310	640	6380	662

We conclude that the contamination of the region remains high despite some efforts to clean quarry waters. We carried out analyses of data on water contamination using GIS, which revealed that the content of toxic components does not decrease significantly along its path from the source to the river, then from the river to fields (Figure 4).

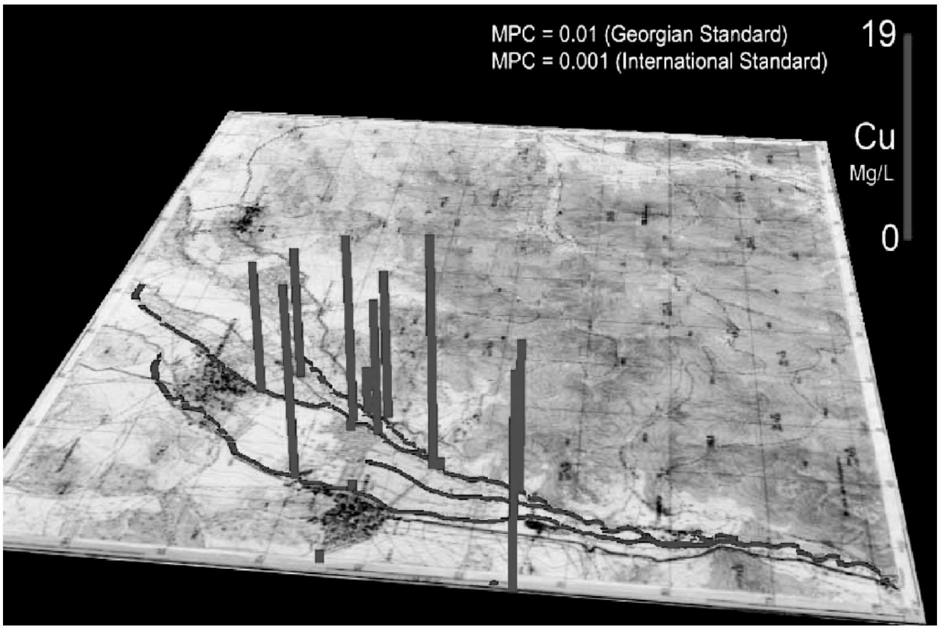

Figure 4. The concentration of Cu in the brooks flowing from the quarry.

High content of these elements in the alluvial deposits are attributed to their high adsorption activity. Their content in agricultural products is also high. That means that contamination comes from the whole ore-bearing formations even intact ones.

Table 2. The contents of heavy metals in cucumber

	Unit	MPC	N 82	N 83	N 84	N 85	N 86	N 87
Na	mg/L	120-175	3.1	1.3	2.3	1.3	4.8	2.5
K	mg/L	12	2.8	0.1	0.4	2.8	5.5	6.3
Mg	mg/L	50	11.3	11.3	54	7.5	16.3	15
Ca	mg/L	100	70	45	65	60	97.5	32.5
Cu	mg/L	5	0.2	0	3.6	0	1.4	1.9
Pb	mg/L	0.5	1.4	0.7	0.9	2	0.6	0.4
Zn	mg/L	10	9.8	0	32	0.2	17	9.4
Ni	mg/L	0.1	0.15	0	0.2	0	0.6	0.35
Co	mg/L	1	0.1	0	0.5	0.05	0.3	0.24
Cd	mg/L	0.03	0.34	0.2	0.3	0.18	0.2	0.05

To the North from v. Tsitelisopeli till v. Rachis-Ubani, the contents of HM increase. To understand the distribution of metal contamination, which after some decay begins to increase in the areas that are far from the quarry-polluted surface water sources. This finding is attributed to the complex geological structure of the ore deposit. Specifically, there is additional leaching of HM from ores in the region of v. Citeli-Sofeli. The situation is illustrated by the section (Figure 5).

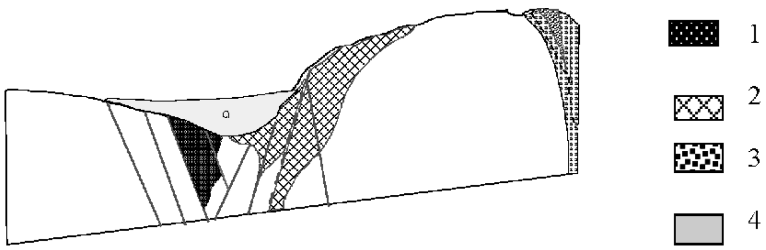

Figure 5. The geological structure of ore deposits in the v.Citeli-Sofeli. 1-Ore deposit; 2- Magmatic body; 3- Volcanic apparatus; 4 - Quaternary alluvial deposits.

Geochemical and contaminant transport modeling was used to estimate the direction, rate and extent of chemical migration from the contaminated site (Figure 6).

The flow-modeling results are used as an input of the transport model. In turn, this information was used to support environmental management and decision-making involving identification of high-risk areas, protection from pollutants, and planning of remediation work. Results of the modeling suggest that the present-day maximum contamination levels will eventually reach the total investigation area posing long-term human-health risks to the local population. Mineral source estimates indicate that the contamination may continue to be mobilized and transported for centuries with no expectation of natural attenuation (Figure 7).

The potential effects of implementing preventive actions was studied by preparing a model scenario in which the present Cd contamination level of from 1 mg/l was lowered to 0.1 mg/l in two streams entering the model area (Figure 8). The results suggest that using remediation, a substantial reduction of contamination can be achieved in relatively short period. Concentration of Cd, and therefore the induced human-health risks, could be lowered close to background levels. Therefore, the adverse effects on human health could be mitigated by redirecting the extraction of drinking water in the Bolnisi mining region to areas located at a sufficient distance from the contaminated stream.

High investment in preventive actions will likely become cheaper than remediation of contaminated groundwater. Without preventive remediation, the adverse effects on human health can be expected worsen (2).

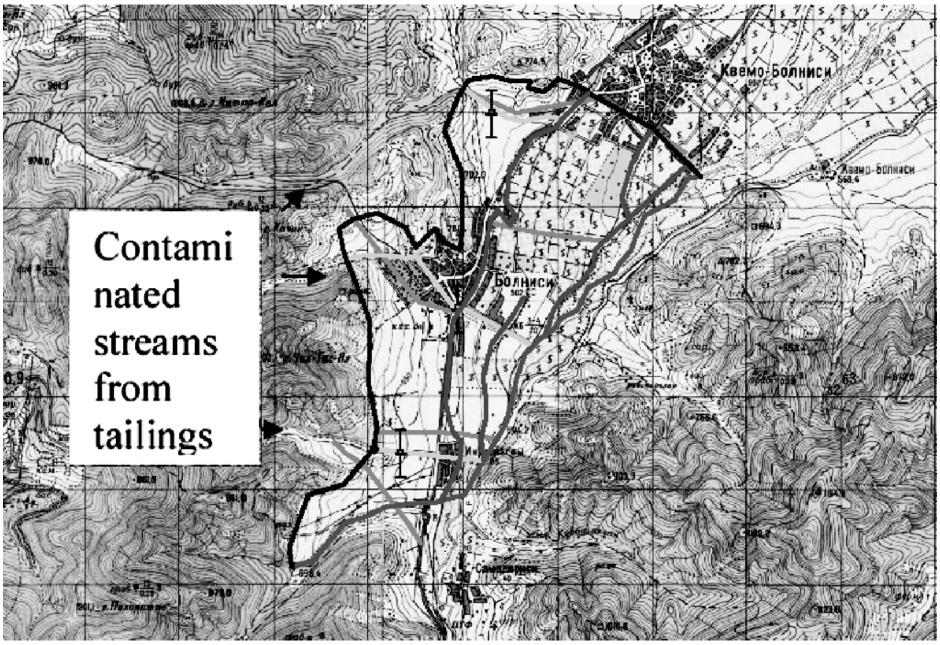

Figure 6. Investigation site. The orange lines represent the geo-electric profiles. The red and the black line indicate the model boundaries where the first and the second type of boundary conditions (specified head or flux) were assumed to prevail, respectively. The light blue lines represent the tributary strems of the Kapanachi River entering the model area. Blue lines indicate manmade irrigation canals.

3. Conclusions

We conclude that contamination by heavy metals in the region will likely remain high posing human-health concerns. For this reason, alternative remediation strategies should be implemented in an effort to mitigate contamination and reduce the likelihood for human-health concerns. Alternative remediation strategies could include methods such as mechanical (sedimentation in pools), biological (bacterial) and chemical (zeolits, dolomites, chalk) cleaning of surface and ground water.

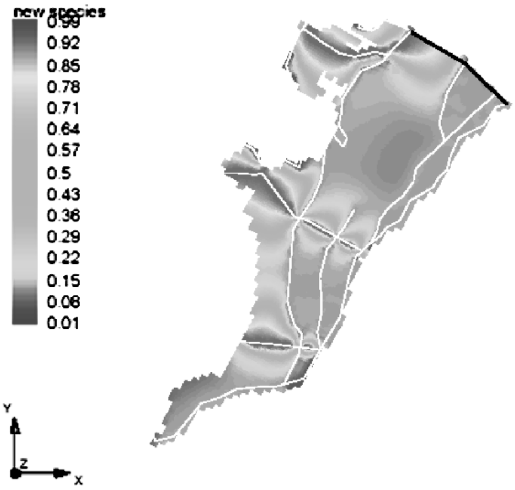

Figure 7. Examples of modeling results obtained by assuming constant Cd concentration of 1 mg/l in the streams. 0.1 mg/l in the channels and 0.005 mg/l for the background and initial concentrations. Dispersivity is assumed to be 200 m. Sorption on Cd on clay and ferrous hydroxides has been approximated to follow linear isotherm with soil-water partitioning coefficient of 4 ml/g.

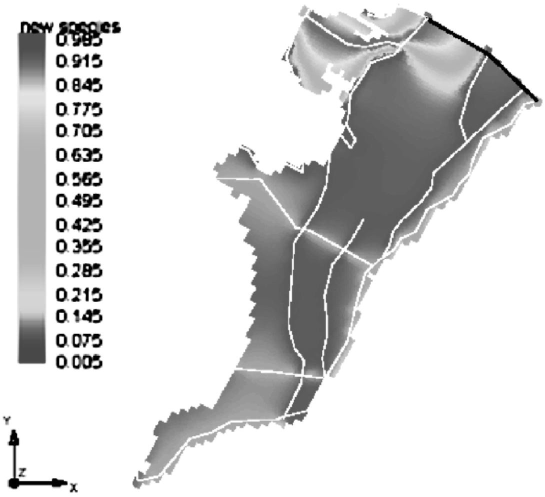

Figure 8. Results of the modeling scenario made to assess the impacts of remediation with initial concentrations as shown in Figure 7.

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USING OF GROUNDWATER FOR INCREASING OF WATER SUPPLY OF IRRIGATION SYSTEMS IN ARID ZONE

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Abstract: At present the problem of efficient using of surface and ground water for irrigation and sub-irrigation can be sorted out using the water-saving irrigation technologies and construction of automatically regulating sluices at open collecting-and-drainage net. These measures improve the water supply of agricultural crop and increase the yield, as well as decrease the technogenic burden of irrigation systems on natural landscapes.

Keywords: groundwater; irrigation; irrigation massif; evapotranspiration; seedbed; sluice-regulator; collecting and drainage network; water balance; furrow; mineralization of soil

1. Introduction

Irrigation systems of Kazakhstan are not full provided with water. In most cases deficiency of irrigation water can become covered due groundwater application for irrigation. Volumes of their using are estimated by natural capacitor volumes of groundwater tapping horizon and resources of others aquifer horizons due to an overflowing in groundwater tapping horizon or artificial

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stocks which are formed at hydraulic engineering construction and irrigation. In places of groundwater using for irrigation and sub-irrigation water-intaking constructions can work at the established and unsteady mode of groundwater. In the first case the water-intaking constructions work due to renewed sources of their feed when there is a good hydraulic connection with the rivers and water basins.

In the second case - volumes of groundwater using for irrigation are defined by water supplies of groundwater tapping aquifer horizon. In such conditions the operating mode of drainage systems should be coordinated with a technical condition of an irrigating network, irrigation technology, a mode of water-submission due to which renewed sources feeding artificial groundwater are formed, and discharges of water for evapotranspiration.

2. Long-Term Groundwater Mode on Irrigation Land

The analysis of references and own researches shows that on the irrigated lands in Kazakhstan groundwater mode is mainly formed by irrigational type. The dynamics of ground water level (rise and recession) is in direct dependence on volumes of water-submission, moisture of year (amount of precipitations), inflow and outflow of underground waters, technical condition of an irrigating network, a high-altitude arrangement of irrigation fields, their drainage rates, a mode and irrigation technology. Therefore the operating mode of drainage systems (especially vertical) should be defined within hydrological year in volumes of possible using of drainage waters on irrigation, and superficial layers of groundwater for sub-irrigation.

In such conditions of groundwater using for irrigation and sub-irrigation are regulated not static, but dynamic stocks which are formed, basically, due to filtration losses in irrigating network and on the irrigated lands, insignificant groundwater inflow from mountains Karatau and filtration losses from the main canal called Arys-Turkestan. At such conditions of formation of groundwater, quantitative values are defined as a difference between volume of a filtration feed and the discharge for evapotranspiration. Therefore the problem of rational using of superficial layers of groundwater on sub-irrigation should be solved by coordination of operating regime of drainage with irrigation regime.

For example, in wet years when water-submission exceeds 600 million m^3, intensity of work of vertical drainage should be defined by pumped volume of groundwater, at which level of groundwater is become stabilize on depth from 1.5 till 2.5 m (during the vegetative period). In droughty years when water consumption sharply grows, and water-submission is reduced, the drainage should work with full loading on an irrigation that will lead to draw off a level of groundwater up to 4 and more meters. The next years for restoration of groundwater dynamic stocks, it is necessary to increase water-submission and to reduce volume of groundwater using for irrigation as the vertical drainage provides only redistribution of water resources on irrigation massif and between years, instead of was an additional source of reception of water resources.

The analysis groundwater mode formation on the irrigated lands shows, that at the beginning of vegetation (April), when farmers irrigated alfa-alfa, winter wheat and carried out pre-irrigation, the level of groundwater rose up to the maximal levels - 0.8-2.5 m. The height of groundwater rise depends on a level of hydraulic relations between water-bearing horizons of overburden, presence of high-permeability layers (gravel-boulder and sandy bed) and a high-altitude arrangement of irrigation massifs (Figure 1). For example, on irrigated territories that occupied command position, groundwater rose up to 2.5-3 m, and on areas with the minimal high-altitude levels up to 0.8-1.5 m. The maximal parameters of groundwater level rise were formed in abounding in moist and damp years and minimal - in droughty and low-water years. Within hydrological year the amplitude of fluctuation of a level of groundwaters changed from 1 up to 1.5 m, and for the long-term period grew up to 2 m.

Fixed laws of groundwater level mode formation allow to develop actions for increasing of water availability for irrigated lands due to using drainage water for irrigation, and groundwater for sub-irrigation as shown by Kovda (1968). In the beginning of a vegetation period, when groundwater has low mineralization (less than 5 g/l) on lay with depth up to 2 m, they most were taken for sub-irrigation. From the middle of a vegetation period when rates of water requirement accrued, and the system of a vertical drainage started to work in a forced mode, for increasing water availability of irrigated lands, the share of groundwater using for evapotranspiration was reduced. At intensive swap-out of groundwater on irrigation speed of level recession of groundwater increased, processes of salt washing is amplified, stable salinity control was provided at reception of additional volume of water on irrigation. Such way of management groundwater mode formation provided negative salt balance on irrigated lands. However the level of their fertility did not rise, as the irrigation and cultivation amount were grew, that inevitably led to destruction of organic substances and leaching of nutritious mobile forms elements. Therefore methods of accelerated depletion of groundwater level are expedient for applying for lands where productivity of cultivated cultures is regulated by presence of increased stocks of salts.

3. Water Balance Depends on Ground Water Depth

On lands where there is no threat of resalinization (soil are not salty, groundwater mineralization is lower than 3 g/l) it is necessary to be guided by a sub-irrigation at which water loading on soils is reduced, the amount of cultivations is reduced, stability of agronomic structure of soils raises, technological leakage of water is decreasing for evaporation from a surface of land, stability in water supply of plants by a moisture is raises as shown by F. Karajeh et al. (2004). The choice of the given direction proves to be true by groundwater using dynamics for sub-irrigation (Table 1).

For example, at groundwater depth of 1 m about 85% of groundwater was spent for evapotranspiration, at groundwater depth of 2 m this parameter was reduced down to 43%. The portion of participation of groundwater in evapotranspiration did not exceed 10% at groundwater on depth of 3 m according to data Kharchenko (1975). Hence, it is possible to raise productivity of agricultural crops by managing of groundwater mode at deficiency of water resources.

4. Regulation of Drainage-Waste Water Regime

Regulation of drainage-waste water flow is reached by constructing of sluices - regulators at open collecting and drainage networks. They are applied on not salted lands where mineralization of groundwater does not exceed 3 g/l.

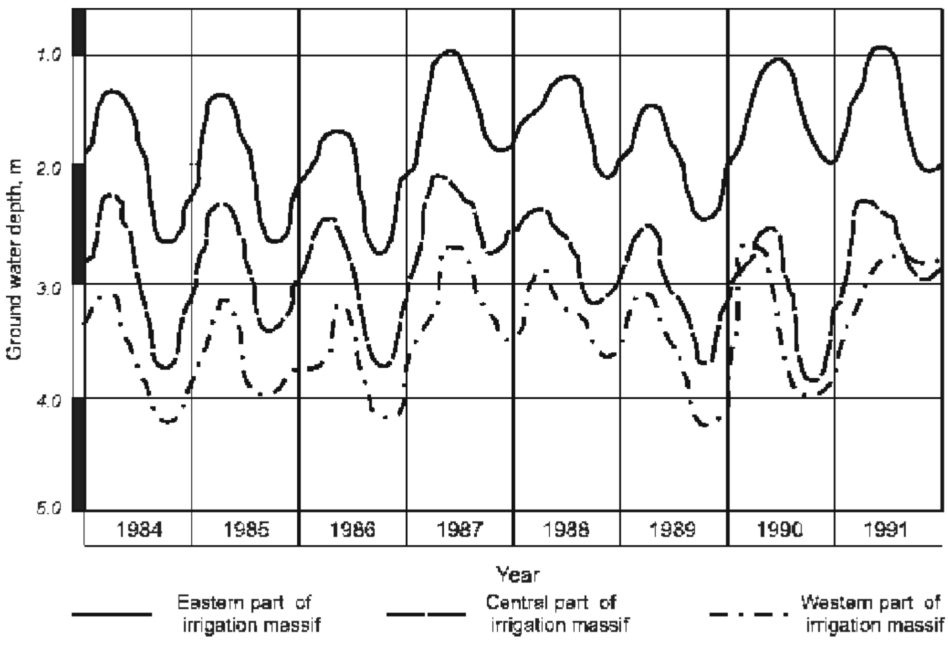

Figure 1. Level regime of groundwater, Stary Ikan village, Southern Kazakhstan oblast.

Table 1. Water balance of lysimeters at different depth of groundwater

Clauses of balance	Depth of groundwater, m					
	1.0		1.5		2.0	
	A	B	A	B	A	B
Income						
Initial moisture content, m^3/ha	3104	3104	5053	4800	6400	6400
Initial moisture content, in % from field cap.	97	97	95	95	95	95
Precipitation, m^3/ha	135	135	135	135	135	135
Irrigation rate, m^3/ha	-	650	-	1500	-	1900
Intake of ground water, m^3/ha	1141	6563	890	3766	734	2531
Income Total	4380	10452	5825	10201	7269	10966
Expense						
Finite moisture content, m^3/ha	2848	2720	4398	1044	5940	5062
Finite moisture content, in % from field cap.	89	85	87	80	88	75
Evapotranspiration, m^3/ha	1532	7732	1427	6157	1329	5904
Expense Total	4380	10452	5825	10201	7269	10966
Yield of cotton, ton/ha	-	3.8	-	3.4	-	3.3
Expenses of water for one ton of cotton, m^3	-	2035	-	1811	-	1789
GW in water requirement of cotton, %	-	84.9	-	61.2	-	42.9

A – lysimeters without cotton plants, B – lysimeters with cotton plants.

Head of drainage and waste waters in collectors carried out only during shortage waters (July-August) when groundwater falls deeply than 2 m. In the rest of the time of year collecting and drainage network should work in a free mode, providing water-downturn and desalinization of soil. Change of an operating mode of a retaining construction will inevitably lead secondary salinization of soil. Low cost of capital investments (from $30 to $40 per hectare) on construction of sluices-regulators predetermines efficiency of the given direction.

At such method of water resources management when irrigation came to an end in August, the vegetative period of a cotton proceeded up to the middle of September due to using of groundwater. It proves to be true experimental researches. For example, in 1999 (middle wet year) the irrigation period has ended at the end of August, and in 2000 (dry year) - in the beginning of August. In the first case by the end of vegetation (the beginning of September) the level of groundwater was lowered up to 2.5 m, and the second case - up to 3 m even at work of retaining construction (sluice - regulator) which reduced speed of draw off groundwater level up to two times. Hence, regional groundwater level is in direct dependence from the sizes of a water-fence, duration of the irrigation period and volumes of their discharge for sub-irrigation.

Figure 2. Sluice-regulator on collecting and drainage net.

Figure 3. Flood-discharge outlet.

As a result of experimental researches it is established, that significant volumes of groundwater (about 30% from evapotranspiration) participated in sub-irrigation when their level from spring by the autumn did not fall deeply than 2.5 m (Figure 4).

In dry year 2000 the period when groundwater level was above of 2.5 m, last from the beginning of carrying out of pre-irrigation and vegetative irrigation of winter wheat from April up to the middle of August, and in 2002 (middle wet year) - up to the middle of September. It is necessary to note, that on a pilot site where used a retaining construction for additional charging of groundwater, marks of their level raised on 0.3-0.5 m concerning adjoining territories. The period of decreasing of groundwater up to depth deeply than 2.5 m increased for 10-20 days. At the specified parameters of groundwater mode change the share of their participation for sub-irrigation was raised, water supply of plants therefore improved, especially during water sharp shortage period.

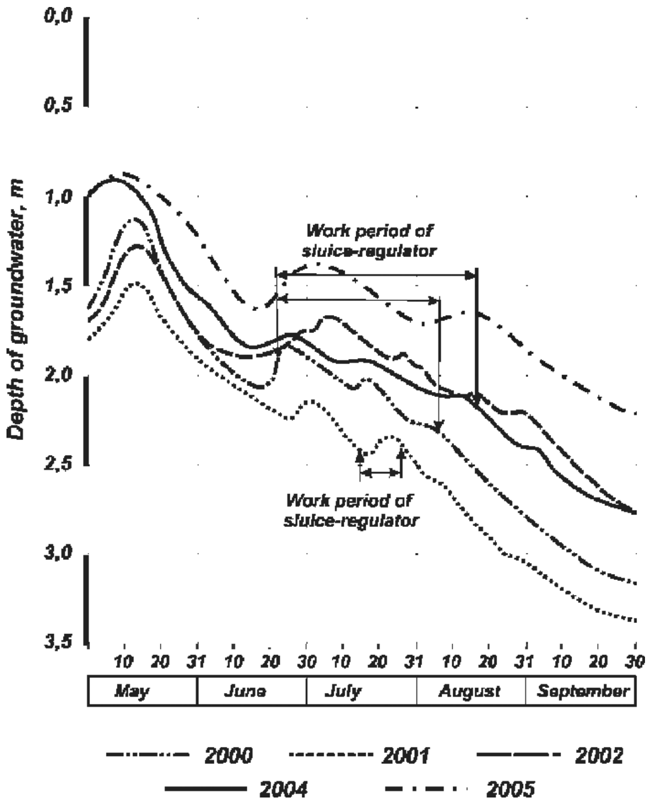

Figure 4. Dynamics of ground water level on pilot site during of vegetation period.

5. The Forecast of Groundwater Using For Sub-Irrigation

At balance researches volumes of groundwater using for sub-irrigation usually calculated by known formulas of Averyanov and Kharchenko given by Kharchenko (1975) which take into account only level of groundwater, and the influence of moisture of root zone on participation of groundwater in evapotranspiration appeared outside of calculation.

From our experimental data follows that the maximal discharge of groundwater (1809-1889 m^3/ha) for sub-irrigation was received in 2000 when the sizes of irrigation rates changed within the limits of 1320-1435 m^3/ha, and irrigation rates changed from 720 up to 800 m^3/ha at the first irrigation and from 600 up to 635 m^3/ha at the second irrigation. In this case decreasing of a moisture content to 50% from FC (field capacity) during the maximal water consumption (second half of August) has led to increase the discharge of groundwater for sub-irrigation. It proves to be true by lysimeters researches from which follows that the discharge of groundwater for evapotranspiration was predetermined not only their level, but also from moisture content in root zone of soil (Figure 5).

For example, after irrigation when moisture of soil in a root zone came close to a field moisture capacity, and parameters of evapotranspiration exceeded 10 mm/day, intensity groundwater discharge into the aeration zone did not exceed 0.5 mm/day. In process of moisture content decreasing in root zone parameters of intensity of the discharge of groundwater in a zone of aeration grew up to 8-11 mm/day when moisture content in root zone fell below 60% from FC as shown by Vyshpolsky et al., (2003).

Hence, quantity indicators of groundwater discharge in a zone of aeration depend not only on a thermal regime of irrigated lands, water and physics properties of soil, depths of ground waters, but also depend on dynamics of moisture content in root horizons of aeration. For updating regimes of irrigation and irrigation rates of agricultural crops we advance mathematical model by definition of groundwater using volumes for evapotranspiration in view of soil moistening regime and level of groundwater.

For development of mathematical model on forecasting the discharge of groundwater for evapotranspiration, in view of change of active moisture content in root zone of soil and a level groundwater, used experimental data and the differential equation of the first order as shown by Vyshpolsky et al. (2003):

$$f'(x) = -\sigma f(x) \tag{1}$$

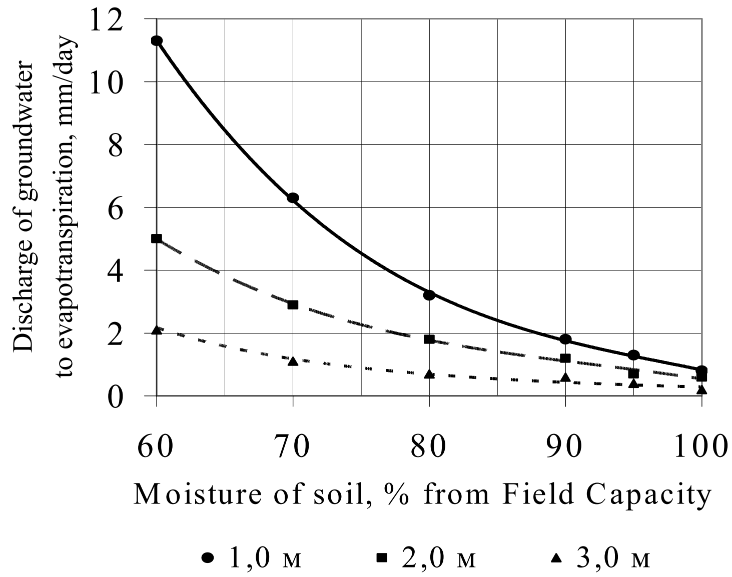

Figure 5. Discharge of ground water to the root zone at different level of ground water: 1.0 m; 2.0 m; 3.0m.

As the initial conditions have been accepted, that the soil is saturated up to field moisture capacity (FC):

$$\beta_{\max} = \beta_{\min} \qquad and \qquad \mathrm{K} = 0 \qquad at \qquad ET = 0 \tag{2}$$

Solving the equation (1) at the initial condition (2), we have received mathematical dependence for forecasting discharge of groundwater into the zone of aeration in view of change of a level groundwater and active moisture content in a settlement thickness of soil:

$$\mathrm{K}_{\text{т}} = \frac{\exp(H_x - H)}{\mu} \ln \frac{\beta_{\max}}{\beta_{\text{т}}} \tag{3}$$

where K_T is the discharge of groundwater into the aeration zone, mm/day; H_x is the depth of groundwater, equal -1 m; H is the real depth of groundwater, ($H \geq 1$ m); exp is the exponent; β_{max} is the moisture of soil, equal Field Capacity (FC), %; μ is the factor which is taking into account the influence of mechanical structure of soil to the discharge of groundwater into the zone of aeration, mm^{-1} ln is the natural logarithm; β_T is the moisture content of soil at the moment of time t, %.

For forecasting volumes of groundwater using for evapotranspiration and establishments of interirrigation periods (quantity of irrigation) in a denominator in the right part of the equation (3), instead of moisture content of soil at the moment of time t (β_t) it is necessary to use equation:

$$\beta_{\max} \exp\left(-\sigma ET\right) \tag{4}$$

where σ is the factor of the discharge of a soil moisture on evaporation, which depends on a thermal regimes of soil and changes: 0.0035 mm^{-1} for gray type of soil; 0.0030 mm^{-1} for chestnut type of soil; 0.0025 mm^{-1} for chernozem type of soil and ET is the evapotranspiration rate in mm/day.

In view of the given replacement the equation (3) will be transformed as follows:

$$K_t = \frac{\exp(H_x - H)}{\mu} \ln \frac{\beta_{\max}}{\beta_{\max} \exp(-\sigma ET)} \tag{5}$$

On the basis of experimental data have established quantitative values of factor - μ, which characterize the discharge of groundwater into the aeration zone and changes from 0.04 (for heavy soil) up to 0.06 (for light soil). In view of the given parameter equation (5) allows to predict of possible volumes of groundwater which can be used by plants in aeration zone for real level of moistening of root zone (Table 2).

Table 2. The discharge of groundwater into the aeration zone at changes of moisture of soil and depths of groundwater, mm/day

Moisture of soil,	Depth of groundwater, m					
% from FC	1.0	1.5	2.0	3.0	5.0	7.0
100	0	0	0	0	0	0
80-90	2.63	1.60	0.97	0.36	0.05	0.01
70-80	5.58	3.38	2.05	0.75	0.10	0.01
60-70	8.92	5.41	3.28	1.21	0.16	0.02
50-60	12.77	7.75	4.70	1.73	0.23	0.03

From our data it follows, that adaptation of agricultural crops to more rigid water regime of soil (settlement values of a threshold of pre-irrigation moisture in a root zone decreased for 5-10% concerning recommended values) will provide reduction of the irrigation rates sizes due to increase in depth of development of root system and volumes of groundwater using for on evapotranspiration. The comparative analysis of settlement parameters of the discharge of groundwater on evapotranspiration, calculated by equation (5), and experimental data has shown that maximum deviations did not exceed 10%.

Using of the given equation for updating irrigation rates, in view of change of a threshold of pre-irrigation soil moisture and a level of groundwater, allows to improve a control system of water resources (superficial and underground), to develop water-savings technologies and increase the productivity of crops at water deficiency, allows to reduce man-caused loadings on the soil and groundwater.

6. Groundwater Salt Regime at Using of Groundwater for Evapotranspiration

Reliability of use of the suggested dependence for development of actions on management of water and soil resources proves to be true by hydrochemical mode of groundwater (Table 3).

From the submitted data it follows, that the level of change of salinity in superficial layers of groundwater is predetermined by intensity of water exchange between loamy and gravels adjournment. During carrying out of irrigation the mineralization in superficial layers of groundwater raised due to receipt of salts with filtration water.

During the interirrigation periods, when not salted amplified (about 1 g/l) groundwater overflow from gravel adjournment into the superficial horizons of integumentary loams, their level of mineralization was reduced. Increasing the frequency of change of water exchange between water-bearing horizons integumentary and gravel adjournment provided desalinization superficial layers of groundwater and slowed down rates of salt accumulation on the irrigated lands. It proves to be true by change of mineralization level of groundwater sampled on the depth 3.6 - 4.0 m. In areas where initial salinity of groundwater did not exceed 1.5 g/l, limits of seasonal change of a mineralization variety from 0.004 up to 0.236 g/l, and in places where the mineralization of groundwater exceeded 1.5 g/l, its mineralization level decreased on 0.323-0.796 g/l.

From the submitted data follows, that the tendency of stabilization or desalinization of groundwater (the main object of land improvement) was formed due to the accelerated water exchange between water-bearing horizons of integumentary loams and gravel adjournment. For the last - high speed of a stream of groundwater which provided displacement of salt weights on a bias of irrigated territory is characteristic. It is necessary to note, that in work of a retaining construction the drainage and waste water which has collected in collecting and drainage network, it was filtered and created the screen which slowed down outflow of groundwater. Temporally reduction in speed of groundwater stream did not lead to amplification seasonal salt accumulation in superficial layers of groundwater. Therefore growth of volumes of participation of groundwater in sub-irrigation was not accompanied by deterioration of meliorative conditions as the level of seasonal salt accumulation did not exceed 5 ton/ha and easily leached by pre-irrigation before vegetation period.

Table 3. Changes of mineralization of groundwater, g/l

Year	Date	Depth of sampling, (m)	Pulse irrigation		Variable irrigation No of wells	Irrigation with constant flow	
			1	3	5	7	9
2001	May	2.2 – 2.6	1.881	1.328	1.438	0.907	4.855
		2.6 – 3.0	1.570	1.426	1.524	0.816	3.461
		3.6 – 4.0	1.660	1.342	1.250	1.031	2.717
	July	2.2 – 2.6	-	1.121	-	-	3.306
		2.6 – 3.0	1.548	1.322	1.623	0.954	3.286
		3.6 – 4.0	1.130	1.058	1.116	1.100	2.378
	September	3.6 – 4.0	1.299	1.119	1.106	0.994	2.183
2002	May	2.2 – 2.6	1.500	1.276	2.461	1.121	3.192
		2.6 – 3.0	1.487	1.378	2.834	1.031	3.879
		3.6 – 4.0	1.352	1.479	2.176	0.976	2.601
	July	2.2 – 2.6	-	-	2.659	-	3.115
		2.6 – 3.0	-	-	2.829	0.786	3.706
		3.6 – 4.0	-	-	1.842	0.626	1.986
	September	2.2 – 2.6	1.305	1.260	2.103	-	2.719
		2.6 – 3.0	1.361	1.210	1.513	0.868	3.055
		3.6 – 4.0	1.116	1.158	1.853	0.761	1.805

The offered actions on management of soil and water resources will provide: reduction of technological losses of irrigation waters on evaporation, a filtration, dump; increase in volumes of groundwater using for sub-irrigation; reduction of a drainage flow due to decreasing of loading on drainage systems; delay of soil degradation speed by reduction of amount of inter-seedbed cultivation and passage of soil-cultivating machines by dry furrows; reception of economically comprehensible crops (for farmers) at deficiency of water; increase of stability in an agricultural production; decrease rates of pollution of water resources and improvement of ecological conditions in areas of residing of agricultural population.

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GROUNDWATER MODELLING IN ATLANTIC COASTAL ECOSYSTEMS (SOUTH PORTUGAL)

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Abstract: Tróia-Melides is a sandy area on SW Portugal, between Lisbon and Algarve, on the Atlantic Portuguese coast. The top north of the tract is a sandy peninsula about 10 km long and 2 to 4 km large, and is represented by recent dynamic sand-dune systems (Plio-Plistocene-Halocene) that extends south to the rocky Sines Cape, in a total extension of 65 km (far beyond the zone covered by this work, that corresponds to northern half of this extension). The central zone of the littoral arch has sandstone cliffs headed by ancient dunes separated from the actual beach, which is continuous along the tract. A great diversity of important ecosystems characterises all the area, conferring it an enormous ecological fragility. Besides the dune and cliffs systems, 3 important coastal lagoons are also present. This diversity of coastal environments was responsible for the definition of protected areas, classified according the more important species (birds, rare plants). Concerning flora, 45 families belonging to 246 different species where identified in a recent study. The major percentage occupies the inner dunes and includes several priority species (Directive 92/43/CEE) – Lonopsidium acaule, Thymus camphoratus, Linaria ficalhoana, etc. A scarce variety occurs in primary and embryonic dunes (being

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Ammophila arenaria the most common). This area has a great touristic potential, which is not always in conciliation with its fragile ecology. However, it's a relatively well-preserved tract and is nowadays under strict land management rules. In the study area, 3 touristic areas are approved, with a total number of 39,300 touristic beds, distributed by hotels and vacation houses. As a result, some delicate situations have occurred in natural ecosystems, namely aquifer exploitation, without regarding its recharge capacity, which lead to the infiltration of saline water in the aquifers of the littoral areas. With this work it was possible to understand the role of the saltwater intrusion due to the raise of buildings and fresh groundwater demands, and the previsible impacts on coastal ecosystems. Some procedures were undertaken in order to know the hydrogeochemistry and hydrodynamical characteristics of the aquifers in the area, namely defining the thickness of the fresh groundwater and the position of the interface with the saline water on the coastal area

Keywords: Saline intrusion; fresh/saltwater interface; FEFLOW.

1. Introduction

The demographic growth and the touristic pressure on Tróia Peninsula, on the Occidental Atlantic coast of Portugal, have lead to a more intensive use of the inhabited areas and to the occupation of new ones, with the following demand of natural resources. Thus, the occurrence of disequilibrium situations on the natural systems, as overexploitation of groundwater resources, can be an important issue in the near future if the aquifer recharge capacity is not considered. In this shore zone these can led to an advance of the freshwater/saltwater interface or to the salinization of deeper aquifers.

The management proposals approved to the shore and near shore between Tróia and Sines are looking to discipline the human occupation, aiming to correspond to the population increase and subsequent growth of groundwater resources consumption. In the area of this study three new areas of touristic development are approved, with a total of 39,300 beds, between hotels and vacation houses. Therefore, it's important to develop studies that can permit the evaluation of the consequences of this increasing consumption and respond to the concerns with the possible advance of the saltwater interface.

In fact, with the increasing abstraction of water in several points of the coastal zones, the groundwater flow, originally in direction of the sea, can invert itself, creating a situation of saltwater intrusion. The abstraction wells in these coastal aquifers can begin to capture saline water when the mixture zone between fresh and saltwater or the salt water itself began to affect the active portion of the well. In other emplacements this contamination can result from the infiltration of saline aquifers waters, by dripping or deficient well construction.

As two fluids with different densities are present, an interface is created in each place where the fluids are in contact. The study of the position of the fresh/saltwater interface was done using several methodologies, between them mathematical models, specially the model of finite elements – FEFLOW 5.0.

Thus, a study between the Tróia Peninsula and Melides was developed, covering the municipalities of Alcácer do Sal and Grândola (Figure 1).

2. Geologic Characterization

The study area is partially included in the river Sado sedimentary basin. The other part is represented by Cenozoic rocks of the occidental coastal plain.

Figure 1. Geographic framing of the area of study.

According to Pimentel (1997), the Tertiary times evolution of Sado basin is essentially marked by the sedimentary answer to the succession of tectonic and climacteric events that act on the southwest of the Iberian Peninsula. It seems that the role of the marine eustatic processes have been secondary.

The structure of Sado basin has a sub-horizontal fulfilling, with the edges coincident with normal faults that where active during the basin subsidence (Ribeiro et al., 1979, in Gomes, 1992).

The occidental coastal plain is limited at West by the ocean and at East by the Grândola Mountain. The mountain marks the occidental limit between the Paleozoic and Mesozoic formations that occupy Santiago do Cacém's zone. The study area is composed by beach and dune sands, but this geology can be interrupted by Miocenic, Plio-Plistocenic, Plistocenic or Holocenic formations (Figure 2).

Miocene outcrops on the coast cliffs north of Sines cape are scarce. However, there are some norths of Fontaínhas brook. According to Tomé (1994), the outcrop of Aberta Nova beach is formed by silt and clay sands, fine grained and micaceous, and the Galé outcrop by clay, sandstone, limestone or calcarenites and lumachelic calcarenites. Plio-Plistocenic outcrops are present along the Tróia-Sines arc, forming the cliffs of all the central zone of the arc. According Gomes (1992), the characterization of the Plio-Plistocenic is common to almost the existing outcrops on the area: medium to coarse reddish sandstones, generally poorly consolidated, friable, with conglomeratic and/or argillaceous levels.

According to Oliveira et al. (1984), the Plistocenic is formed by old beach deposits, fluvial terraces, consolidated dunes and travertine limestones. The sands and gravels of modern beaches extend themselves between Fontaínhas brook, Melides, Santo André lagoons and Sines cape.

The Holocenic is formed by alluviums, namely the ones of Sado River, beach sands, dunes, aeolic sands and turf. The sandy coastal band, between Sines and Tróia, is totally covered by beaches, contacts with coastal dunes and forms on the northern part the Peninsula of Tróia. The sedimentary deposits are present along the Sado River, its tributaries and in some other rivers that discharge directly into the sea, on Santo André and Melides lagoons and in small wetlands either south of Santo André or in the region of Carvalhal.

3. Hydrogeological Framing

The study area is partially on the Tejo-Sado basin aquifer system, left margin, and part on the aquifer system of Sines (Figure 3).

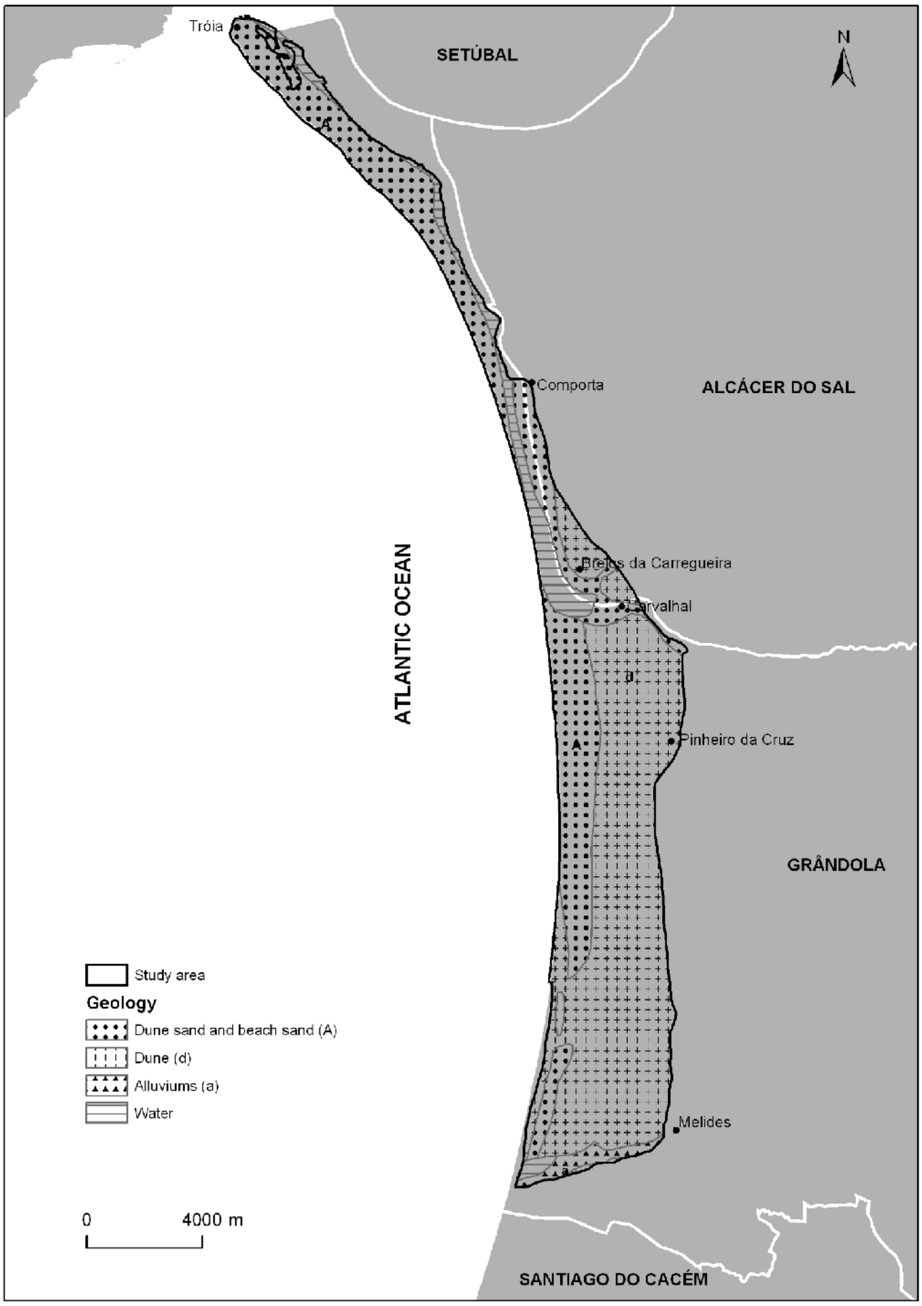

Figure 2. Geology of the study area (adapted from geologic map 1/500,000).

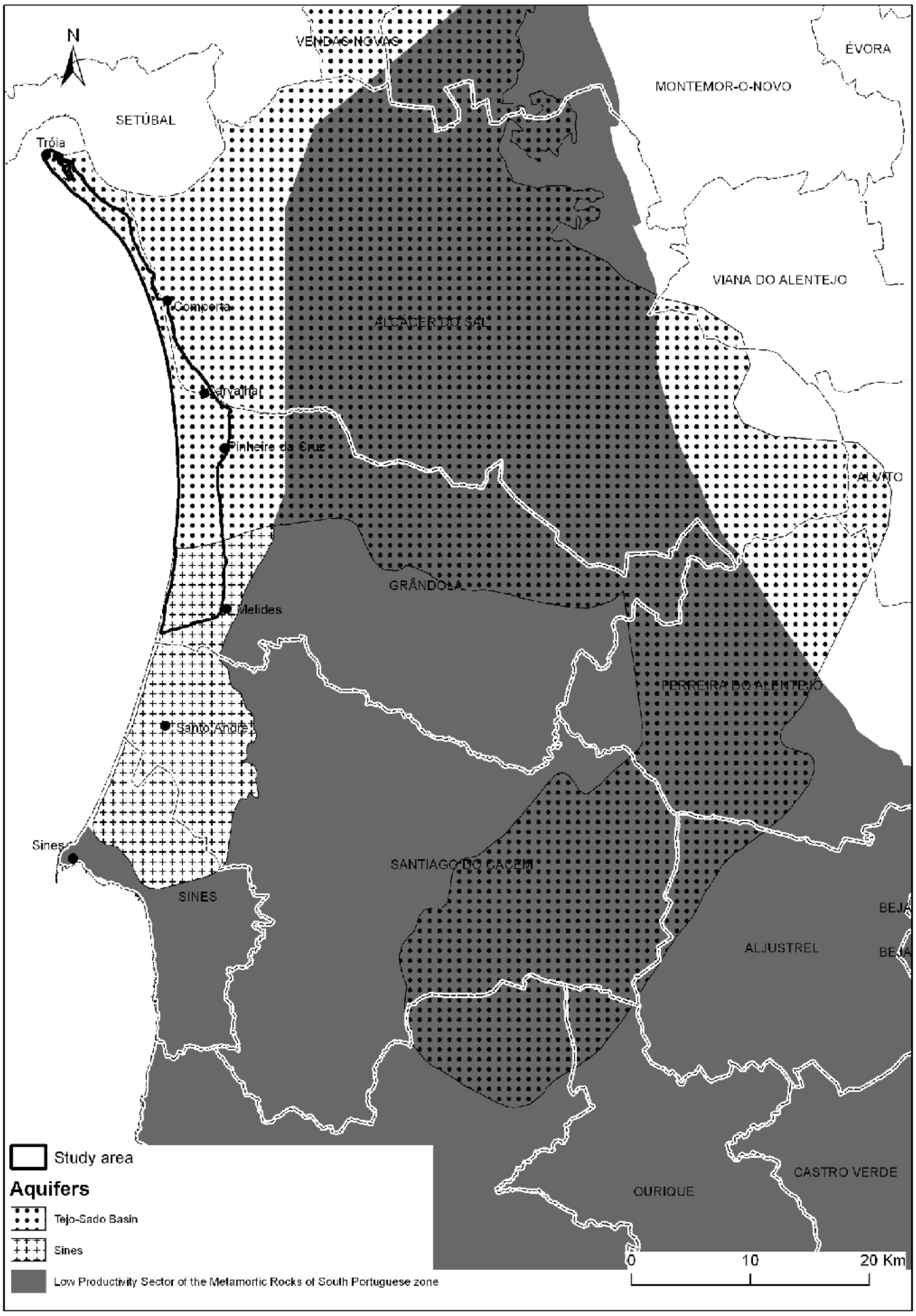

Figure 3. Hydrogeological framing.

The aquifer system of Tejo-Sado basin, left margin, assigned as T3, is part of the Tertiary basin unit of Baixo Tejo. According to Almeida et al. (2000), it integrates the biggest aquifer system of the Portuguese territory. Its groundwater resources constitute an important factor of development, once it assures numerous urban, industrial and agricultural supplies. In the study area the aquifer system is formed, according to Almeida et al. (2000), by a free aquifer, settled on the top layer of the Pliocenic and the most recent detrital deposits. Under these one is a multi-layer confined aquifer, supported by the base layers of Pliocene and the limestone layers attributed to the upper Helvetian. Under these ones, separated by thick marl formations, another multi-layer confined aquifer is present, supported on sandstone and limestone formations of the base of the Miocene.

The aquifer system of Sines, identified by O32, is formed, according Almeida et al. (2000), by a deep artesian aquifer on Jurassic limestones and dolomites, and a superficial porous multilayer aquifer installed on Miocenic and Plio-Plistocenic formations. These one is a free to confined aquifer.

The most important aquifer system on the region corresponds to the limestone Jurassic formations. The recharge is direct where the Jurassic formations outcrop (on the eastern limit of the system), and it’s done by the liking of the upper aquifers (Almeida et al., 2000).

The most superficial porous aquifer receives direct recharge from the precipitation, as it has a big outcrop area and hydraulic connection with the superficial waters.

4. Coastal Ecosystems /Conservation Policy

The Alentejo coast is one of the best examples of conservation of coastal areas in Europe, keeping the biophysics characteristics in all its extension, what confers it world importance. On the national context it can be said that some of the most important ecosystems of the Portuguese territory are on the Alentejo coast (Green Chart of the Alentejo Coast, 1998). Due to the importance, diversity and condition of coastal ecosystems, practically all the area is under status of protection. In particular on the present area of study, all the area was, in 1997, proposed to integrate the 2000 Nature Network, as Place PTCON 0014 – Estuary of the Sado, and Place PTCON0034 – Comporta-Galé, actually both of them approved. This fact attests the great value of the zone, concerning natural patrimony.

As the dunes and cliffs of the area, the three existing coastal lagoons assume extreme ecological importance. The lagoons of Melides (Figure 4)., Santo André and Sancha form an interface between the marine, surface and terrestrial ecosystems and they lodge a big diversity of fauna and flora (Green Charter of the Alentejo Coast, 1998). This fact was responsible by the creation of the so-called Natural Reservation of the Santo André and Sancha Lagoons. The three lagoon systems were formed at 5,000-7,000 years (Freitas, 1996), and they are separated from the sea by sand bars, establishing the connection to the sea under the sand bar; or by sporadic pass over. They are directly connected to the superficial water net by the brook that constitutes the main source of water. They are also connected with the most superficial aquifer, through the sandy substrate. The presence of the three environments – continental, marine and underground – coexisting on the water mass and varying along the year contributes to the diversity of the existent ecosystems.

Figure 4. Melides lagoon.

Still important is the existence of the Sado Natural Reservation, equally classified by its richness and diversity of fauna and vegetation. According to Gomes et al. (2003), in a recent project concerning the recovery of a dune system, fifteen vegetal communities where identified only on this coastal fragment. From those, eleven correspond to natural habitats of EC interest (Annex I and II – Directive 92/43/CEE). Differentiated according to the typology of coast environment where its occurrence is verified, several flora communities were identified, belonging to the following natural habitats of EC interest: high cliffs with vegetation of Mediterranean coast (Figure 5), mobile embryonic dunes, mobile dunes of the coastal belt, fix dunes or meadow/brush (Romão, 1996).

Figure 5. Two images of the Mediterranean sandstone cliffs vegetation (Galé-Fontainhas beach)

Concerning flora, 45 families where identify, corresponding to 246 different species (Gomes et al., 2003). The major percentage occupies the inner dunes and includes several priority species (Directive 92/43/CEE) – Lonopsidium acaule, Thymus camphoratus, Linaria ficalhoana, etc. A scarce variety occurs in primary and embryonic dunes (being Ammophila arenaria the most common).

The variations on the groundwater regime, being it due to natural causes or to overexploitation, will surely induce dangerous alterations in the dune and lagoon environments, due to its narrow connection and permeable substrate. In case of saline intrusion, it's highly probable that a progression of some species of plants on the inner dunes can occur, or even that some species disappear, namely on the lower parts, where the groundwater levels are closer to the surface. Concerning the lagoons, despite its short relation with the sea, the increase of salinization will have implications basically on the fish species level and consequently on its diversity.

5. Determination of the Hydraulic Parameters

In order to achieve the pretended goals, the knowledge of elements related with the upper aquifer of the Sado basin, and attending the scarcity of available information, namely the reduced number of wells with trustable data, it becomes necessary to create six piezometers and a well in selected places (Figure 6).

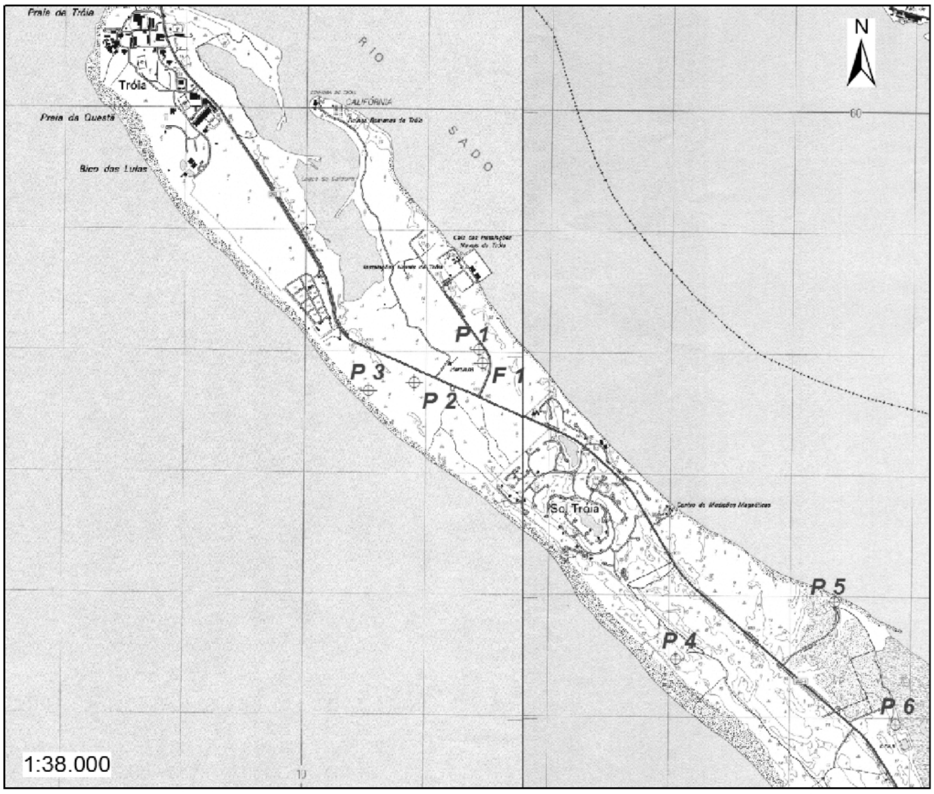

Figure 6. Geographic localization of the piezometers and wells that were created.

These new well and piezometers had permitted the analysis of the cyclic fluctuations of the piezometric level on the aquifer under influence of the variations of sea level, allowing to determine the hydrodynamic parameters of the aquifer.

The hydraulic parameters obtained were latter used to apply the mathematical model FEFLOW (Finite Element Subsurface Flow & Transport Simulation System), in order to study the actual position of the fresh/saltwater interface in Tróia Peninsula.

5.1. TIDE FLOW TEST

The analyses of the piezometric cyclical fluctuations of an aquifer under influence of a water free plan variation, like the sea, allow to determine the aquifer hydromechanical characteristics. After the definition of the diffusivity, the permeability and transmissivity are estimated.

The difusivity (D) was defined by two methods:

a) The tide amplitude (ΔH_0)

$$D = T/S = \frac{\pi \times x^2}{t_0 \times \ln^2\left(\frac{\Delta h_0}{\Delta H_0}\right)} \tag{1}$$

where D is the difusivity, in m^2/day; T is the trasmissivity, in m^2/day; S is the storage coefficient; X is the distance from the piezometer to the shore, in m; t_0 is the tide period, 12h 25m; Δh_0 is the semi-amplitude of the sinusoid obtained from the sea water level oscillation in the piezometer (oscillation amplitude in the piezometer); ΔH_0 is the semi-amplitude of the sinusoid obtained from the sea water level, in shore (tide amplitude).

b) From the tide delay (t_L):

$$D = \frac{x^2 t_0}{4\pi t_L^2} \tag{2}$$

If the two diffusivity values are not the same, then the aquifer has semi-confined characteristics. In these cases, the following equation must be used:

$$D = \frac{x^2}{2t_L \ln^2\left(\frac{\Delta h_0}{\Delta H_0}\right)} \tag{3}$$

The tide flow test was realized during 27 hours. The values of the tide oscillation were registered in the Sado River and, simultaneously, the hydrostatic levels were measured in the piezometers. The values of the tide oscillation were latter obtained on the Portuguese Hydrographic Institute.

The tide oscillation in the river and in sea had maximum amplitude of 2.79 m. Of the six observation piezometers, three don't present any variation in level (piezometers P_1 , P_2 , P_4, respectively at 302, 432 and 255 m of the shore). From all the others, the piezometer P5, at 62 m of Sado River, had presented as maximum variation of the amplitude, 0.40 m. The difference in relation with the tide maximum was 1h 30 m, and with the tide minimum was 2 h 30 m.

The piezometer P_6, south of P_5 and 72 m from Sado River, presented an amplitude variation of 0.17 m and a difference in relation with tide, between maximums and minimums, of 2 h and 3 h, respectively. The piezometer P_3, on the western area of the Tróia limit, 110 m from the sea, presented an amplitude variation of 0.15 m and a difference of 2 h and 3 h 30 m relatively to the tide.

That way, with the help of the previous equations, the diffusivity, transmissivity and permeability values were estimated for the piezometers P_3, P_5 and P_6 (Table 1). As it can be seen in the same table, higher storage coefficients correspond to higher transmissivity and permeability values.

Difusivity, transmissivity and permeability values estimated through the tide amplitude are very distinct from the ones obtained by the retard of the tide effects. The last values are always higher. However, the estimated values using the mathematical expression for semi-confined aquifers are very similar to the ones obtained from the tide amplitude, which could indicate the existence of some degree of confinement.

Considering the tide amplitude data for a storage coefficient equal to 15% (value that is considered adequate for the natural characteristics of the aquifer), medium values of permeability of 16 m/day, transmissivity of 950 m^2/day and diffusivity of 6300 m^2/day are obtained. Similar values were obtained with the granulometric curves.

Diffusivity is directly proportional to the transmissivity and permeability. However, despite the diversity of diffusivity values, the diffusivity always increases with the increasing distance of the piezometers to the shore.

Dill et al. (2001) have organized tide flow tests in 7 wells of the study area. A maximum tide amplitude of 0.60 m and a difference of about 2 h 30 m with the tide time were observed. The values are very similar with the estimated in this work. These authors also estimated the diffusivity values using the tide amplitude method and the delay method, obtaining, in the last one, 148 m^2/day, resulting in a hydraulic conductivity of 10 m/day (considering a saturated thickness of 60 m).

6. Mathematical Modelling

The mathematical modelling applied to the study of the saline intrusion that assume the existence of a transition zone formed due to the miscibility properties of two fluids with different densities (fresh and saltwater) and the dynamic dispersion, represents the salt intrusion problem in a more correct and real way than considering the simplification of the existence of a sharp interface between fresh and saltwater (Diamantino and Lobo Ferreira, 2002).

Table 1. Median of the hydraulic parameters estimated for the piezometers P_3, P_5 and P_6. D – Diffusivity; T – Transmissivity; K – Permeability; S – Storage Coefficient.

P_3								
Tide Amplitude								
			S = 5% e b = 60m		**S = 10% e b = 60m**		**S = 15% e b = 60m**	
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	387,68	9304,31	465,22	7,75	930,43	15,51	1395,65	23,26
Standard Deviation	8,36	200,70	10,04	0,17	20,07	0,33	30,11	0,50
Tide Delay								
			S=5% e b=60m		**S=10% e b=60m**		**S=15% e b=60m**	
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	3905,00	93719,99	4686,00	78,10	9372,00	156,20	14058,00	234,30
Standard Deviation	528,41	12681,86	634,09	10,57	1268,19	21,14	1902,28	31,70
Mathematical Equation for Semi-Confined Aquifers								
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	432,32	10375,65	518,78	8,65	1037,57	17,29	1556,35	25,94
Standard Deviation	35,70	856,73	42,84	0,71	85,67	1,43	128,51	2,14
P_5								
Tide Amplitude								
			S=5% e b=60m		**S=10% e b=60m**		**S=15% e b=60m**	
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	255,82	6139,70	306,98	5,12	613,97	10,23	920,95	15,35
Standard Deviation	8,80	211,27	10,56	0,18	21,13	0,35	31,69	0,53
Tide Delay								
			S=5% e b=60m		**S=10% e b=60m**		**S=15% e b=60m**	
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	1688,54	40525,08	2026,25	33,77	4052,51	67,54	6078,76	101,31
Standard Deviation	258,64	2563,17	128,16	2,14	256,32	4,27	384,48	6,41
Mathematical Equation for Semi-Confined Aquifers								
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	330,95	7942,87	397,14	6,62	794,29	13,24	1191,43	19,86
Standard Deviation	33,72	809,24	40,46	0,67	80,92	1,35	121,39	2,02
P_6								
Tide Amplitude								
			S=5% e b=60m		**S=10% e b=60m**		**S=15% e b=60m**	
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	144,69	3472,52	173,63	2,89	347,25	5,79	520,88	8,68
Standard Deviation	5,44	130,54	6,53	0,11	13,05	0,22	19,58	0,33
Tide Delay								
			S=5% e b=60m		**S=10% e b=60m**		**S=15% e b=60m**	
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	2277,16	54651,92	2732,60	45,54	5465,19	91,09	8197,79	136,63
Standard Deviation	575,19	13804,57	690,23	11,50	1380,46	23,01	2070,69	34,51
Mathematical Equation for Semi-Confined Aquifers								
	D (m^2/h)	**D (m^2/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**	**T (m^2/day)**	**K (m/day)**
Median	189,75	4554,07	227,70	3,80	455,41	7,59	683,11	11,39
Standard Deviation	30,96	743,00	37,15	0,62	74,30	1,24	111,45	1,86

The mathematical model FEFLOW (Finite Element Subsurface Flow and Transport Simulation System) was developed by WASY Institute for Water Resources Planning and System Research, Berlin, Germany, and is a sophisticated software that considers the above-mentioned characteristics. It consists on a 3D and 2D interactive simulation model of the groundwater flow, variable in function of the density, mass and heat transport processes in groundwater, using the numeric method of the differential partial equation resolution by finite elements (FEM).

6.1. FEFLOW APPLICATION IN TRÓIA PENINSULA

6.1.1. *Definition of the area*

The first phase of the salt intrusion modelling consists on the definition of the limits of the area to model. This corresponds to the area where the piezometers were placed. The west and east limits are the Atlantic Ocean and the Sado River, respectively. The north and south limits do not match natural hydrogeologic barriers. Once the area to model is relatively homogeneous, the option was to apply the FEFLOW accordingly an AB profile, in the area of the piezometers P_1, P_2 and P_3, because this is the place where more information is available.

The intention was to simulate in 2D the flux in stationary regime and the transport in transitory regime, in a saturated media and a vertical projection that is characterized by a 60 m thickness aquifer (Figure 7). In this 2D simulation, the area to model is formed by one single layer in which western and eastern limits match, has was referred previously, with the Atlantic Ocean and Sado River, respectively.

6.1.2. *Data for the flux model*

1. Starting conditions – The starting conditions relatively to the spatial distribution of the piezometric levels were fixed in zero m.
2. Border conditions – The border conditions were based on the following assumptions:
   - Condition of piezometric level imposition (1st order) – On western and eastern limits the imposed piezometric levels were estimated in function of the density and salinity of water in each superficial water body, considering the aquifer topographic level, according the formula proposed by Diersch (2002), for a constant density in depth:

$$h = h_S - \overline{\alpha} z \tag{4}$$

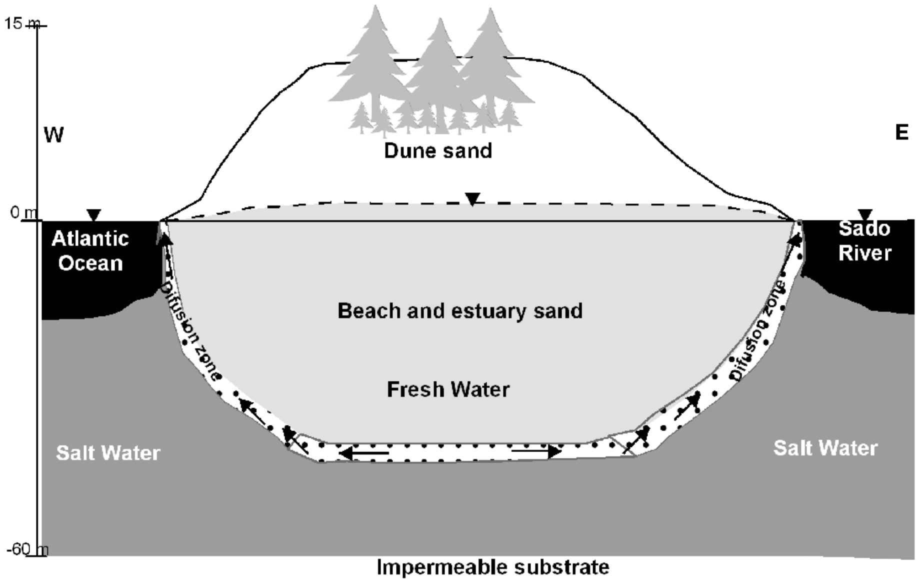

Figure 7. Schematic W-E profile of the upper aquifer of the Sado basin.

where h is the piezometric level measured in accordance with ρ_0, m; h_s is the piezometric level measured in accordance with ρ_s, m; α is the difference ratio of densities ($\bar{\alpha} = 0.7C_s / \rho_o$) and z is the topographic level.

Assuming that water level on the sea and on the river its zero, the previous expression takes the following form:

$$h = -\bar{\alpha} z \tag{4.1}$$

In spite of being both saline, the density of the sea water differs from the density of the river mouth water. In this case, $\bar{\alpha}$ will have different values, resulting in the equations:

$$h_{sea} = -0.0034z \tag{4.2}$$

$$h_{river} = -0.0032z \tag{4.3}$$

The salinity of the sea water and river water were measured in the area. For the sea the value was 48,575 mg/L and for the river 45,024 mg/L.

- Condition of the imposed flow (2nd order) – It corresponds to the aquifer recharge. The parameters of the materials considered for the flow equations were the following ones:

  – hydraulic conductivity (K_{max}) – 16 m/day
  – recharge – 2.33 x 10^{-4} m/day, or either, 85 mm/year
  – anisotropy (Ξ_{aniso}) – 0.1
  – ratio between the difference of fluid densities ($\overline{\alpha}$) – 0.0343
  – compressibility or specific storage $S_o = (S - \varepsilon_e)/b$ – 0.0001

6.1.3. *Data for the transport model*

1. Starting Conditions – The initial conditions for the transport model correspond to a salinity of 1 x 10^{-12} mg/L (it corresponds to the freshwater condition).
2. Border conditions – The border conditions were defined on the basis of the existence of two saltwater zones, in the side of the sea and the river. These border conditions are described after the:
   - Condition of imposed concentration (1st order) – Salinity values of 48,575 and 45,024 mg/L were considered in the west and east borders, respectively; in the recharge zone a concentration of 0 mg/L was considered.
   - The parameter values of the materials inserted for the transport equations had been the following ones:
     - porosity – 0.3
     - molecular diffusion – 1x10^{-9} m^2/s
     - longitudinal dispersivity – 15 m
     - transversal dispersivity – 1.5 m
   - On the sea side and river borders a condition of minimum flow was defined as zero, what means that the borders are only active if it exists a mass flow entrance, and not the reverse.

6.2. OBTAINED RESULTS

After the introduction of all these data, the model was calibrated. The values initially considered and interpolated for the areas without information, the hydraulic conductivity and recharge, were modified in order to confirm that the piezometric values estimated by the model approached the values measured in the piezometers P_1, P_2 and P_3.

The final map of the hydraulic conductivity distribution values can be observed in Figure 8, after the model calibration.

The value of the hydraulic conductivity initially considered (16 m/day) suffered a slight reduction, with the values between 8.4 m/day (in the center of the Peninsula) and 13 m/day (in the edges of the Peninsula). Dill et al. (2001) had obtained hydraulic conductivity values in the order of 10 m/day in another study also carried out in the Peninsula of Tróia.

The map of groundwater flux lines allows to understand that the flow direction is from the centre of the Peninsula to the edges (Atlantic Ocean and River Sado). It is also observed that saltwater proceeding from both the ocean and the river mouth enters in contact with the aquifer freshwater and is later discharged again. As can be observed in Figure 9, once fluids of different densities are involved, a dynamic mixture zone is formed, called interface. In spite of the interface conception, evolving the notion of gradual transition between medias, the salinity value of 13,500 mg/L was considered as threshold of the assigned transition zone, more or less 20,000 µS/cm in EC.

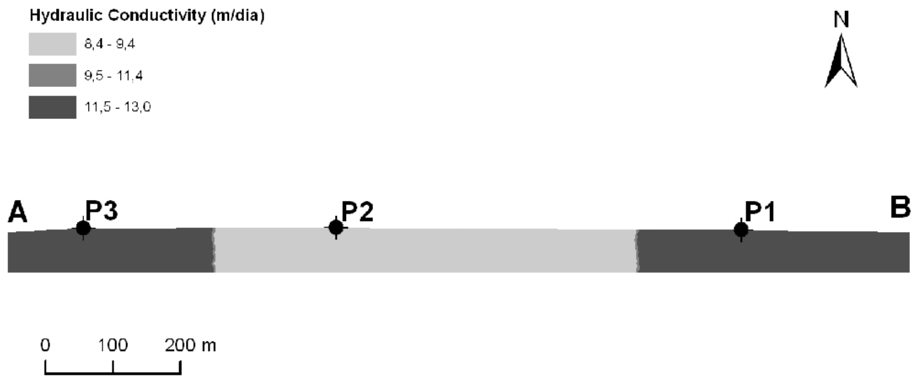

Figure 8. Hydraulic conductivity map.

In general terms, the thickness of this interface is about 20 m and its position varies along the AB profile, being deeper in the center of the Peninsula and more superficial in the border. Near the ocean the interface is placed 5 m bellow the surface and it increases to the centre of the profile. In the piezometer P_3 (110 m off the shore) the interface is 12 m depth. A similar value was estimated through the analytic model. At 430 m of the shore, in the piezometer P_2, the interface is at 32 m from the surface. In the centre of the profile, the interface reaches the maximum depth, about 44 m of the surface. Similar value was obtained by the analytical model. Near Sado River, the interface is placed 6 m under the surface.

7. Final Considerations

As graphical version of the model results, the map of the groundwater flux lines had permitted to conclude that the flow direction is from the Peninsula centre to the edges.

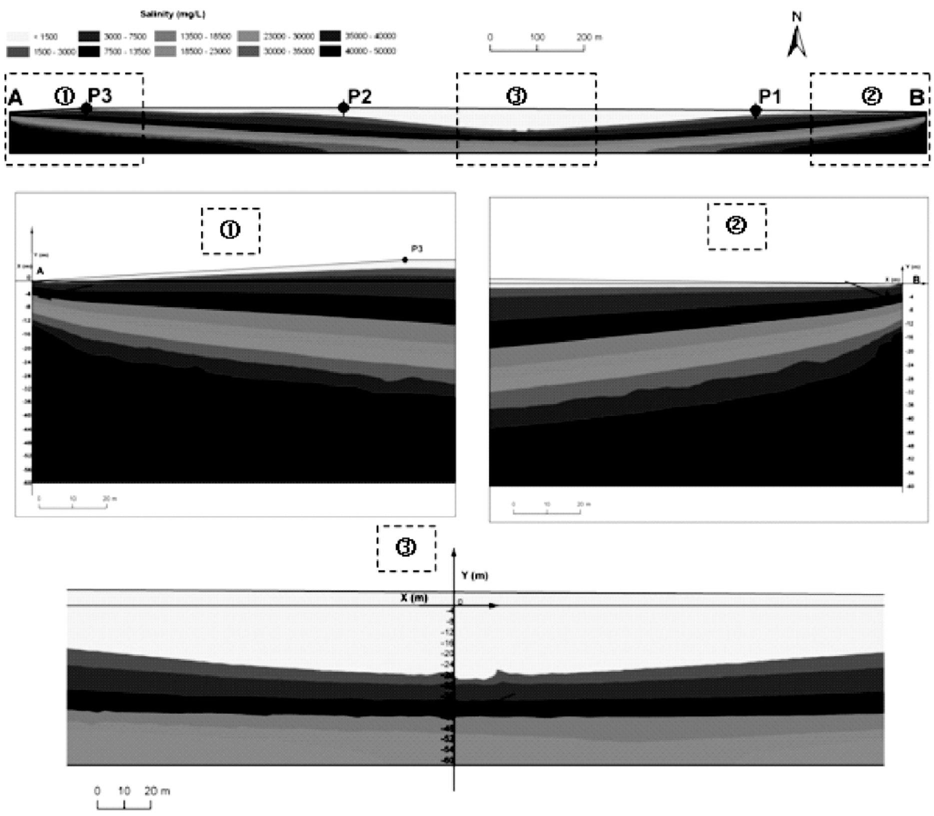

Figure 9. Salinity variation (mg/L) along the AB profile.

Otherwise, the final map of the salinity distribution shows that the interface has an average thickness of about 20 m. As expected, its position is deeper on the centre of the Peninsula. Here, the interface is at 44 m of the surface (similar value was estimated by the analytic model).

The result of the reflection resulting from the crossing of all the collected information and the result of this study permits to point out that, presently, the process of saline intrusion in the superficial aquifer of the Sado basin, occurs in a context of natural balance, that is, the position of the fresh/saltwater interface is only established by the actions of natural factors (tide oscillation, recharge, etc.). The aquifer is only dependent of the lateral pressure induced by the sea and the river.

Beyond these lateral pressures in the Peninsula of Tróia the Sado basin deeper aquifer his dependent of the vertical interaction with the saltwater. According to Mendonça (1992), since 1960 the water levels had go down, with the beginning of important water supplies to industry. The static levels had come down from + 6.5 m in 1960 to –20 m nowadays and the dynamic levels in the wells are now – 40 m.

This way, the drawdown of the piezometric levels in the Sado basin deep aquifer, in association with the presence of ruptures in the well's casing in the zone of the superficial aquifer, led to inter-aquifer contamination, with influence in the increase of the water mineralization in the analyzed points.

Thus, it's important to assure that the human activities with direct influence on the water resources of the Peninsula of Tróia are managed with the right precautions.

As an example, the following proceedings are suggested:

- If possible, monitoring the piezometric levels in all the active wells in the area of study, in order to compare the data with the existing historical data.
- To install, in some wells, a monitoring system of some physical and chemical parameters.
- During the construction of wells, it must be assured that the deep aquifer, responsible by all the supply water to the populations, is not contaminated by the liking of saltwater from the upper aquifers.
- The drilling and well construction must be followed by specialists.
- To promote environmental education actions concerning the necessity to ensure that the exploitation flow is not exceeding recharge aquifer capacity, as a form to prevent undesirable situations of overexploitation.

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GLOBAL CHANGE, ITS IMPACT ON FUNCTIONS OF AQUIFER SYSTEMS AND THEIR DEPENDENT ECOSYSTEMS

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Abstract: Terrestrial changes are taking place across the globe that far outpace the global climate change in the short term time scale of the forthcoming couple of decades. These changes include the world wide mobilisation of natural lands into agriculture, transformation of rural lands into urbanised areas and the intensification of industry. Change of this nature, that the natural scientists might have till recently considered less relevant, now needs to be considered in the management of aquifers. In this paper *globalisation* is taken to refer to the onset of global change derived from such drivers. If all impacts of change occurring in a hydrological system are quantified, then available evidence suggests that only 20% of the impact could be ascribed to 'climate change', while 80% would be ascribed to the globalisation as defined here. Given that between 30% to 60% of annual flows, essentially baseflows, in watersheds arise from aquifer systems within them, the sound functioning of aquifer systems, in terms of sustained discharge to coastal areas and wetlands is fundamental to the sound functioning of those aquatic ecosystems. Research is needed to assess, evaluate and develop investment policies to ensure sustainability in groundwater dependent ecosystems.

Keywords: global change, globalisation, aquifers and ecosystems dependent on them, aquifer discharge to coastal ecosystems, aquifer discharge to wetlands, sustainability of groundwater dependent ecosystems

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1. Introduction and Overview

It is generally agreed that apart from the growing impact of the forthcoming greenhouse gas induced climate change, terrestrial global change is already making a measurable impact within watersheds and in aquifers. If follows thus, that for aquatic ecosystems, in addition to the forecast impact of climate change, terrestrial impacts, as transmitted through a watershed's hydrology & hydrogeology, could be more significant in the short term. Closely related to this is the impact on the *functions* of aquifer systems, as receptors of annual recharge, its storers and its transmitters. Aquifer systems are also dischargers of groundwater; after enabling it to gain characteristic physico-chemical properties, they discharge into streams, inland water bodies, wetlands or coastal zones, through mangroves and lagoons. Many of these latter water bodies support innumerable and highly valuable aquatic ecosystems on which a significant proportion of human kind is also directly dependent.

1.1. GLOBAL CHANGE VS CLIMATE CHANGE

The terrestrial global change referred to above comprises of the rapidly accelerating worldwide mobilisation of natural lands for agriculture, transformation of rural land into urbanised areas and increasing population pressures in urban areas as well as intensification of industry. There is clear evidence that the rapid contemporary change in land use and land cover has significant impacts on local environmental conditions and economic and social welfare. For the geo-scientist and the engaged experts, change has long been a consistent feature of the world. Contemporary change, however seems quite distinctive in its pace and geographic scope. The drivers of the current global change, which eventually impacts aquifer systems, feature rapid conceptual reordering of time and space, facilitated by the integration of financial systems, the internationalisation of production and consumption, and the spread of global communication networks. In the past, changes were arguably more spatially bounded and limited in scope, and people and places had more time and space to adjust. Today, changes that natural scientists might have till recently considered less relevant, include the free flow of capital, the growth of multinational corporations, international labour migration, and the emergence of a global mass culture, and these predominate over climate change, at least in the short term. In this paper *globalisation* is taken to refer to the onset of global change derived from such drivers.

It is argued that if all the contemporary impact of change that occurs in a hydrological system is quantified, then only 20% of the impact could be ascribed to climate change, while 80% would be ascribed to the globalisation as defined above. Experience, and the sparse data that is available, would suggest that the source of this 80% impact may be attributed to change in vegetation, to land surface characteristics, soil & subsoil properties, water related infrastructure (e.g., dam construction, irrigation, canalisation, and drainage of wetlands) as well as roadways, railway lines and buried pipelines. In addition, "non eco-friendly" land-use and land-cover change, soil management, coupled with climate variability all interact to produce impacts on watershed hydrology, especially to the recharge of aquifer systems that significantly exceed the impact of climate change.

2. Changes in Aquifers in Global Terms

Given that between 30% to 60% of annual flows in a typical watershed arise from aquifer systems within it, the sound functioning of aquifer systems is fundamental to the sound functioning of aquatic ecosystems. Loss of aquifer storage over the last two decades has been significant, for example the volume of water depleted from the High Plains Aquifers alone (in the western US) is estimated as being equivalent to 0.75 mm rise of sea level (Konikow 2005). The gradual impact of loss of this type of storage will be reduction in discharge to surface bodies and increased stress on ecosystems. Just as broader ecosystems shape social and cultural networks, by providing essential renewable resources, aquatic groundwater dependent ecosystems sustain biodiversity by providing goods including nutrition, fiber, clean water, recycling of harmful elements. In some cases aquatic ecosystems are basic to cultural, spiritual, and aesthetic returns.

2.1. ECOSYSTEMS DEPEND ON AQUIFER DISCHARGE

Aquifer dependent ecosystems also provide services that include the regulation of the hydrological cycle, quality of water, storage and biogeochemical cycling of nutrients, and indirectly, the provision of habitat, and provision of beauty and inspiration. While these ecosystem generated goods can pass through markets, the aquatic ecosystem services rarely do, and thus remain difficult to value.

2.2. THE SHORT TERM ISSUES IN MANAGING AQUIFERS

It is argued that while climate changes have significance at the planetary scale and in geological time scales, for aquifers and their dependent ecosystems, globalisation is more urgent and significant for present planning horizons (the next tens of years). This premise requires the consideration of the short range outcomes of malfunctioning of an aquifer system and following on from this, the impact on groundwater dependent ecosystems; further, there is a need to consider the immediate adaptation measure would be needed, and most importantly how they may be financed. Ensuring the desired provision of broader ecosystem goods and aquifer dependent services will require understanding of interactions among basic ecosystem processes and developing approaches to reduce the vulnerabilities to, or take advantage of early opportunities that arise because of, global change and long term implications of climatic changes. Science based practical research can contribute to this societal global goal by addressing relevant questions that focus on linkages and feedbacks between ecosystems and drivers of global change, important consequences of global change for ecological systems, and societal options for sustaining and enhancing ecosystem goods and services as environmental conditions change. This research will produce critical knowledge and provide a forecasting capability that will continuously improve decision making for resource management and policy development.

3. Research Needs to Develop Policy

Two specific areas for research and study can be identified: the coastal environments in which groundwater discharge plays a significant role and some inland wetlands that are supported by groundwater flows.

3.1. ECOSYSTEMS DEPENDENT ON COASTAL AQUIFER DISCHARGE

Coastal aquifers discharge to the sea, directly or through estuaries, coastal lagoons and mangroves. The groundwater discharge, which is generally fresh water, seeps out and mixes with sea water, but at the interface, can create appropriate conditions for specific marine aquatic communities, which are often fragile and susceptible to changes. Inland groundwater extraction or contamination can reduce natural coastal discharge or pollute the environments for the near coastal biota and fish hatcheries. Insufficient research has been carried out to quantify these risks, but part of global loss of coastal fisheries can be attributed to this.

The management of coastal aquifers has not featured prominently in global, national or local coastal zone studies (see e.g. Post 2005), which are heavily biased towards surface water. It is, therefore, not surprising that coastal groundwater governance and practical management are not well understood. In consequence important opportunities for integrating this vital resource and managing it conjunctively with surface water, to optimize coastal zone benefits, and reduce risks are lost. Such lost opportunities include preservation of terrestrial and marine biodiversity, reduction of saline intrusion, loss of lagoons and mangrove, as well as water for human consumption and use.

3.2. ECOSYSTEMS DEPENDENT ON AQUIFER DISCHARGE TO WETLANDS

The function of aquifers, in relation to wetlands, tend to be less understood than functions of surface waters. While wetlands exist because they overlie impermeable soils or rocks and there is, therefore, little or no interaction with groundwater, there are innumerable cases of wetlands that are dependent on aquifers fed largely or wholly by groundwater, for example, wetlands that form at springs, oases and many lakes. In some instances, wetlands & the biodiversity they support, derive from the very character of the aquifers underlying them in terms of the quantity and the quality of the discharge water. In other cases, such as floodplains the alluvial aquifers act as the off-line storage, which can provide the vital discharge during extreme drought.

4. Concluding Remarks

There are global changes that are occurring in parallel ways, which have a significant impact on the way in which groundwater will have to be managed in the forthcoming decades for environmental and ecosystem sustainability. However, there is limited quantification of the ecosystems that are dependent on groundwater and the services it provides. Available evidence suggests that poorly understood and recognised linkages between groundwater outflows to coastal and wetland ecosystems could threaten their sustainability. In effort to address these issues the Global Environment Facility (GEF) has consented to assist countries and to co-finance some projects in key ecosystems. The outcome of these studies together with others that are underway will help to define policy and economic instruments.

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THE VULNERABILITY OF GROUNDWATER DEPENDENT ECOSYSTEMS: A STUDY ON THE PORSUK RIVER BASIN (TURKEY) AS A TYPICAL EXAMPLE

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Abstract: Groundwater dependent ecosystems can maintain their current composition and functioning by the groundwater input. Ecosystems are significantly influenced by changes in groundwater depth, pressure, flow rate and quality. Various pollutants such as pesticides, herbicides, fertilizers and other chemicals affect groundwater quality. Other pollutants having highly toxic features for environment such as industrial effluents, irrigation return-waters, and leachate from waste disposal areas may also affect groundwater quality. The Porsuk River Basin (NW of the Central Anatolia, Turkey), increasingly polluted by industrial, agricultural and municipal activities accompanied by growing population, can be considered as a typical example for groundwater dependent ecosystems. Organic and inorganic contaminants in surface and groundwater of Porsuk River Basin were assessed by collecting 32 water samples from the river and its canals and shallow and deep wells during two sampling campaigns conducted in 2001 and the obtained results were compared to the previous analysis results. These results show that both Porsuk River and groundwater in some parts of the basin have been intensively polluted in terms of NO_2, NO_3, NH_3, NH_4, phenol, AOX, phosphorus, free chlorine, sulfur, Fe, Al, Pb, Cr, Mn, Cd, and Zn. The pollution is over the allowable limits for drinking and domestic usage according to both TDWS and WHO. The abundance of aquatic Oligochaeta (especially Tubifex tubifex and Limnodrilus hoffmeisteri), bioindicators of organic pollution, along the Porsuk

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River also confirms these results mentioned above. On the other hand, the groundwater-surface water interaction plays a very important role on the deterioration of the groundwater quality in the Porsuk River Basin particularly during the dry period.

Keywords: groundwater, Porsuk River, ecosystem, pollution

1. Introduction

Groundwater dependent ecosystems are of particular importance ecologically because of their unique fauna and flora. Removal of groundwater from these ecosystems, or a change in the timing, quantity, quality or distribution of groundwater may influence ecosystems. Disruption of the natural groundwater-ecosystem interaction via recharge or extraction can further imperil some species and communities. Changes in groundwater depth, pressure, quality and flow rate influence these ecosystems rapidly. Changes in land-use can also affect the quality of groundwater through increased use of pesticides, herbicides and fertilizers, which may ultimately reach the water table and then influence groundwater dependent ecosystems. Even slight alteration of groundwater dependent ecosystems may potentially lead to biodiversity declines and affect ecosystem structure and function (Melanie et al., 2003). The degradation of groundwater quality depends on the pollution load and the behavior of pollutants as well as the geological and hydrogeological factors that control the flow and dispersion.

The Porsuk River Basin (NW of the Central Anatolia, Turkey), increasingly polluted by industrial, agricultural and municipal activities accompanied by growing population, can be considered as a typical example for groundwater dependent ecosystems. Agricultural runoff, urban runoff, leaking septic systems, sewage discharges, factories, surface water-groundwater interaction increase the contamination level of the water resources within the Porsuk River Basin (PRB).

Previous studies on the field (DSI, 1975; Özbek, 1976; Kaçaroğlu, 1991; Kaçaroğlu and Günay, 1997a and b; Bakış, 1996; Özçelik, 1998 and EGDE, 1999) revealed that there is a significant pollution in the groundwater of the PRB. However, heavy metal and pesticide contaminations were not studied. The main objectives of this study are to evaluate the effects of industrial and agricultural activities on groundwater quality of PRB in details. Organic and inorganic contaminants in surface and groundwater of PRB were assessed by collecting 32 water samples from the river and its canals, and shallow and deep wells during two sampling campaigns conducted in 2001.

The obtained results show that both Porsuk River and groundwater in some parts of the basin have been intensively polluted in terms of NO_2, NO_3, NH_3, NH_4, phenol, AOX, phosphorus, free chlorine, sulfur, Fe, Al, Pb, Cr, Mn, Cd, and Zn. The pollution is over the allowable limits for drinking and domestic usage according to both TDWS and WHO. Besides this, the leachate, rich in heavy metals, organic substances water from the municipal waste deposition area located on a valley causes excessive pollution due to the unsanitary landfill.

2. Location and Geological-Hydrogeological Setting

The study area is located in Eskisehir (Figure 1), NW of the Central Anatolia, Turkey, in an area of 500 km^2 between 39^{o}41'00" - 39^{o}32'30" latitudes and 30^{o}22'46"-30^{o}45'00" longitudes. Based on the data of the Eskisehir meteorological station for the period of 1930-90, mean annual precipitation is 382 mm and mean annual temperature is 10.90^{o}C.

The oldest formations in the region are schist, crystalline limestone and ophiolitic mélange in Triassic age. Eocene units, conglomerates, marls, claystone, limestone, clays and tuffs overlie the Triassic units with an angular unconformity. Lower and Upper Miocene units which consist of tuffs, agglomerates, basalts, conglomerates, sandstones, marls, gypsum, marl-clay with gypsum, and claystone are found above the Eocene units uncomfortably. Pliocene overlies all older formations by unconformable with the marl-claystone series. Old alluvium of Pleistocene (Villafrancian) crops out at the edges of Eskisehir plain, formed of silt, clay, sand-fine sand, fine gravel, sandstone layers and gravels (Gozler et al., 1996). The total thickness of old and young alluviums are 120 m, while the young alluvium in Quaternary age consists of unconsolidated and actual river sediments with a thickness of 15 m.

The crystalline limestone, gabbros and serpentines of the mélange yield limited amounts of groundwater within their fracture systems. Conglomerate and sandstone levels of Eocene and limestone levels of Neogene also yield limited amounts of groundwater. Old and young alluviums are the main aquifers in the area.

The water table contour map was prepared by evaluating the water levels of drilled wells. The groundwater is discharged to the Porsuk River, according to the flow line map at the left bank of Porsuk River. Although this trend is observed in higher altitudes, this situation disappears due to the intensive pumping by the drilled wells at the riverbank. At the right bank, groundwater recharges to the river (Figure 1).

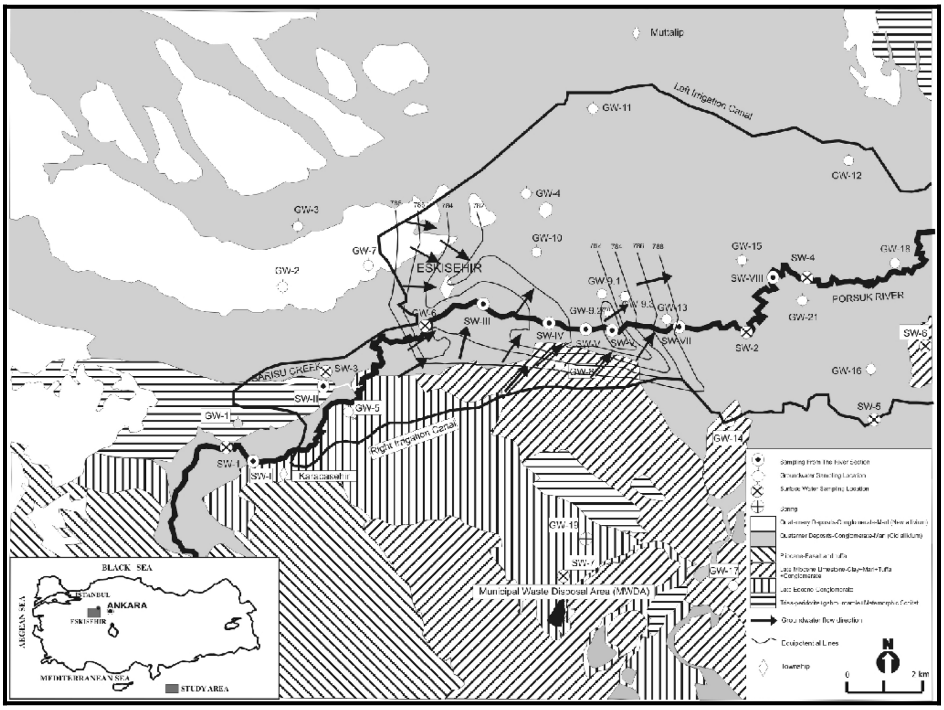

Figure 1. Geological Map and Water Samples Location in the Study Area (After Yuce et al., 2005).

3. Material and Method

Water samples from surface-water bodies (8 samples) and shallow and deep wells (24 samples) were analyzed for their major and trace ion and pesticides concentrations. The sampling campaigns were covered from May 2001 to October 2001. In addition, the Eskisehir branch of the General Directorate of Environment -EGDE (1999) data from the eight sections along the Porsuk River were also utilized for the comparison of the last and previous analyses results.

The sampling campaigns were performed in May and October 2001. The samples were collected according to the Turkish Standards for Water Sampling and Analyzing Methods. In sampling, one liter double tapped hard plastic and dark colored bottles were used. In order to prevent complex formation of heavy metals with oxygen, pH ≤ 2 conditions should be maintained. The heavy metal analyses were made by Atomic Absorption instrument. TDS (Total Dissolved Solids) was measured by evaporation. COD and BOD were sampled and measured using standard methods (Greenberg et al., 1985).

Pesticide samples were analyzed at the Refik Saydam Central Poisons Research Laboratory by using the instruments such as HPLC, GCL-ECD, GCL-TID, and GCL-FID.

4. Results and Discussion

4.1. GROUNDWATER (GW)

The analysis results evaluated according to the Turkish Drinking Water Standards (TDWS, 1997) Code: TS-266 and Turkish Irrigation Water Standards (TIWS) are given in Table 1 and 2, respectively. Water samples in wells GW-7, GW-9.2, GW-1, GW-19, GW-5, GW-6, GW-13 and GW-19 are over the admissible limits according to TDWS with respect to nitrite, nitrate, boron and phenol. The phenol contents are much higher than the allowed value for drinking water in GW-3, GW-5, GW-8, GW-9.2, GW-13, GW-19 and DW-1 (Table 1). The possible sources of this high phenol content are chlorine pesticides and industrial wastes.

The nitrogen derivatives (NO_2, NO_3, NH_3 and NH_4) are high in groundwater samples collected from the investigation area. In the shallow aquifer, under sufficient drainage conditions, the NO_3 content increase with the thickness of the unsaturated zone (Postma, 1992). If the drainage is weak and permeability is low, the denitrification process occurs and NO_3 decreases. The increase in NO_3 with the thickness of the unsaturated zone is indicated in Figure 2. In the wells GW-1, GW-7, GW-10, with high NO_3 contents, the sand and gravel layers are predominant.

Samples from GW-1, GW-3, GW-5, GW-6, GW-7, GW-15, and GW-18 have high phosphorus contents. This high content of phosphorus may source from anthropogenic substances taken away by the sewage system (Vanloon and Duffy, 2000).When the same samples are evaluated with respect to the water irrigation criteria (Table 2), most of the samples take place in Class III and Class IV in terms of free chlorine, oil-grease, sulfur, nitrate, nitrite, ammonia, ammonium, and boron contents.

The spatial distribution of the seasonal nitrate and nitrite concentrations of the groundwater in Eskisehir basin are presented in Figure 3. This figure reveals that the highest nitrate concentrations were observed in the central parts of the region where the densely populated and industrialized parts of the city. Also, nitrate concentrations in the wells, in general, are higher than those of water from the Porsuk River. Agricultural impact with the application of nitrogen containing fertilizers such as $(NH_4)_2SO_4$ in irrigation has been mostly associated with nitrate pollution in groundwater.

Table 1. Evaluations of the Analysis Results of water sample according to the TDWS, 1997 (After Yuce et al., 2005)

Sample No	Sampling Date	Boron (mg/l)		Free Chloride (mg/l)		Nitrite (mg/l)		Nitrate (mg/l)		Phenol (mg/l)		SSM (mg/l)		Ammonia (mg/l)		Ammonium (mg/l)	
		Results	RL 1 MCL 2	Results	RL 0.1 MCL 0.5	Results	RL 0 MCL 1	Results	RL 25 MCL 50	Results	RL 0 MCL 5.10^{-4}	Results	RL 0 MCL 1	Results	RL 0 MCL 0.05	Results	RL 0 MCL 0.05
GW-1	22.5.01	2.10	OMCL	0.03	RL	0	RL	39.0	ORL			3.5	OMCL	4.65	OMCL	3.82	OMCL
	18.10.01	0.55	RL	0.02	RL	3	OMCL	26.0	ORL			4.0	OMCL	7.95	OMCL	7.85	OMCL
GW-3	22.5.01	0.70	RL	0.08	RL	4	OMCL	19.0	RL			5.0	OMCL	3.54	OMCL	2.90	OMCL
	19.10.01			0.02	RL	4	OMCL	20.0	RL	0.12	OMCL	9.0	OMCL	2.20	OMCL	2.35	OMCL
GW-5	24.5.01	0.75	RL	0.03	RL	6	OMCL	22.0	RL	0.01	OMCL	4.0	OMCL	3.47	OMCL	2.84	OMCL
	18.10.01			0.01	RL	3	OMCL	15.0	RL	0.00	RL	5.0	OMCL	2.90	OMCL	2.85	OMCL
GW-6	23.5.01	0.90	RL	0.04	RL	1	MCL	23.3	RL			5.0	OMCL	2.74	OMCL	2.17	OMCL
	18.10.01												OMCL	8,06	OMCL	7,98	OMCL
GW-7	23.5.01	0.95	RL	0.05	RL	10	OMCL	56.0	OMCL			1.5	OMCL	3.40	OMCL	2.78	OMCL
	19.10.01	0.39	RL	0.00	RL	4	OMCL	59.0	OMCL			9.0	OMCL	2.45	OMCL	2.46	OMCL
GW-8	23.5.01	0.95	RL	0.05	RL	0	RL	44.0	ORL			4.5	OMCL	4.70	OMCL	3.57	OMCL
	19.10.01	0.45	RL	0.04	RL	3	OMCL	37.0	ORL	0.16	OMCL	5.0	OMCL	3.61	OMCL	3.53	OMCL
GW-9.2	25.5.01	1.35	ORL	0.07	RL	6	OMCL	18.0	RL	0.06	OMCL	3.0	OMCL	4.19	OMCL	3.22	OMCL
	19.10.01	0.65	RL	0.00	RL	4	OMCL	15.0	RL	0.08	OMCL	10.0	OMCL	2.18	OMCL	2.11	OMCL
GW-10	25.5.01	0.50	RL	0.05	RL	0	RL	38.0	ORL			2.0	OMCL	3.32	OMCL	2.71	OMCL
GW-11	25.5.01	0.40	RL	0.06	RL	1	RL	17.0	RL			1.5	OMCL	2.83	OMCL	2.31	OMCL
GW-12	25.5.01	0.60	RL	0.05	RL	2	OMCL	5.3	RL			2.0	OMCL	3.53	OMCL	2.67	OMCL
GW-13	30.5.01	0.75	RL	0.01	RL	3	OMCL	20.0	RL			2.0	OMCL	4.25	OMCL	3.35	OMCL
	19.10.01			0.01	RL	3	OMCL	22.0	RL	7.70	OMCL	3.0	OMCL	6.73	OMCL	7.12	OMCL
GW-15	25.5.01	0.90	RL	0.07	RL	0	RL	15.8	RL			10.5	OMCL	4.36	OMCL	3.42	OMCL
GW-16	25.5.01	1.20	ORL	0.01	RL	3	OMCL	11.0	RL			1.5	OMCL	2.27	OMCL	1.64	OMCL
	19.10.01	0.65	RL	0.02	RL	3	OMCL	21.0	RL			3.0	OMCL	<0.02	ORL	<0.02	ORL
GW-18	25.5.01	0.55	RL	0,01	RL	4	OMCL	20.2	RL			4.5	OMCL	3.31	OMCL	2.6	OMCL
GW-19	25.5.01	0.55	RL	0.02	RL	2	OMCL	42.0	ORL			1.5	OMCL	2.02	OMCL	1.55	OMCL
	19.10.01			0.00	RL	3	OMCL	42.0	ORL	6.70	OMCL	2.0	OMCL	1.01	OMCL	1.07	OMCL
DW-1	19.10.01			0.70	MCL	4	OMCL	9.0	RL	6.60	OMCL	2.0	OMCL	2.76	OMCL	2.92	OMCL
GW-22	19.10.01			0.02	RL	3	OMCL	5.0	RL	0.00	RL	8.0	OMCL	2.06	OMCL	2.19	OMCL

OMCL: Over MCL MCL: Maximum Contamination Level

ORL: Over RL RL: Recommended Level

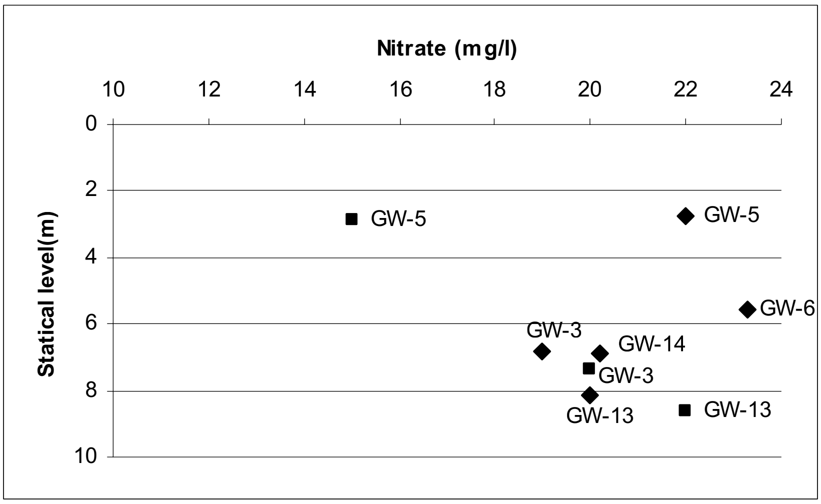

Figure 2. The thickness of unsaturated zone vs. nitrate variations (▪ Dry Period, ◆Wet Period) (After Yuce et al, 2005).

Table 2. Evaluations of the analysis results of water samples according to the TIWS, 1997 (After Yuce et al, 2005)

Sample No	Sampling Date	Boron (mg/l)		Free Chloride (mg/l)		Nitrite (mg/l)		Nitrate (mg/l)		Phenol (mg/l)		SSM (mg/l)		Ammonia (mg/l)		Ammonium (mg/l)		Oil-Gress		Sulfide		COD		TDS		P		BOD	
		Results	Class *	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*	Results	Class*
SW-1	24.05.2001					0.17	IV	3.68	I			20	I											363	I				
	18.10.2001			0.03	II	2	IV	10	I			21	II			3.84	IV			0.02	IV								
SW-2	24.05.2001	5.5	IV	0.09	IV	2	IV	4.4	I			16	I			1.93	III			0.04	IV	44	II	302	I	0.08	II	22	IV
SW-4	24.05.2001	5.5	IV	0.1	IV													3.6	IV	0.08	IV	92	IV	397	I	0.68	IV	46	IV
SW-5	25.05.2001	0.85	II	0.19	IV	0	I	11.0	I			20.5	I			1.84	III	16.4	IV	0.03	IV	31	II	279	I	0.17	III	14	III
SW-6	25.05.2001	1.4	III	0.05	III	0	I	5.3	I			11	I			2.29	IV			0.04	IV	27	II	280	I	0.18	III	23	IV
GW-1	22.05.2001	2.10	IV	0.03	III	0	I	39.0	II			3.5	I	4.65		3.82	IV			0.08	IV	25	I	475	I	0.026	II	11	III
	18.10.2001	0.55	II	0.02	II	3	IV	26.0	II			4.0		7.95		7.85	IV			0.02	IV								
GW-3	22.05.2001	0.70	II	0.08	IV	4	IV	19.0	I			5.0	I	3.54		2.90	IV			0.05	IV	16	I	466	I	0.031	II	7	II
	19.10.2001			0.02	II	4	IV	20.0	I	0.12		9.0		2.20		2.35	III			0.04	IV								
GW-5	24.05.2001	0.75	II	0.03	III	6	IV	22.0	I	0.01		4.0	I	3.47		2.84	IV	23.6	IV	0.06	IV	17	I	375	I	0.034	II	6	II
	18.10.2001			0.01	I	3	IV	15.0	I	0.00		5.0		2.90		2.85	IV	36.2	IV	0.02	IV								
GW-6	23.05.2001	0.90	II	0.04	II	1	IV	23.3	II			5.0	I	2.74		2.17	III			0.06	IV	4	I	597	II	0.03	II	2	I
	18.10.2001						IV							8.06		7.98													
GW-7	23.05.2001	0.95	II	0.05	II	10	IV	56.0	III			1.5	I	3.40		2.78	IV			0.06	IV	21	I	430	I	0.036	II	9	III
	19.10.2001	0.39	I	0.00	I	4	IV	59.0	III			9.0		2.45		2.46	III			0.03	IV								
GW-8	23.05.2001	0.95	II	0.05	II	0	I	44.0	II			4.5	I	4.70		3.57	IV			0.07	IV	22	I	613	II	0.008	I	8	II
	19.10.2001	0.45	I	0.04	I	3	IV	37.0	II	0.16		5.0		3.61		3.53	IV			0.04	IV								
GW-9.1	25.05.2001	1.35	III	0.07	III	6	IV	18.0	I	0.06		3.0	I	4.19		3.22	IV	2.8	IV	0.04	IV	9	I	541	II	0.019	I	8	II
	19.10.2001	0.65	II	0.00	II	4	IV	15.0	I	0.08		10.0		2.18		2.11	III			0.03	IV								
GW-10		0.5	I	0.05	I	0	I	38	II			2	I	3.32		2.71	IV			0.07	IV	23	I	376	I	0.004	I	13	III
GW-11		0.4	I	0.06	I	1	IV	17	I			1.5	I	2.83		2.31	III			0.08	IV	45	I	223	I	0.013	I	20	III
GW-12		0.6	II	0.05	II	2	IV	5.3	I			2	I	3.53		2.67	IV			0.04	IV								
GW-13	30.05.2001	0.75	II	0.01	II	3	IV	20.0	I			2.0	I	4.25		3.35	IV			0.03	IV	13	I	294	I	0.02	I	16	III
	19.10.2001			0.01		3	IV	22.0	I	7.70		3.0		6.73		7.12	IV			0.03	IV								
GW-15		0.9	II	0.07	II	0	I	15.8	I			10.5	I	4.36		3.42	IV			0.05	IV	8	I	575	II	0.018	I	8	II
GW-16	25.05.2001	1.20	III	0.01	III	3	IV	11.0	I			1.5	I	2.27		1.64	III			0.03	IV	9	I	551	II	0.025	II	7	II
	19.10.2001	0.65	II	0.02	II	3	IV	21.0	I			3.0		<0.02		<0.02	I			0	I		I	477	I	0.007	I	3	I
GW-17	25.05.2001	0.85	II	0.19	IV	0	I	11.0	I			20.5	I			1.84	III	16.4	IV	0.03	IV	31	II	279	I	0.17	III	14	III
GW-18		0.55	II	0.01	II	4	IV	20.2	I			4.5	I	3.31		2.6	IV			0.04	IV	4	I	385	I	0.027	II	3	11.0
GW-19	25.05.2001	0.55	II	0.02	II	2	IV	42.0	II			1.5	I	2.02		1.55	III			0.03	IV	9	I	343	I	0.012	I	3	I
	19.10.2001			0.00		3	IV	42.0	II	6.70		2.0		1.01		1.07	I			0	I								
DW-1	19.10.2001			0.70		4		9.0		6.60		2.0		2.76		2.92													

* The class (I- Very good, II- Good, III- Bad, IV-Very bad) is based on Turkish Irrigation Water Standard

Besides intensive usage of fertilizers and pesticides, the high concentration of N and P in groundwater of Porsuk River Basin is sourced from river-groundwater interaction. The river is hydraulically connected to some parts of the unconfined alluvium aquifer. Before 1998, groundwater had been used for irrigation and domestic purposes. Because of the pumping till 1998, the cone of depression intercepted the river, thus polluted river easily affected the groundwater quality. But after municipal water treatment plant has worked, all these wells were stopped, and then water table has risen up to 3-5 m from the ground. However, the river infiltration has still continued due to the high water table conditions, naturally. Seasonal variations may also control the groundwater elevation and thus the direction of flow between the river and aquifer. This relation affects groundwater quality negatively, particularly in dry periods. Thus, during the dry periods, contaminants were easily loaded from the river into groundwater.

The results and evaluation of heavy metal contents in groundwater samples are given in Table 3. Heavy metal contents of most sampled from groundwater was also higher than allowable limits for drinking and domestic usage according to both TDWS and WHO. The sample (GW-19) from a spring located in the discharging area of the municipal waste deposition area (MWDA) has high Pb, Cd, Cr and Mn elements.

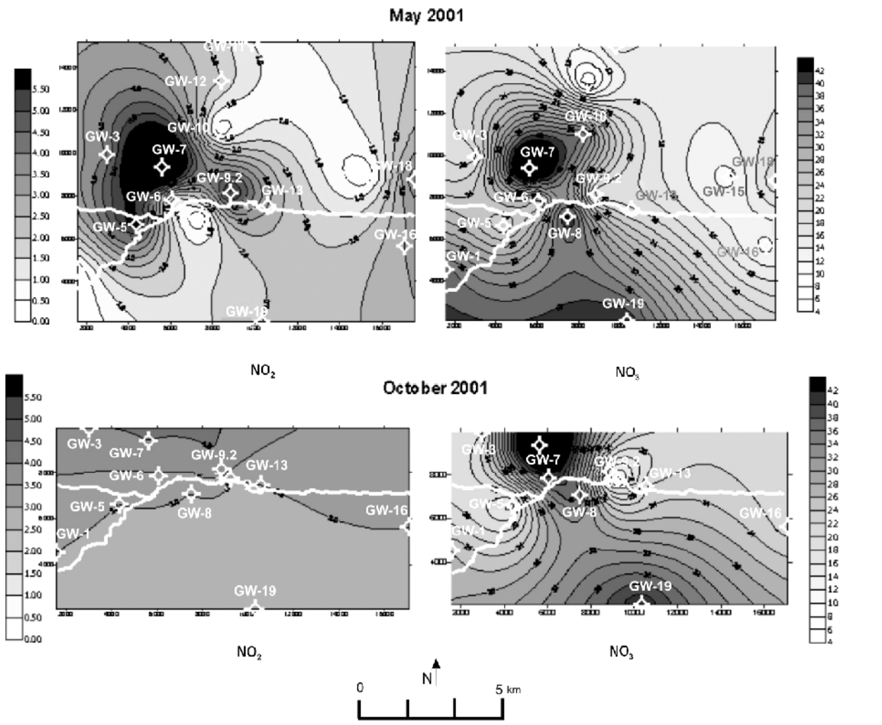

Figure 3. Distribution of nitrite (NO_2) and nitrate (NO_3) in groundwater concentrations from May and Novenber 2001, (contour interval 0.5 mg/l). (After Yuce et al, 2005)

The analyzed pesticide species are chloropyrifos methyl, endosulphan, delthamethrin, lambda-cyalothrin, imidacloprid, tebuconazole, metalaxyl, mancozeb (with Zn and Mn additive), propineb, 1,2,4-D isooctylester, cycloate, HCB, α and β–HCH, heptachlorine, aldrin, dieldrin, pp DDE, pp DDD, pp DDT, op DDT. Pesticides applications in Turkey increased to 45.29% between 1979 and 2002. The percentage of the most toxic pesticides within the total amount is 14.8% (Delen et al., 2005). The op-DDT content in one sample from the Porsuk River is over having 0.02 μg/L according to WHO, 1998 standards (0.002 μg/L). Pesticides values in groundwater sampled from GW-5 and GW-19 were not suitable in terms of β–endosulphan with 0.2 μg/L and in terms of 2, 4 D with 3.8 μg/L according to the European Union Standards (EU, 1998 which accepts the limits 0.1 μg/L for all types of pesticides), respectively. However those values were not over according to EPA, 2003 which accept allowable limits 74 μg/L for β–endosulphan and for 70 μg/L 2, 4 D. The high content of Zn in GW-3; GW-13, GW-14; GW-16 and GW-19 may be from mancozeb and propineb types of pesticides.

Table 3. Evaluations of Heavy Metals Analysis Results* of GW Samples (After Yuce et a.l, 2005)

Sample No	Sampling Date	Total Zinc	Total Lead	Total Crom	Total Manganese	Total Iron	Total Copper	Total Mercury	Total Cadmium	Total Aluminium	Total Arsenic
RL		*0.1*	*0.01*	*0*	*0.02*	*0.05*	*0.1*	*0*	*0*	*0.05*	*0*
MCL		**5**	**0.05**	**0.05**	**0.05**	**0.2**	**3**	**0.001**	**0.005**	**0.2**	**0.05**
GW-1	26.05.2001	0.034	<0.005	0.017	<0.005	0.04	0.011	<0.005	<0.005		
	18.10.2001	0.042	<0.005	<0.005	<0.005	0.02	<0.005	<0.0005	<0.005	*0.158*	<0.003
GW-2	25.05.2001	0.045	<0.005	0.021	<0.005	0.02	0.009	<0.005	<0.005		
GW-3	25.05.2001	0.019	<0.005	0.005	<0.005	*0.12*	0.009	<0.005	<0.005		
	19.10.2001	*0.321*	*0.014*	0.005	0.008	*0.19*	0.006	<0.0005	<0.005	*0.187*	<0.003
GW-4	26.05.2001	0.019	<0.005	0.012	<0.005	<0.05	0.009	<0.005	<0.005		
GW-5	25.05.2001	0.023	<0.005	<0.005	<0.005	0.02	0.007	<0.005	<0.005		
	18.10.2001	0.087	*0.013*	0.007	<0.005	<0.05	<0.005	<0.0005	<0.005	**0.308**	<0.003
GW-6	25.05.2001	0.021	*0.012*	<0.005	<0.005	0.03	0.013	<0.005	**0.007**		
	18.10.2001	0.044	*0.028*	<0.005	<0.005	<0.05	<0.005	<0.0005	<0.005	0.02	<0.003
GW-7	27.05.2001	0.045	<0.005	<0.005	<0.005	0.05	0.024	<0.005	0.005		
GW-8	26.05.2001	0.023	*0.017*	*0.021*	<0.005	**0.34**	0.008	<0.005	<0.005		
	19.10.2001	*0.107*	*0.014*	*0.006*	0.009	**0.6**	<0.005	<0.0005	<0.005	*0.092*	<0.003
GW-9.1	26.05.2001	0.026	*0.035*	*0.014*	<0.005	0.04	0.011	<0.005	0.005		
GW-9.2	25.05.2001	0.012	*0.023*	*0.006*	**0.367**	0.03	0.022	<0.005	<0.005		
	19.10.2001	0.062	0.007	<0.005	*0.037*	0.03	<0.005	<0.0005	<0.005	*0.105*	<0.003
GW-9.3	25.05.2001	0.024	*0.019*	*0.017*	**0.086**	0.03	<0.005	<0.005	<0.005		
	19.10.2001	0.081	*0.011*	<0.005	**0.323**	*0.06*	<0.005	<0.0005	<0.005	*0.174*	<0.003
GW-10	25.05.2001	0.039	*0.013*	*0.012*	<0.005	0.05	0.01	<0.005	<0.005		
GW-11	25.05.2001	0.029	*0.014*	*0.028*	<0.005	0.04	0.017	<0.005	<0.005		
GW-12	27.05.2001	0.074	<0.005	*0.017*	<0.005	0.05	<0.005	<0.005	<0.005		
GW-13	26.05.2001	0.029	*0.022*	**0.083**	**0.093**	0.04	<0.005	<0.005	<0.005		
	19.10.2001	*0.124*	0.009	<0.005	**0.105**	*0.07*	<0.005	<0.0005	<0.005	0.031	<0.003
GW-14	27.05.2001	*0.184*	*0.017*	<0.005	<0.005	*0.15*	<0.005	<0.005	<0.005		
	19.10.2001	*0.71*	0.01	*0.005*	<0.005	*0.12*	<0.005	<0.0005	<0.005	*0.097*	<0.003
GW-15	25.05.2001	0.012	*0.021*	<0.005	**0.487**	0.05	<0.005	<0.005	<0.005		
GW-16	26.05.2001	0.01	*0.02*	<0.005	<0.005	0.03	<0.005	<0.005	<0.005		
	19.10.2001	*0.297*	*0.038*	<0.005	**0.053**	**1.34**	*0.219*	<0.0005	<0.005	*0.134*	<0.003
GW-17	25.05.2001	0.013	*0.017*	<0.005	<0.005	*0.08*	<0.005	<0.005	<0.005		
GW-19	27.05.2001	0.031	<0.005	*0.017*	<0.005	0.05	<0.005	<0.005	**0.009**		
	19.10.2001	*0.315*	0.006	<0.005	<0.005	0.05	<0.005	<0.0005	<0.005	*0.139*	<0.003
GW-20	26.05.2001	*0.827*	*0.023*	*0.025*	**0.614**	**5.69**	*0.108*	<0.005	<0.005		
GW-21	19.10.2001	0.044	0.005	<0.005	0.015	**0.21**	<0.005	<0.0005	<0.005	**0.325**	<0.003
GW-22	19.10.2001	0.031	<0.005	<0.005	<0.005	<0.05	<0.005	<0.0005	<0.005	*0.118*	<0.003

Bolds are represented as over the MCL (Maximum Contamination Level).

italics are represented as over the RL (Recommended Level)

According to Turkish Drinking Water Standard (Code No: TS/266)

* All results are presented in mg/L.

4.2. SURFACEWATER (SW)

The classification of water samples taken from the various sections of Porsuk River and its irrigation canals are given in Table 2. According to Table 2, the usability for irrigation purposes of the Porsuk River and its irrigation canals are not appropriate. The possible contamination sources of the Porsuk River are shown in Figure 4. Especially in Kutahya part, the wastewater of the Sugar Factory is discharged to the Felent River. The slaughterhouse of Kütahya city also discharges its sewage water to the Felent River from its insufficient wastewater treatment plant. The discharges of the wastewaters from many factories and especially from Seyitömer Thermal Electric Power Plant contribute to the pollution of the Porsuk River. The polluted waters reach the Porsuk Dam Reservoir. The existence of high nitrogen and phosphorus at the outlet of Porsuk Dam Reservoir (PDR) are sourced from decomposed organic substances accumulated in the hypolimnion zone of the reservoir (Drever, 1997). PDR experiences seasonal high blooms of algae because of this excessive nitrogen and phosphorus loads (Muhammetoglu et al., 2005). Large algal blooms are the result of eutrophication which decreases dissolved oxygen levels in the water and in turn decreases aquatic vegetation, harms wildlife, damages water supplies, decreases the aesthetic value (Boesch 2000). Water quality of Porsuk River is relatively good between the outlet of dam and the sampling point SW-VI (sugar factory) apart from nitrogen and phosphorus. However, from this point (SW-IV) the quality of the Porsuk River water becomes Class IV for irrigation water according to TIWS (1997) due to the insufficient conditions of the wastewater treatment plant of the Sugar Factory and other factories. This situation conforms to the increase in NH_3, NH_4, phenol, NO_2, B, SSM, free chlorine and Mn values in the samples taken from GW-9.2 well in the Sugar Factory. The water quality of the river worsens at the sampling point SW-VI because of discharged wastewaters from slaughterhouse and The Air Force and Maintain Center. The results and evaluation of heavy metal contents of surface water samples are given in Table 4.

As can be seen on this Table, Mn ad Fe contaminations are over the maximum limits for drinking water most of the sampling points. The comparison of the analysis results of the water samples of Porsuk River collected from the previous (EGDE, 1999) and this study are given in Figure 5.

Table 4. Evaluations of Heavy Metals Analysis Results* of Surface Water Samples (After Yuce et al., 2005)

Sample No	Sampling Date	Total Zinc	Total Lead	Total Crom	Total Manganese	Total Iron	Total Copper	Total Mercury	Total Cadmium	Total Aluminium	Total Arsenic
RL		*0.1*	*0.01*	*0*	*0.02*	*0.05*	*0.1*	*0*	*0*	*0.05*	*0*
MCL		**5**	**0.05**	**0.05**	**0.05**	**0.2**	**3**	**0.001**	**0.005**	**0.2**	**0.05**
SW-1	25.5.01	0.017	*0.017*	<0.005	*0.039*	*0.19*	0.08	<0.005	0.005		
	18.10.01	0.06	*0.012*	<0.005	**0.054**	**0.27**	<0.005	<0.0005	<0.005	**0.319**	<0.003
SW-2	25.5.01	0.032	<0.005	<0.005	**0.063**	*0.1*	0.013	<0.005	0.005		
SW-3	27.5.01	0.065	<0.005	0.024	*0.027*	**0.32**	0.015	<0.005	<0.005		
	19.10.01	0.1015	*0.013*	<0.005	0.014	*0.15*	<0.005	<0.0005	<0.005	**0.219**	<0.003
SW-4	19.10.01	*0.325*	*0.016*	0.012	**0.108**	**0.47**	0.013	<0.0005	<0.005	**0.448**	<0.003
SW-7	25.5.01	0.053	**0.055**	**0.083**	**0.141**	0.05	0.015	<0.005	**0.016**		
SW-III	19.10.01	0.075	0.008	<0.005	0.008	<0.05	<0.005	<0.0005	<0.005	*0.142*	<0.003
SW-VI	19.10.01	0.075	*0.011*	<0.005	**0.08**	**0.37**	0.01	<0.0005	<0.005	**0.343**	<0.003
SW-VII	19.10.01	0.039	0.009	<0.005	**0.07**	*0.19*	0.01	<0.0005	<0.005	**0.298**	<0.003
DW-1	19.10.01	*0.292*	0.006	<0.005	0.006	<0.05	<0.005	<0.0005	<0.005	*0.097*	<0.003

Bolds are represented as over the MCL (Maximum Contamination Level). Italics are represented as over the RL (Recommended Level). According to Turkish Drinking Water Standarts (Code No: TS/266).
* All results are presented in mg/L. (Yuce et al., 2005)

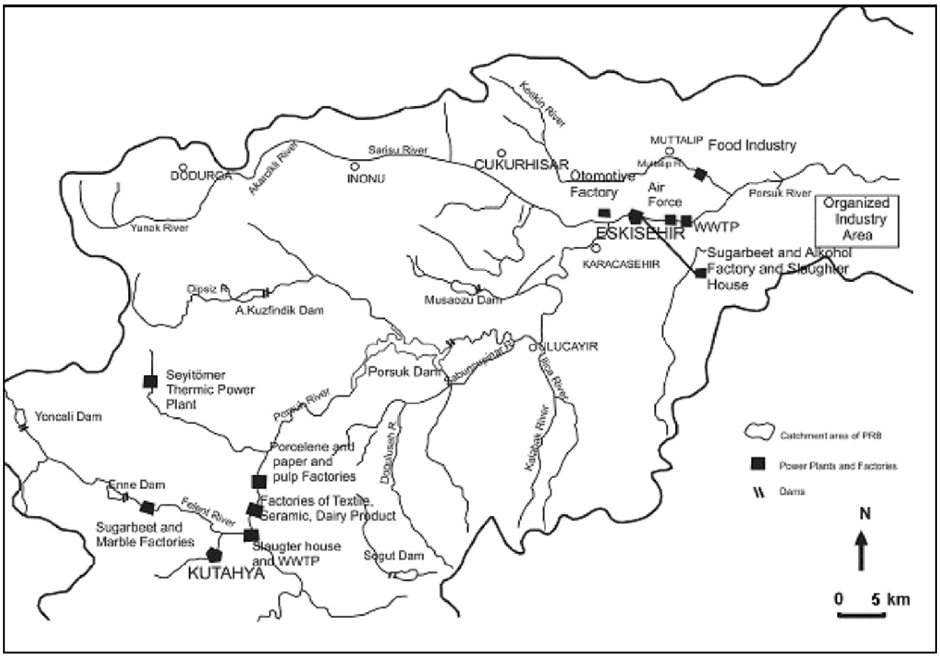

Figure 4. Plants which discharge their waste water to Porsuk River in Kutahya and Eskisehir Region (After Yuce et al., 2005).

An increase in COD and BOD are observed when old and new water analyses results of Porsuk River are compared. This increase is mainly due to the contribution of the contamination loads discharging from the Sugar Factory and the “Organized Industrial Area” as observed in sections SW-VI, SW-VII and SW-VIII. The increases of phenolic substances, boron, free chlorine and sulfur in section SW-VIII indicate the probability of the reductive environment because of the sewage. Detailed investigation shows that at the sewage-polluted sites, oligochaetes (especially Tubifex tubifex and Limnodrilus hoffmeisteri) have been seen in remarkable abundance (Malard et al., 1996; Kazanci and Girgin, 1998; Arslan et al., 2003). Low DO and the high abundance of *L. hoffmeisteri* in Porsuk River along the vicinity of Kutahya Nitrogen Factory and Suger-beet Factory reflects organic enrichment associated with fecal contamination and can be related to frequent wastewater discharged by the industrial organizations in this area (Fig. 4). The maximum fecal coliform concentration in Porsuk River was measured in Kutahya Nitrogen Factory and Organized Industrial District in Eskisehir (Arslan et al., 2003).

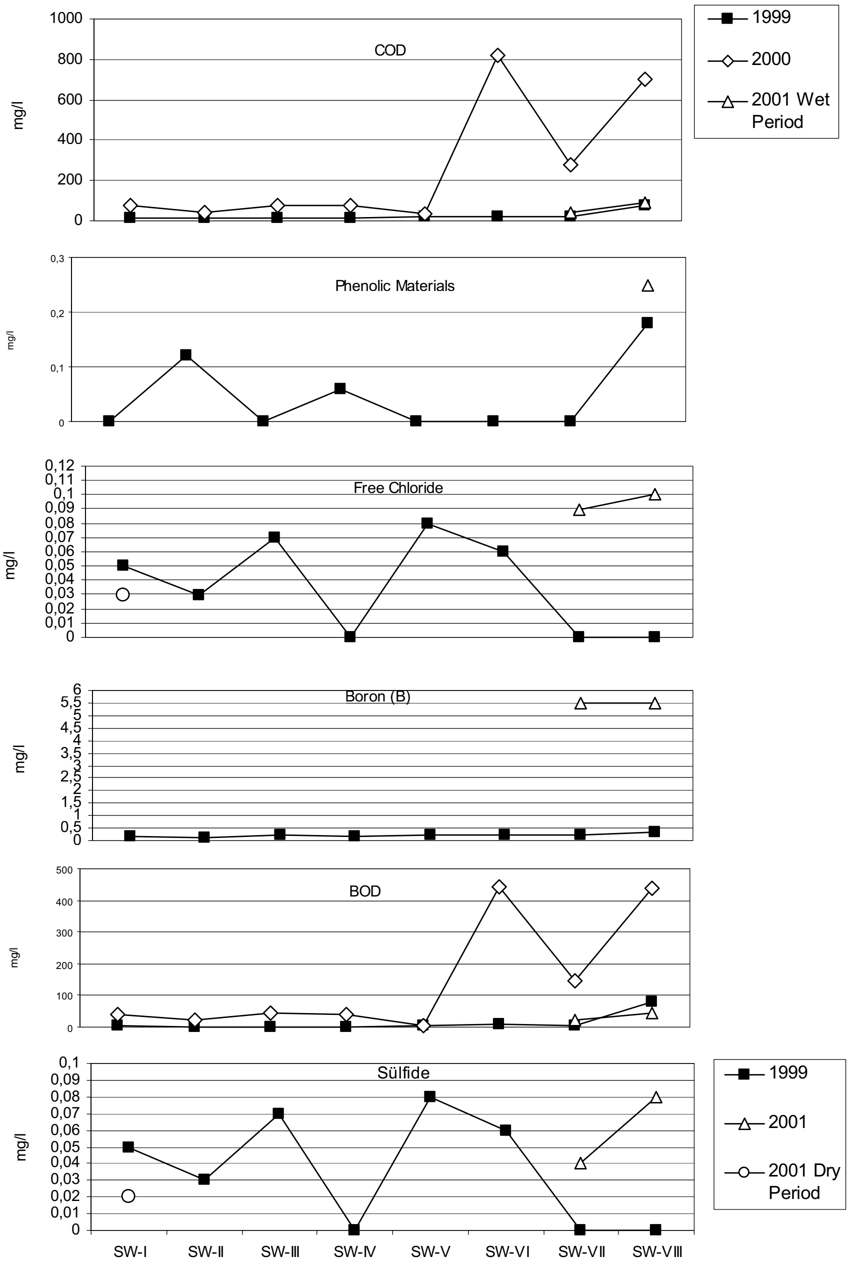

Figure 5. Analyses Results of Samples Taken At Different Times From The Different Locations At Porsuk River (Roman numbers represent Sampling Points on Porsuk River Section) (modified from Yuce et al., 2005).

Phenol content of the treated water from Porsuk River has highly exceeded the allowable limits of TDWS. In spite of the fact that, phenol is an extremely toxic material, it is widely used in agriculture and pesticides (Ren, 2002) and in various industries such as coal conversion, metal casting, paper manufacturing, and resin production. Incomplete combustion of 2,4,5-trichlorophenol may cause the formation of TCDD (dioxin). High phenol content may be sourced from the chlorification and chlorate pesticides.

Another problem in the treated water from Porsuk River is high AOX (Adsorbable Organic Halogene) caused by the intensive chlorification of water (Barber et al., 1995) or the bleaching process that has been used by Porcelain factories located in the catchment area of Porsuk River. AOX is a measure of dissolved chlorinated matter, including both volatile and nonvolatile compounds, some of which may be toxic. The concentrations of AOX in the Porsuk River ten times exceeded the allowable limit (0.25 mg/l according to TDWS, 1997). The chlorination of drinking water has been the most common method of purifying water. The small quantities of unpurged organic material, especially humic substances, react rapidly with chlorine during the purification process to form many volatile halogenated organic compounds. Among them are THM trihalomethanes, which were first recognized as a product of the chlorination process. THM are carcinogenic to humans (Galapate et al., 1999). Porsuk River water after the Treatment Plant should be analyzed in terms of THM content.

The municipal waste deposition site (MWDA) is located on Takahasan valley, mainly consists of calcium carbonate rocks, near Eskisehir-Seyitgazi highway. The carbonate layers are semipermeable. The leachate from the MWDA is rich in heavy metals, organic substances, bacteria and salts (Bakis, 1996) that may decompose readily by chemical and biochemical ways infiltrates to surface and to groundwater, causing one of the most important pollution factors. The discharge of leachate high in ammonia into a river exerts an oxygen demand on the receiving water. In addition, ammonia is toxic to fish (lethal concentrations 2.5-25 mg/L) and, as ammonia is a fertilizer, it may alter the ecology of the river. Besides this, the leachate contained Pb, Cr^{6+}, Mn, Cd, and Zn in high levels.

5. Conclusions

The NH_3, Oil and grease, COD, BOD, phenolic material, free chlorine and sulfur values of Porsuk River have increased compared to those of previous years and become worse with the wastewater loadings from Sugar Factory and Organized Industrial Area. In the dry period, the water qualities grow worse compared to wet period. Most of the groundwater samples are not eligible for drinking according to the both TDWS and EPA standards with respect to NO_3 and NO_2 values. NO_3, NO2, boron, NH_4, S, phenol, ammonia, ammonium, free chlorine, COD, organic nitrogen, oil and grease, sulfur, and phosphorus in the waters of PRB are higher than the allowable drinking waters limits in general. Heavy metal contents (Al, Cr^{6+}, Mn, Fe, Cd) of groundwater in the study area exceed drinking water limits. In surface water samples from Porsuk River, heavy metal contents increase towards downstream associated with the wastewater contamination from Sugar Factory and Organized Industrial Area. Cadmium value is high in sample GW-19 (a spring at discharged area of the MWDA). The increase in heavy metal contents downstream of Porsuk River sourced from the industrial activities and discharge of their wastewater directly to the river. Concerning pesticides pollution, no values are over the limits according to EPA. However, two of the samples from groundwater and one from Porsuk River have over the limits according to the EU and WHO standards for drinking water.

As a conclusion, N and P daughters and heavy metal contents are continuously increasing in groundwater of the Eskisehir plain and Porsuk River according to the comparison of the recent and previous analyses results. River-groundwater interaction plays an important role on this contamination, besides intensive usage of fertilizers and pesticides.

6. Recommendations

In order to sustain industrial development with a minimum of environmental pollution, it is necessary to make a rational classification of industries based on the type of pollutants released and the magnitude of the pollution potential of each. Some of the measures could be:

- A systematic sampling and analyzing program should be implemented in the area and precautions should be taken for all kinds of pollution.
- Stringent measures should be adopted for strict adherence to environmental protection laws.
- Water treatment plant, municipal wastewater treatment plant, as well as Sugar Factory and the other treatment systems should be rehabilitated and direct discharge of wastewaters to the Porsuk River must be stopped.
- The effect of the municipal waste deposition should be investigated in detail and precautions should be taken to rehabilitate the depositional conditions.

- It is necessary to build up a realistic model for the optimal water usage in the Porsuk River Basin taking into account groundwater quality deterioration.
- Environmental isotopes and tracer techniques should be used in the investigations, which will provide additional data to evaluate surface and groundwater interactions.

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