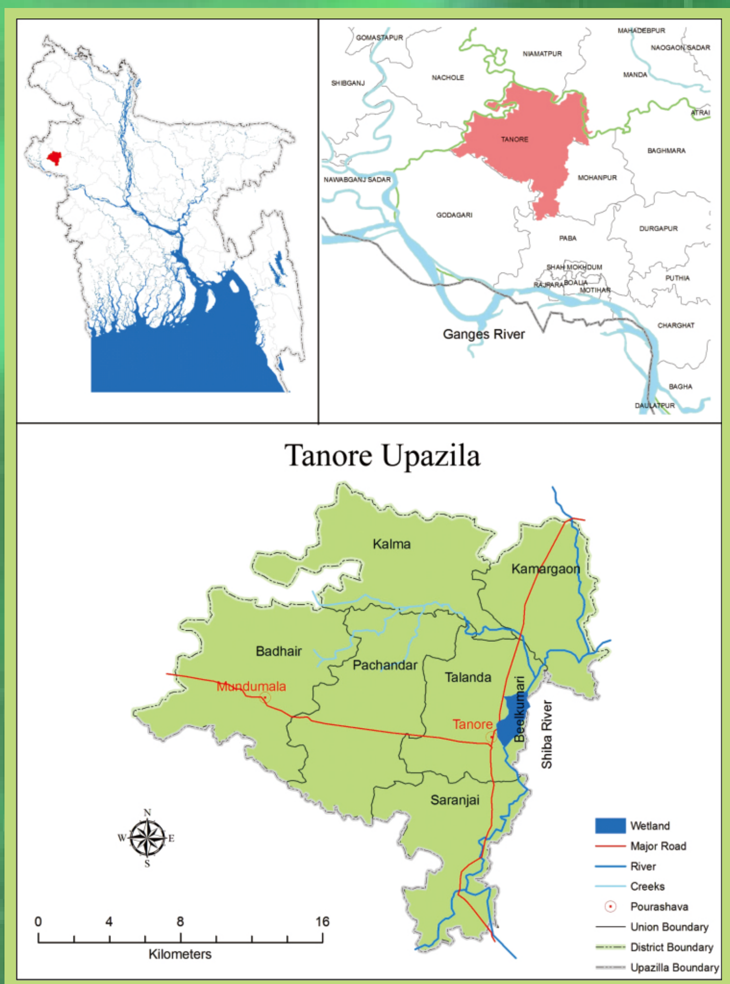
ISSN: 1945-3094 Vol. 4, No. 8, August 2012

Scientific Research

Journal of Water Resource and Protection

Editors-in-Chief : Jian Shen Ni-Bin Chang



ISSN: 1945-3094

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Journal Editorial Board

ISSN: 1945-3094 (Print)   ISSN: 1945-3108 (Online)
http://www.scirp.org/journal/jwarp

Journal of Water Resource and Protection, **2012, 4, 567-693**
Published Online August 2012 in SciRes (http://www.SciRP.org/journal/jwarp/)

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Volume 4 Number 8 August 2012

The figure on the front cover is from the article published in Journal of Water Resource and Protection, 2012, Vol. 4, No. 8, pp. 567-575 by Md. Marufur Rahman and A. Q. M. Mahbub.

Journal of Water Resource and Protection (JWARP)

Journal Information

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Journal of Water Resource and Protection, **2012, 4, 567-575**
doi:10.4236/jwarp.2012.48066 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Groundwater Depletion with Expansion of Irrigation in Barind Tract: A Case Study of Tanore Upazila

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Received May 12, 2012; revised June 22, 2012; accepted June 30, 2012

ABSTRACT

In this paper successive depletion of groundwater level with expansion of groundwater irrigation in Barind Tract has been discussed from mid 1960s to 2010 in the context of Tanore Upazila, which is located in severely drought prone area of northwest Bangladesh. After starting of groundwater irrigation in Bangladesh, it spread rapidly all over the country, and about 80% of agricultural land is now supplied irrigation from groundwater. Availability of irrigation in Barind Tract has revolutionized its agriculture, but groundwater level is successively falling all over the country due to excessive withdrawal, and this process is accelerating due to water withdrawal from major rivers by upstream countries. In northwestern part of Bangladesh groundwater depletion problem is severe because this part is free from seasonal flooding. Only source of recharging of groundwater aquifer in this area is rainfall, but rainfall is also lowest here among the country. In this context, this paper presents the change of groundwater level with the spreading of groundwater irrigation in Barind Tract. Hydrograph analysis, groundwater level mapping, groundwater depletion rate calculation are done from groundwater level observation well data of Bangladesh Water Development Board (BWDB) and Barind Multipurpose Development Authority (BMDA). Climatic condition is analyzed by calculation of rainfall deviation from the data of Bangladesh Meteorological Department (BMD). Focus Group Discussion (FGD) and interviews with farmers and experts of different branches are conducted to understand the nature of problems in the study area. Agricultural pattern, cropping intensity (262% in study area and national intensity is 180%), methods of cultivation, crop variety and yields all show a positive change after starting of groundwater irrigation in mid 1980s, but water level is continuously lowering at the rate of 1.37 ft/y in wet season and 0.72 ft/y in dry season. Water is the main input for agriculture but successive depletion of groundwater level can be a serious problem for water stressed Barind Tract. Crop diversification, artificial recharging, increasing dependency on surface water, increasing irrigation efficiency, rainwater harvesting etc., can be option for the area.

Keywords: Groundwater Depletion; Irrigation; Barind Tract; Tanore

1. Introduction

Globally, irrigation is responsible for more than 65% of all fresh water withdrawals. At present, one quarter of world's irrigated land is supplied by groundwater and 75% of these lands are located in Asia [1]. Agriculture in Bangladesh was entirely dependent on surface water and monsoon rainfall prior to 1970s [2]. Now in Bangladesh 79.1% lands are supplied water in boro season from underground source [3]. Agricultural land in Bangladesh was irrigated by traditional means up to 1950s without any institutional base and was institutionalized with the formation of East Pakistan Water and Power Development Authority (Now, BWDB) in 1959 [4]. Bangladesh Agricultural Development Corporation (BADC), the then East Pakistan Agricultural Development Corporation (EPADC) was created in 1961 and act as the main organization for the expansion of both groundwater and surface water irrigation [5].

Barind Tract is a physiographic unit located in northwestern part of Bangladesh having gross area of 7727 sq km [6]. Geographically this unit lies between 24˚20'N and 25˚35'N latitudes and 88˚20'E and 89˚30'E longitudes. Barind Tract made up of Pleistocene Alluvium also known as Older Alluvium and floored by reddish brown, sticky Pleistocene sediment; Madhupur Clay [7]. Pleistocene Dupi Tila Sand act as aquifer in Barind Tract [8]. Barind Tract was excluded during 3000 Deep Tube Well (DTW) installation programme of BADC in North-west Irrigation Project considering as low potential area for groundwater development [9]. Groundwater development occurs in Barind with the formation of Barind Integrated Area Development Project (BIADP) in 1985 under BADC and later with formation of Barind Multipurpose Development Authority (BMDA) in 1992 [10].

*Corresponding author.

Tanore is an upazila of Rajshahi District located in northwest Bangladesh (**Figure 1**). This upazila is located between 24˚28'N and 24˚44'N latitudes and between 88˚24'E and 88˚39'E longitudes. Physiographically Tanore consist of Barind Tract (81.8%), Old Gangetic Floodplain (3%) and Tista Floodplain (4.8%) others including homestead, wetland, ponds, river (10.4%) in respect of total area [11]. Textural class of soil is Clay loam 46%, Loam 35%, and Clay 8% [12]. With the expansion of irrigation in Tanore Upazila from mid 1980s, a revolutionary change is occurred in its agricultural sector. Introduction of High Yielding Varieties (HYV) paddy with transforming from broadcasted to transplanted cultivation, increase of yields all bring by groundwater irrigation. Single cropped agricultural land now producing three crops in one agricultural year after starting of groundwater irrigation. Cropping intensity of Tanore is now 262%, much higher than national cropping intensity of Bangladesh 180% [13].

Groundwater recharging in Bangladesh is mainly occur by monsoon rainfall and flooding. Due to high elevation of Barind, it is located in flood free zone. So, only source of groundwater recharging in this area is rainfall, but lowest amount of rainfall occur in northwestern part of Bangladesh and it is also a very severely drought prone area. Moreover, thick sticky clay surface of Barind

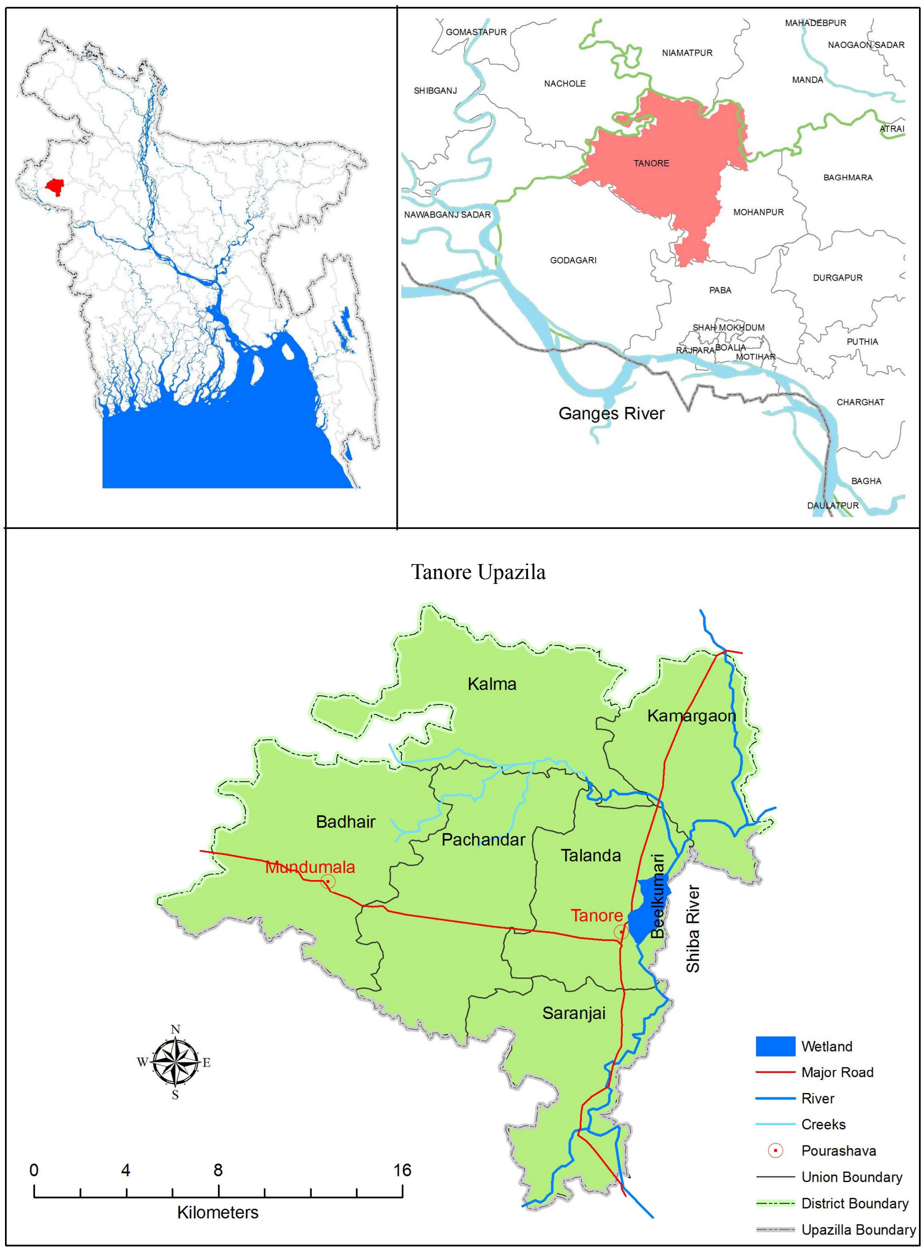

Figure 1. Location map of study area.

Tract act as aquitard which impede groundwater recharging and increase surface runoff. As a result, groundwater level in this part is successively falling by years with increasing withdrawal of water for irrigation.

2. Methods

Secondary data is mainly used for this study. The study is based on groundwater monitoring piezometer data of Bangladesh Water Development Board (BWDB) from 1966-2010 and monitoring well of Barind Multipurpose Development Authority 1986-2010 (BMDA) (see **Figure 2**). Rainfall data is collected from Bangladesh Meteorological Department (BMD). Lithology of the study area is studied from borehole logs collected from BMDA.

Fourteen piezometer data in and around Tanore Upazila are used for mapping of contour elevation of groundwater level of study area. Groundwater levels are referenced to a common datum (Public Works Datum, PWD) which was originally set to the mean sea level (msl) with a vertical error of ±0.45 m during the Great Trigonometric Survey in the Indian Subcontinent throughout the nineteenth century [1]. Groundwater depletion rate is calculated from the data of BMDA and BWDB monitoring well. Mapping software ArcGIS 9.3.1 is used for mapping. Personal interview with different expert groups and Focus Group Discussion (FGD) in the study area with the local people are conducted to understand the nature of the problem.

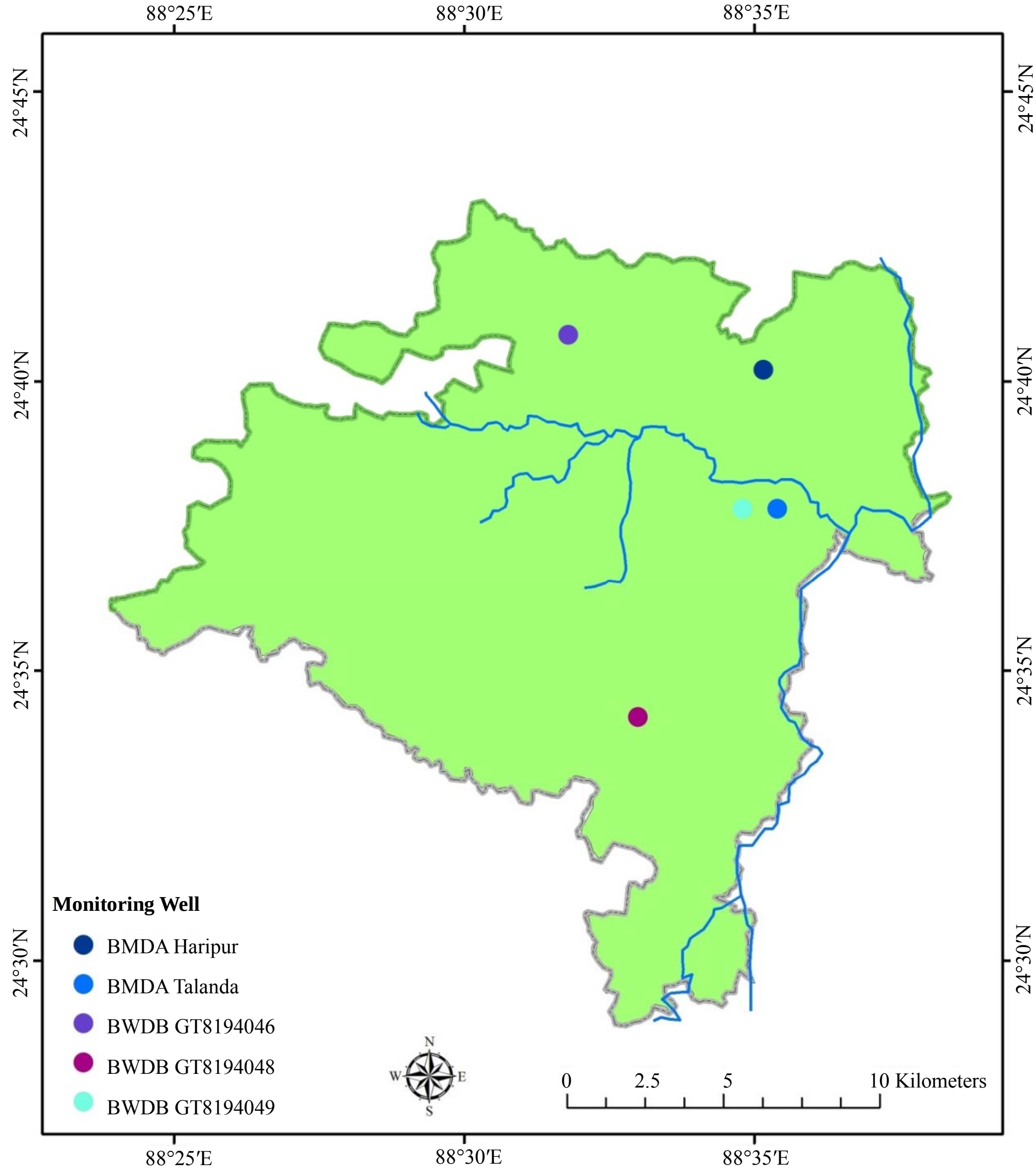

Figure 2. Location of groundwater level monitoring well in Tanore Upazila.

3. Result and Discussion

3.1. Hydrograph Analysis

Long term groundwater level trend is observed by analysis of hydrograph from groundwater observation well data. For analysis of long term trends of groundwater level of Tanore Upazila, two observation wells of BWDB and two observation wells of BMDA are used. The data of BWDB's wells used from 1966-2010 and data of BMDA wells from 1987-2010. All four observation wells of two organizations of Tanore Upazila are showing a successive depleting trend of groundwater level over time. In Bangladesh groundwater level rise in wet season (due to monsoon rain and flooding in main river) and reach in maximum level in August to September and after the wet season it start to fall and reach in minimum level in the pre-monsoon months of April to May. In hydrograph with seasonal fluctuation the declining trend of groundwater level in the study area is very clear.

The elevation of study area is increase from eastern to western side and the thickness of clay and depth of aquifer is also show same nature as a result depth of groundwater level is vary from eastern to western side. Groundwater level found in maximum depth from the surface ground in western side and minimum depth in eastern side, but it is clear that in all area groundwater table is continuously going down or elevation of groundwater level from mean sea level is continuously decreasing although still it found above the mean sea level (msl) (see **Figure 3)**.

Kamargaon observation well of BMDA is located in

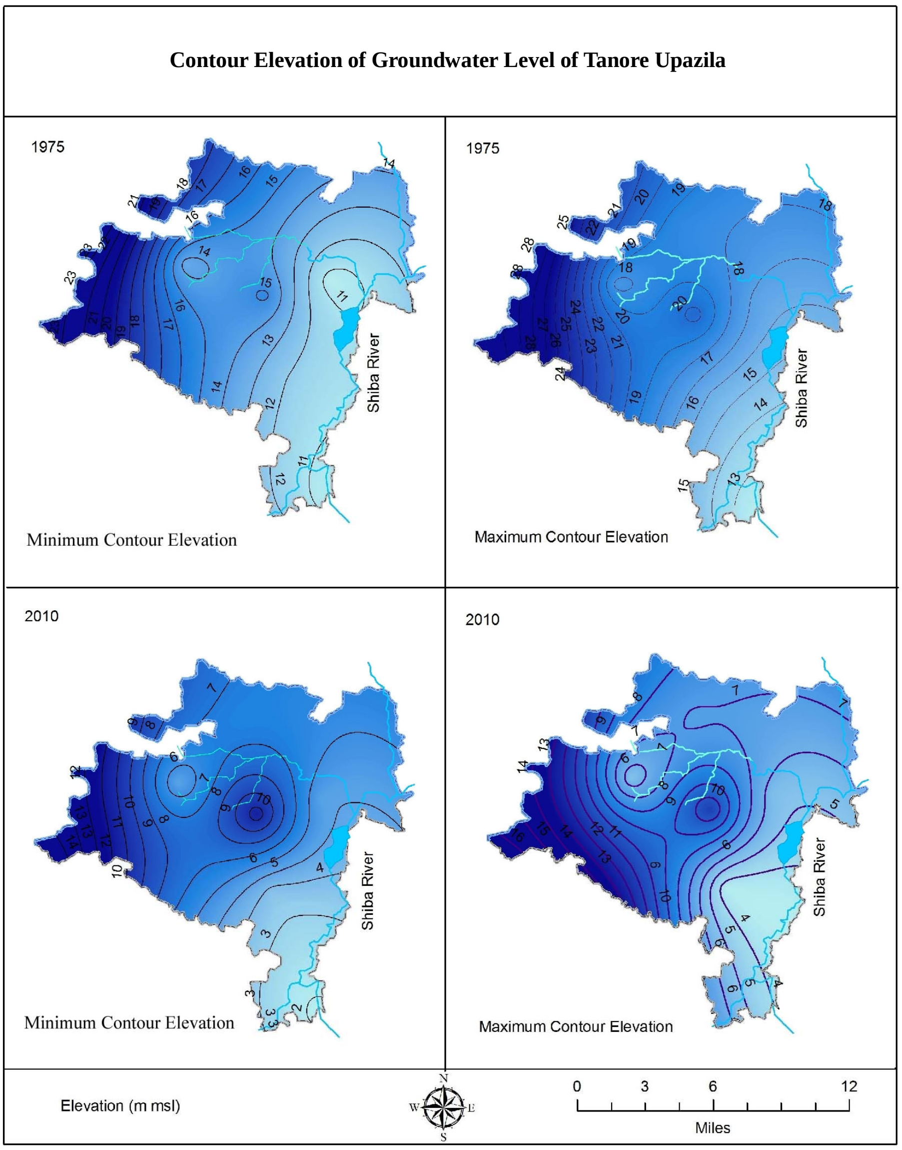

Figure 3. Contour elevation of groundwater level of Tanore Upazila. Data source: BWDB.

Haripur of Kamargaon Union where clay thickness is about 25 - 30 ft and sand aquifer located below 20 -30 ft from surface. Hydrograph of Kamargaon Union showing groundwater level fluctuation from 1986-2010. Data is collected in fifteen days interval in 15th and last day of month. Hydrograph in **Figure 4** depict both seasonal fluctuation and long term trends of water level in the area. In 1986 seasonal fluctuation occur in between 5 - 23 ft which stands in 29 - 54 ft in the year of 2009. The difference between maximum and minimum water level in one season was 2.67 ft (2010). Hydrograph show a declining trend of groundwater level which starts to decline in a rapid rate after 2002.

Talanda observation well of BMDA is in Talanda Mouza and located very close to River Shiba and Beelkumari. Hydrograph in **Figure 5** depict the groundwater level of Talanda from fifteen July 1986 to thirty June 2011. Interval of data collection is similar to Haripur observation well of BMDA and it is fifteen days and in fifteenth and last day of month, twenty four observation in a year. Clay thickness and aquifer depth are varied in Talanda Union. Thickness of clay layer of observation well is about 20 ft and below 20 ft from surface sand aquifer is located. Hydrograph in **Figure 5** present both long term trend of groundwater level and seasonal fluctuation. In 1987 seasonal fluctuation of groundwater

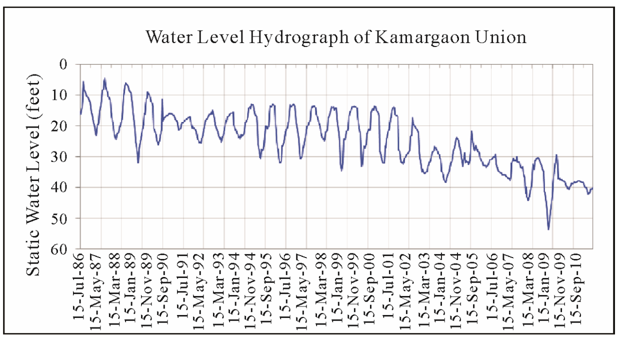

Figure 4. Water level hydrograph of Kamargaon Union, Tanore. Data source: BMDA (Mouza: Haripur, J.L. No-236, Plot No-211).

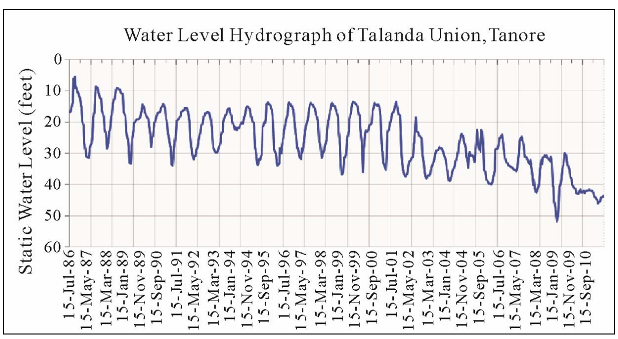

Figure 5. Water level hydrograph of Talanda Union, Tanore. Data source: BMDA (Mouza: Talanda, J.L. No-224, Plot No-1138).

occur 8.50 - 31.54 ft and it standed 41.67 - 42.83 ft in 2010. Fluctuation of water level in 2010 was very small because low rainfall in wet season. Recorded rainfall from BMD's Rajshahi station was only 792 mm.

Two observation well of BWDB is used in hydrograph analysis of groundwater level of Tanore Upazila. New well ID of BWDB well is GT8194046 and GT8194048 and old ID was RJ039 and RJ086 respectively. Reading interval and observation well type of BWDB both is different from BMDA observation well. Reading interval is seven days in BWDB well. In collected data reading is taken every week but not in a fixed date like BMDA and data missing is a common problem. BWDB wells are used only for groundwater level monitoring purpose, whereas BMDA wells are used both for irrigation and water level monitoring. Static water level is collected from BMDA wells for monitoring of groundwater level. Unit of BWDB's collected data is meter, whereas BMDA collect data in feet.

From 14th February 1966 to 18th October of 2010 data of well GT8194046 is used for hydrograph analysis. From 14th June 1971 to 1972 data of this well was not available, and the reason is easily understandable. The War of Liberation occurs in 1971, as a result data collection was interrupted and it was again started from first January 1973. Prior reading was collected from dug well and later it was replaced by piezometer. Hydrograph in **Figure 6** show that groundwater level fluctuation was almost stable till 1978 and after 1978 fluctuation level is increased but after 1983 the groundwater level decline rapidly and the sharp negative trend is observed till present.

Data from 12th December 1966 to 11th October 2010 of BWDB's well GT8194048 is used in hydrograph of **Figure 7**. Data was unavailable from 5th April 1971 to October of 1972 due to Liberation War and from November 1972 data collection was again started. After 1980 depletion trend is observed and after 1994 groundwater level deplete rapidly. Spread of irrigation in Tanore Upazila occurs in 1980s which is the main factor of groundwater level depletion.

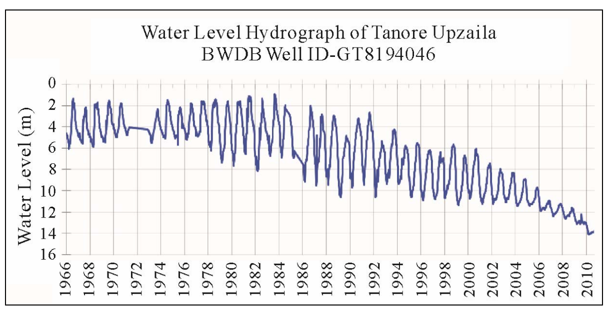

Figure 6. Water level hydrograph of Tanore Upazila. Data source: BWDB.

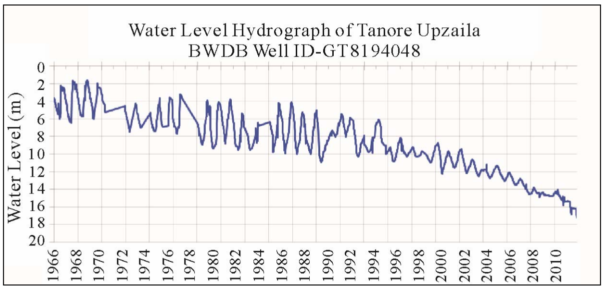

Figure 7. Water level hydrograph of Tanore Upazila. Data source: BWDB.

3.2. Groundwater Development and Water Level

Groundwater irrigation was started in Tanore Upazila in 1970s and spread after formation of Barind Integrated Area Development Project (BIADP) in 1985 [10]. First time, irrigation started in eastern part (see **Figure 8**) and western part was avoided due to high depth of aquifer (70 - 90 ft below the surface), later irrigation spread all parts of upazila with the formation of Barind Multipurpose Development Authority (BMDA) in 1992.

Groundwater level is continuously going down every year. **Figure 9** shows the relationship between deep tube well development and water level in dry season from 1966 to 2010. Graph show a clear relationship between water level and development of DTW over time. With increasing number of deep tube wells, every year the rate of depletion of groundwater level is accelerating in dry season. From 1966-1975 hydrograph show almost no change in groundwater level. In 1975 number of deep tube wells was only 5 and from 1986 number is increasing in an accelerating rate and groundwater level also starts to deplete rapidly from 1986. According to groundwater observation well GT8194046 of BWDB minimum water level was fluctuate in between 6.08 m to 6.86 m from 1966 to 1984, and total number of deep tube wells was 77 in 1984. In 1986 minimum water level reached in 9.03 m and number of deep tube wells was 122. In 2010 water level reaches in 14.09 m from 1966's 6.08 m and number of DTWs cross 500 and huge number of shallow tube wells are also come in operation (which known as mini-deep in the study area) by private sector in the meantime.

Figure 10 shows relationships between maximum water level and annual rainfall. Annual rainfall data is used from Rajshahi station of BMD from 1979 to 2010. From the graph it is clear that although rainfalls show a regular pattern, but maximum water level elevation is continuously going down or depth of water level increasing from surface. In 1981 annual rainfall was 2241 mm and water level was in 1.12 m depth from surface and in 2007 annual rainfall was 2018 mm but groundwater level reached in 11.22 m below surface ground. Water level of

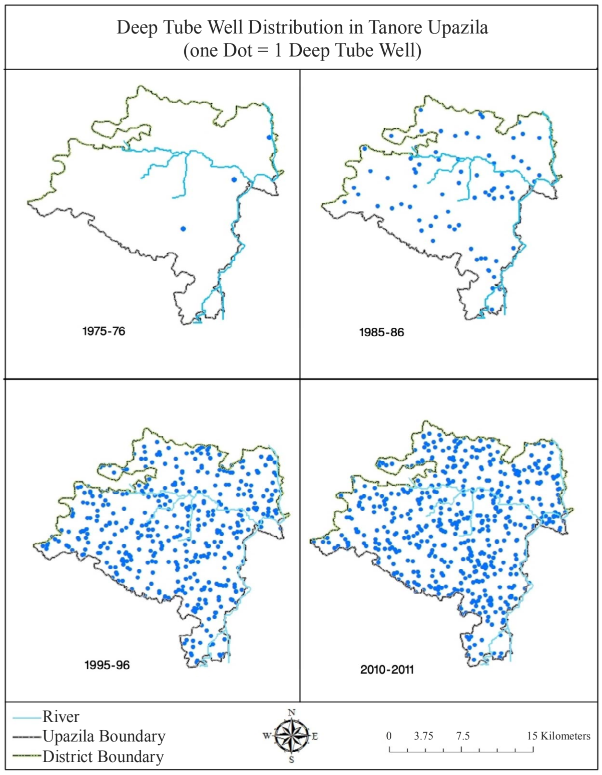

Figure 8. Deep tube well distribution for irrigation in Tanore Upazila over time. Data source: BMDA.

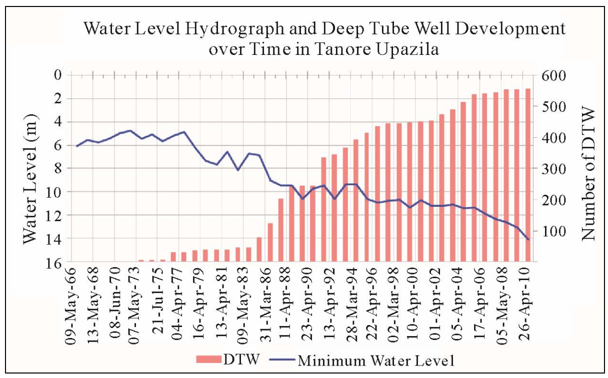

Figure 9. Water level hydrograph and deep tube well development over time in Tanore Upazila. Data source: BMDA, BWDB (Well ID-GT8194046).

well GT8194046 was stayed 1.35 to 1.95 m till 1984 (1.35 m in 1966 and 1.95 m in 1984) in wet season. In 1986 water level reached in 2.03 m and in 2010 it was in 13.77 m below surface. In 1992 and 2010 annual rainfall was lowest 843 mm and 792 mm and water level was 5.3 m and 13.77 m respectively. So, recharging in wet season is overruled by withdrawal of water all year round in study area.

Figure 8 shows the spreading of deep tube well in study area over time and **Figure 3** shows the change in groundwater level elevation in the meantime.

Figure 11 shows rainfall deviations from normal annual rainfall from 1979-2010. It also depicts a regular positive and negative anomaly over time, but maximum groundwater level show continuous negative trend ***(*Figure 10)**.

Due to continuous depletion of water level many hand tube wells suffer layer failure problems and abandoned or replaced by tap (**Figure 12**). Some hand tube wells are also abandoned for availability of tap line not for layer failure. Tap is much convenient than hand tube well which need muscle power during water withdrawal, and when groundwater layer stay at minimum level elevation (msl) in dry season water withdrawal become very tedious job by hand tube well for drinking and other domestic purposes. Water supplied to tap from irrigation deep tube well through overhead tank for drinking purpose. But people now use tap water for drinking and for all other domestic and household uses. Ponds and other sources of surface water are now used only for cultivation of fish and animal birds rearing.

Figure 12. Abandonment of hand tube well due to water layer failure in study area.

3.3. Groundwater Depletion Rate of Tanore

According to five piezometers of BWDB, the groundwater level of Tanore Upazila yet found above the mean sea level but depleting in an accelerating rate. The groundwater depletion rate in Tanore is calculated from three piezometers data of BWDB and two monitoring wells data of BMDA (see **Figure 2**).

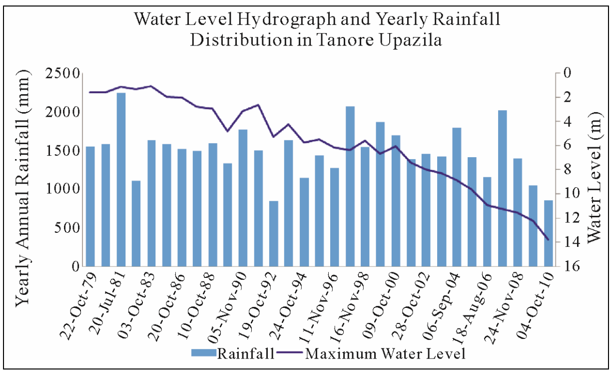

Figure 10. Water level hydrograph and yearly rainfall distribution in Tanore Upazila. Data source: BMD, BWDB (Well ID-GT8194046).

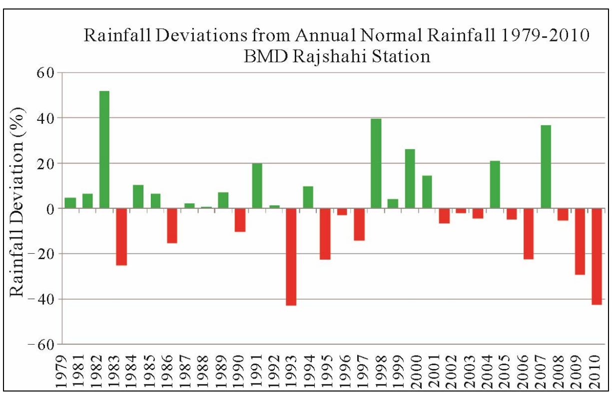

Figure 11. Rainfall deviations from annual normal rainfall in study area.

3.3.1. Depletion Rate in Wet Season

From three BWDB monitoring wells data groundwater depletion rate in wet season when water table stayed at minimum depth from surface and maximum elevation from mean sea level (msl) is higher than one ft per year. The piezometer GT8194046 show depletion rate 1.54 ft/y, GT8194048 and GT8194049 show rate of depletion 1.15 ft/y and 1.26 ft/y respectively (see **Table 1**).

In all three monitoring wells calculation are done from the maximum water level elevation (msl) of 1983 and 2010.

Rate calculation from BMDA monitoring well is done from 1987 and 2010 data. One monitoring well of BMDA is located in Talanda Union and other is in Haripur of Kamargaon Union (**Figure 2**). Calculated depletion rate of both monitoring wells are higher than one ft per year in wet season and the rate is same, 1.44 ft/y. Average rate from five monitoring wells of two organizations in wet season (depletion rate of maximum water level elevation) is 1.37 ft/y. It is clear that the recovery rate is lower than the withdrawal rate as a result the groundwater table is depleting in wet season in an alarming rate.

3.3.2. Depletion Rate in Dry Season

Depletion rate of minimum water level elevation (msl) of Tanore Upazila is also calculated from the same five monitoring wells of two organizations used in depletion rate calculation of maximum water level elevation (see **Table 2**). In dry season water table fall and reach in maximum depth from surface and the elevation stayed minimum from mean sea level.

Table 1. Calculation of depletion rate of maximum water level elevation (wet season) in study area.

Name of the Well	Water Level (ft)	Water Level (ft)	Difference (ft)	Rate of Depletion (ft/y)
BWDB Well ID	**1983**	**2010**		
GT8194046	3.54	45.17	41.63	1.54
GT8194048	22	53.04	31.04	1.15
GT8194049	8.98	42.96	33.98	1.26
BMDA Well	**1987**	**2010**		
Talanda	8.50	41.67	33.17	1.44
Kamargaon	4.67	37.83	33.16	1.44
Average Rate of Depletion of Water Level in Wet Season				1.37

Data source: BWDB, BMDA.

Table 2. Calculation of depletion rate of minimum water level elevation (dry season) in study area.

Name of the Well	Water Level (ft)	Water Level (ft)	Difference (ft)	Rate of Depletion (ft/y)
BWDB Well ID	**1983**	**2010**		
GT8194046	22	46.22	24.22	0.89
GT8194048	31	53.04	22.04	0.82
GT8194049	26.33	43.95	17.62	0.65
BMDA Well	**1987**	**2010**		
Talanda	31.58	42.83	11.25	0.49
Kamargaon	23.17	40.50	17.33	0.75
Average Rate of Depletion of Water Level in Dry Season				0.72

Data source: BWDB, BMDA.

Depletion rate of minimum water level elevation calculated from 1983-2010 using three piezometers of BWDB is below one ft/y. From the well GT8194046 calculated depletion rate is 0.89 ft/y and according to well GT8194048 and GT8194049 the rate is 0.82 ft/y and 0.65 ft/y respectively in dry season. Two BMDA wells also show depletion of groundwater level in dry season and the rate of depletion is 0.49 ft/y in Talanda well and 0.75 ft/y in Haripur well of Kamargaon Union. The well of Talanda is showing lowest depletion rate in dry season may be due to very close location from the River Shiba and wetland Beelkumari which give this area a higher recharging rate than all other parts of the study area located in a distant location from wetland and river (see **Figure 2**). From five wells of two organizations, average value of rate of depletion in dry season is 0.72 ft/y in Tanore Upazila.

From the groundwater monitoring well data, average depletion rate of maximum water level elevation is 1.37 ft/y and average depletion rate of minimum water level elevation in Tanore is 0.72 ft/y. Depletion rate in dry and wet season is different. Rate of depletion in wet season is higher than the rate in dry season. From this nature of depletion it can be stated that groundwater level in the study area is depleting in an accelerating rate from year to year.

Recharging of groundwater occur mainly in between four monsoon months June-September (about 80% of rainfall occur in monsoon period in Bangladesh) and replenishment of water level by annual rainfall is overruled by annual increasing amount of withdrawal. As a result, all five wells of two organizations located in different parts of the upazila are showing depleting nature of groundwater level.

The average value of yearly maximum rate of depletion and minimum rate of depletion in Tanore Upazila from five monitoring wells of two different organizations spread over different parts of the study area is 1.04 ft/y.

4. Conclusions

Result shows that both maximum and minimum groundwater levels of the study area are depleting. Average depletion rate of maximum water level 1.37 ft/y is much higher than average depletion rate of minimum water level, which is 0.72 ft/y. Agriculture is very important in the context of Bangladesh because to feed a huge number of population. Agricultural sector also provide employment to a huge number of unskilled and semi-skilled labors so in this context it is also important. Due to expansion of groundwater irrigation cropping intensity of the study area have crossed even national level, but it is also the prime reason for groundwater depletion of the study area. Intrusion of saline water is not an issue for the area which located in a distant location from the sea and for high thickness of clay layer, land subsidence risk is also comparatively low due to groundwater depletion. Excessive withdrawal is lowering water level in the area successively and surface water bodies can be severely affected by this process.

Crop diversification from water consuming crop (paddy) to less water consuming crops (vegetables, fruits etc.), artificial recharging, increasing dependency on surface water, increasing irrigation efficiency including application of Alternate Wetting and Drying (AWD) method, rainwater harvesting etc, can be option for the study area.

5. Acknowledgements

The authors are thankful to Professor Dr. Kazi Matin U. Ahmed, Department of Geology of University of Dhaka; Barind Multipurpose Development Authority (BMDA) and Bangladesh Meteorological Department (BMD) for financial and other supports.

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Journal of Water Resource and Protection, 2012, 4, 576-589
doi:10.4236/jwarp.2012.48067 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Impact of Land Use and Aquatic Plants on the Water Quality of the Sub-Tropical Alpine Wetlands in India: A Case Study Using Neuro-Genetic Models

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Received April 30, 2012; revised June 1, 2012; accepted June 12, 2012

ABSTRACT

The suspended and dissolved waste in the incoming storm water of wetlands largely depends on the adjacent land use which can influence the quality of the water body. The micro- and macro-floral population of a wetland can absorb, convert, transform and release different organic or inorganic elements, which can also change or impact the overall quality of the wetland water. The present study investigates the influence of the land use and the plant species in the waterbed on the water quality of a high-altitude, sub-tropical wetland in India. The estimation capabilities of neuro-genetic models were utilized to identify the inherent relationships between the Biochemical Oxygen Demand (BOD), Dissolved Oxygen (DO), chlorine (Cl) and Chemical Oxygen Demand (COD) with the land use and wetland zoology. A thematic map of the quality parameters was also generated based on the identified relationship to observe the influence that the morphological and biological diversity in and around the study area has on the quality parameters of the wetland. According to the results, the BOD, COD and Cl were found to vary with differences in land use and the presence of different plant species, whereas the DO was found to be largely invariant with changes in these parameters. The reasons may be contributed to the impact of uncontrolled eco-tourism activities around the wetland.

Keywords: Wetland; Neural Network; Water Quality; Land Use; Aquatic Plants

1. Introduction

Nowadays, maintaining the water quality of freshwater wetlands has become a significant issue because, for this kind of wetland, municipal and industrial wastewater discharge constitutes a constant polluting source, whereas surface run-off is a seasonal phenomenon [1]. For this reason, the water environment quality issue is a subject of ongoing concerned for the development of an economy in any country [2]. Naturally, a well-designed water-quality monitoring plan should preserve scarce resources by minimizing the redundancy of nearby monitoring stations and the plethora of possible variables monitored, while at the same time maximizing the information content of the collected data [3]. However, it is also true that to construct a well-designed water-quality monitoring plan, detailed testing of the water quality is essential for any freshwater wetland. In this paper, a multivariate statistical characterization of water quality of a tropical, freshwater lake in eastern India is discussed.

Some studies are available that deal with multiple purposes, including water quality monitoring. The usefulness of multivariate statistical techniques demonstrated [4] for the evaluation and interpretation of large complex water-quality data sets and the apportionment of pollution sources/factors with the intention of obtaining better information about the water quality and the design of the monitoring network for the effective management of water resources. A data set of 10 years analyzed [5] surface water quality data pertaining to a polluted river using partial least squares (PLS) regression models. In their study, both the unfold-PLS and N-PLS (tri-PLS and quadri-PLS) models were applied to the multivariate, multi-way data array with the intention of assessing and comparing their predictive capabilities for the biochemical oxygen demand (BOD) of a river water in terms of their relative mean squared errors of cross-validation, prediction and variance captured. However, the principal component analysis applied [6] 16 water-quality parameters that had been collected monthly over a 6-year period in an effort to describe the spatial dependence and inherent variations of water quality patterns in the Florida Bay-Whitewater Bay area. Moreover, Researcher [7] tried to establish how the last years of artificial management have affected the ecosystem of the Tablas de Daimiel National Park, a Spanish continental wetland. To carry out this study he analyzed the water physico-chemical characteristics over the period 1995-1997, and using different statistical techniques, these data were compared with those obtained from a survey conducted from 1974-1975, which represented the original situation. However, geochemical characteristics and the apparent ages of sampled groundwater were used [8] to determine which of the two regionally extensive bedrock aquifers, the lower bedrock aquifer or the upper bedrock aquifer, is a more likely source of water discharging into the springs in the Nine Springs watershed, which is located in south-central Dane County, WI. Even, the potential of systematic and formalized interdisciplinary research concepts and methods for sustainable water and wetland policy and management were reviewed and examined [9], as advocated by the recently adopted European Water Framework Directive. However, the multivariate data analysis applied [10] large water quality data sets on the Buyuk Menderes River Basin to analyze the surface water contamination and establish correlations between water quality parameters. Later, the water quality was examined [11] of the Tahtali River Basin in Turkey. In this study, multivariate statistical methods, including factor, principal component and cluster analyses, were applied to surface water quality data sets obtained from the Tahtali River Basin. The factor and principal components analyses results revealed that the surface water quality was mainly controlled by agricultural uses and domestic discharges. The cluster analysis generated two clusters. Moreover, a strategy was presented [3] to reduce the measured parameters, locations, and frequency without compromising the quality of the monitoring program. Even so, the surface water quality data sets were analyzed [2] to obtain from the Xiangjiang watershed, which were generated over 7 years (1994-2000) and monitored 12 parameters at 34 different profiles with the help of multivariate statistical methods, including factor, principal component and cluster analysis. The multivariate statistical methods, *i.e.*, cluster analysis (CA) and discriminate analysis (DA) were used [12] to assess temporal and spatial variations in the water quality of the watercourses in the Northwestern New Territories of Hong Kong over a period of five years (2000-2004) using 23 parameters at 23 different sites. The principal component analysis (PCA) was used [13] to reduce the data dimensionality from the 18 original physico-chemical and microbiological parameters which determined in drinking water samples to six principal components that explained about 83% of the data variability to analyze 126 drinking water samples taken from a city water network in North Moravia, the Czech Republic, over the course of six months, according to a monitoring plan. In addition, some samples were collected [14] to analyze the parameters such as Temperature, pH, DO, Conductivity, Turbidity, Total Suspended Solids (TSS), Nitrate, Phosphate, COD and BOD from ten sampling locations distributed along the Juru Estuary in the Penang state of Malaysia to analyze the water quality data with help of CA and descriptive statistics. However, the environmental economics for wetland construction, restoration and preservation, and the net ecosystem services values of constructed, human-interfered and natural wetlands explored [15] as a comparative study for the case of a typical human-interfered wetland in Wenzhou, China. Nonetheless, Turner [16] emphasized an integrated wetland research framework, which suggests that a combination of economic valuation, integrated modeling, stakeholder analysis, and multi-criteria evolution can provide complementary insights into sustainable and welfare-optimizing wetland management and policy. Furthermore, Ouyan [17] applied principal component analysis (PCA) and principal factor analysis (PFA) techniques to evaluate the effectiveness of the surface water quality-monitoring network in a river, where the variables are evaluated at the monitoring stations. The objective of his study was to identify the monitoring stations that are important in assessing the annual variations of river water quality. Detenbeck [18] even developed a method that evaluated the cumulative effect of wetland mosaics on the water quality, which was applied to 33 lake watersheds in the seven-county region surrounding Minneapolis-St. Paul, Minnesota. Wayland [19] compared biogeochemical data from three synoptic sampling events, which represents the temporal variability of base flow chemistry and land use, using R-mode factor analysis. At the same time, a meta-analysis was described [20] to estimate relationships between the non-use components of willingness to pay (WTP) for surface water quality improvements and a combination of resource, context, and study design attributes, where these attributes include estimated use values for identical improvements. The relationships between water quality and six different land uses were investigated [21] to offer practical guidance in the planning of future urban developments. In terms of safeguarding the water quality, high-density residential development, which results in a smaller footprint than sparse development; should be the preferred option according to his study. The R-mode factor analysis and Q-mode cluster analysis were applied [22] to a set of 1349 groundwater analyses to determine the factors controlling the groundwater composition and the main resulting water types. The PCA was used [23] to assess the degree of contamination and spatial distribution of heavy metals, such as Ag, As, Cd, Co, Cr, Cu, Hg, Ni, Pb, Sr, Zn, and nutrients (Org-C, Tot-N and Tot-P) in different areas of Taihu Lake in China.

Wetlands are also believed to play a significant role in global climate change by acting as a source of an atmospheric greenhouse gases, such as methane, carbon, and nitrogen [24]. Global biodiversity is also enhanced by wetlands, which are vital for the survival of disproportionately large number of threatened and endangered species [25]. However, uncontrolled domestic discharges caused by rapid urbanization are threatening the surface water quality [26] of wetlands. Consequently, various physico-chemical and microbiological parameters of water and different biogeochemical cycles of wetlands are affected intensely. In such a situation, the present wetland is really important because it represents a major group of Indian wetlands that are endangered by a lack of appreciation of the importance of their role and as soft target of developers.

The land use and quality of water in the wetlands are correlated. The runoff from the industries, roads normally has higher concentration of metallic compounds and dissolved wastes than from hills or forests whereas organic compounds are found in higher concentration in runoff from the latter. The type of floral species that reside in the wetland bottom can also influence the water characteristics. Shrubs can uptake pollutants from the muck layer, small algae can increase dissolved oxygen. The microbes can increase the BOD by decomposing the organic wastes. **Table 1** depicts the impact of different land use on the water quality of wetlands as observed by [27,28] and many other scientists.

The floral population found in the wetlands was also found to be influential in controlling water quality of such water bodies. The quality of the wetlands can be accessed by observing the presence of different kind of plant species. Some examples of such indications are shown in **Table 2**.

1.1. Impact of Major Quality Parameters on the Wetland Quality

As per recommendation by the APHA [29] the observation of the following parameters can yield an overview of the overall water quality of water bodies. The indicative properties of such parameters can give a clear idea about the overall quality of the wetlands.

1.1.1. Biochemical Oxygen Demand (BOD)

Oxygen is used for respiration in animals. Fish require the highest concentrations of oxygen. If the dissolved oxygen falls below 5 ppm (part per million), fish are the first to suffer and die. Then, the population of bacteria rises to abnormal levels. Imbalances between species are a sign of water pollution. Substances that consume dissolved oxygen and add to the biochemical oxygen demand are pollutants. Such substances come from human

Table 1. Relationship between land use and the water quality of water bodies, as documented in different scientific articles.

Land Use	Possibility of Water Contaminant	Reason
Road (R)	High concentrations of chloride, nitrate and pesticides can be observed in the adjacent wetlands. The amount of concentration depends on the population and road density of the contributing area.	Salt-treated roads, surface runoff from adjacent residential, agricultural and industrial regions, population density and soil porosity can vary the intensity of contamination.
Forest (F)	The lowest chloride and nutrient concentrations can be observed in wetlands adjacent to high density forests.	As mixing of residential and industrial wastes with surface as well as groundwater increases the chloride and nutrient concentration, absence of the same has decreased the extent of contamination.
Road & Agriculture	Presence of pesticides and herbicides.	The fertilizers applied in the adjacent agricultural area can contaminant surface runoff as well as seepage from aquifers, thereby increasing the toxicity and nutrient content of the wetland water.
Road & Industry	Presence of volatile organic compounds like Trichloroethane.	The effluents from petroleum and organic industries, if mixed with ground and surface water, can severely contaminate the wetland water.
Road & Residential	Domestic sewage can increase the concentration of Nitrates, Chlorides and dissolved nutrients. The municipal sewage water will have an elevated concentration of organic compounds and ammonia which can deplete the Singh *et al*., (2006) DO of the water body.	High population densities and human activity can contribute to the contamination of water bodies.
Hill	The surface runoff that flushes in from a forest may filter the dead bodies of macro-phytes. The mixing of such runoff with the water body is quick and also increases the diffusion of oxygen from the atmosphere due to the aeration of surface water by high flow velocities. The lowest Chloride and nutrient concentrations can be observed in wetlands adjacent to high density forests in the hills.	Because the mixing of residential and industrial wastes with surface and ground water increases the chloride and nutrient concentration of the wetland, the absence of the same has decreased the amount of the contaminants.

Table 2. Relationship between plant species and the water quality of wetlands as documented in different scientific and government reports.

Types of Aquatic Species	Symptoms	Water Quality	Reason
Rooted floating leaved plants like the water lily	Increased growth	Sediment contaminants may reach plant bodies. The Chloride or Nitrate concentration will be reduced due to absorbance by the aquatic plants. The DO will also be reduced under extreme conditions.	Presence of forest or agriculture fields can increase the growth of such plants because the surface runoff can flush in nutrient rich water and sometimes act as a carrier agent of the rooted plants.
Submerged plants	Increased growth	Decrease in the DO due to the respiration of the submerged plants. The dead bodies of such plants will attract microorganisms, which will increase the BOD of the water body. Such plants can trap nutrient-attached sediments from reaching the algal population, thereby preventing the occurrence of algal blooms.	Increase in the nutrient concentration due to human activities like the deposition of wastes, industrial effluents, etc. The aeration of water, which will increase the DO in the epilimnion and create an environment conducive for the germination of such plants.
Free Floating Plants like water hyacinth (Eichhornia crassipes)	Increased growth	Increase in the DO but because the growth of free floating plants is normally aggressive, the native species of the water body face severe depletion. The chloride or nitrate concentration will decrease due to the absorbance of the metallic ions by such plants.	Increase in the nutrient concentration due to human activities, like the deposition of wastes, industrial effluents etc.
Algae like *Spirogyra* sp., blue green algae, etc.	Increased growth	Minor increase in the DO, but the lake color and odor will change. Some types of filamentous algae may produce scums or mats. Due to the parasitism of microorganisms, lake water will show higher BOD values.	Increase in the nutrient concentration due to human activities.

waste. The amount of dissolved oxygen used up during oxidation by bacteria of the organic matter in a sample of water is called the biochemical oxygen demand (BOD). Water is rated as pure if the BOD is 1 ppm or less, fairly pure with a BOD of 3 ppm and suspect when the BOD reaches 5 ppm.

1.1.2. Chemical Oxygen Demand (COD)

The chemical oxygen demand (COD) test is used to indirectly measure the amount of organic compounds in water that can be oxidized with both organic and inorganic oxidizing agents. The regulated amount of COD for surface water is generally 200 - 1000 mg/L but differs with respect to country and state.

1.1.3. Dissolved Oxygen (DO)

Dissolved oxygen (DO) is the oxygen that is dissolved in water by diffusion from the surrounding air or aeration of water. Fish and aquatic animals cannot split oxygen from water (H_2O) or other oxygen-containing compounds. Only green plants and some bacteria can do that through photosynthesis and similar processes. Virtually all of the oxygen we breathe is manufactured by green plants. A total of three-fourths of the earth's oxygen supply is produced by phytoplankton in the oceans. If water is too warm, there may not be enough oxygen in it. When there are too many bacteria or aquatic animal in the area, they may overpopulate, and consume the DO in great amounts. Oxygen levels also can be reduced through the over-fertilization of water plants by run-off from farm fields containing phosphates and nitrates (the ingredients in fertilizers). Under these conditions, the numbers and sizes of water plants increase a great deal. Then, if the weather becomes cloudy for several days, respiring plants will use much of the available DO. When these plants die, they become food for bacteria, which in turn multiply and use large amounts of oxygen.

Numerous scientific studies suggest that 4 - 5 parts per million (ppm) of DO is the minimum amount that will support a large, diverse fish population. The DO level in good fishing waters generally averages about 9 parts per million (ppm). When DO levels drop below about 3 parts per million, even hardy fish will die.

1.1.4. Chlorine (Cl)

Chlorine is used as disinfectant due to its capacity of oxidation. Chlorine can be found as free chlorine or as total chlorine. Free chlorine is highly toxic for aquatic inhabitants and microbes due to its highly oxidizing nature. Aquatic animals can tolerate up to 1 mg/L of free chlorine and fish will usually die if more than 0.36 mg/L of chlorine is found in the water. However, low concentrations of chlorine, *i.e.* less than 0.1 mg/L, can improve the quality of the water body. The total chlorine can represent the salinity of a water body where an excess amount of chloride can harm the aquatic inhabitants.

The present investigation aims to identify the relationship between the land use, type of aquatic plant and the above water quality parameter. A brief introduction and the common methodology followed in achieving the objectives through neural networks are discussed next.

1.2. Artificial Neural Network (ANN)

An (ANN) is a flexible mathematical structure that is capable of identifying complex nonlinear relationships between input and output data sets. Artificial Neural Networks (ANNs) offer a relatively quick and flexible means of modeling, and as a result, the application of ANN modeling has been widely reported in various hydrological studies [30-32]. In the context of hydrological forecasting, recent papers have reported that ANNs may offer a promising alternative for rainfall-runoff modeling [33-36], stream flow prediction [37-40], reservoir inflow forecasting [41,42] and the prediction of water quality parameters [43]. All the papers reported a high degree of satisfaction with the neural network estimations.

Artificial neural networks are viable computational models for a wide variety of problems. These include pattern classifications, speech synthesis and recognition, adaptive interfaces between humans and complex physiccal systems, function approximation, image compression, associative memory, clustering, forecasting and predicttion, combinatorial, combinatorial optimization, nonlinear system modeling, and control. These networks are "neural" in the sense that they may have been inspired by neuroscience but not necessarily because they are faithful models of neurobiological or cognitive phenomena. In fact, the majority of the networks covered in this book are more closely related to traditional mathematical and statistical models, such as non-parametric pattern classifiers, clustering algorithms, nonlinear filters, and statisticcal regression models than they are to neurobiologycal models.

1.2.1. Mathematical Representation of Artificial Neural Network

An (ANN) (see **Figure 1**) is a flexible mathematical structure that is capable of identifying complex nonlinear relationships between input and output data sets. The ANN model of a physical system can be considered as n input neurons $(x_1, x_2 \cdots x_n)$, h hidden neurons $(z_1, z_2 \cdots z_n)$ and m output neurons $(y_1, y_2 \cdots y_n)$. Let *tj* be the bias for neuron z_j and f_k for neuron y_k. Let w_{ij} be the weight of the connection from neuron x_i to z_j and beta is the weight of the connection z_j to y_k. The function that ANN calculates is:

$$y_k = g_A\left(\sum z_j b_{jk} + f_k\right) \quad (j = 1 - h) \tag{1}$$

In which,

$$z_j = f_A\left(\sum x_j w_{ij} + t_j\right) \quad (i = 1 - h) \tag{2}$$

where g_A and f_A are the activation functions [44].

The development of an artificial neural network, as prescribed by ASCE [45] follows the following basic rules:

1) Information must be processed at many single elements called nodes.

2) Signals are passed between nodes through connection links, and each link has an associated weight that represents its connection strength.

3) Each of the nodes applies a non-linear transformation called an activation function to its net input to determine its output signal.

The numbers of neurons contained in the input and output layers are determined by the number of input and output variables of a given system. The size or number of neurons of a hidden layer is an important consideration when solving problems using multilayer feed-forward networks. If there are fewer neurons within a hidden layer, there may not be enough opportunity for the neural network to capture the intricate relationships between indicator parameters and the computed output parameters. A network with too many hidden layer neurons not only requires a large computational time for accurate training but may also result in overtraining. A neural network is said to be "over-trained" when the network focuses on the characteristics of individual data points rather than just capturing the general patterns present in the entire training set. The network building procedure is divided into three phases, which are described next in a broad way.

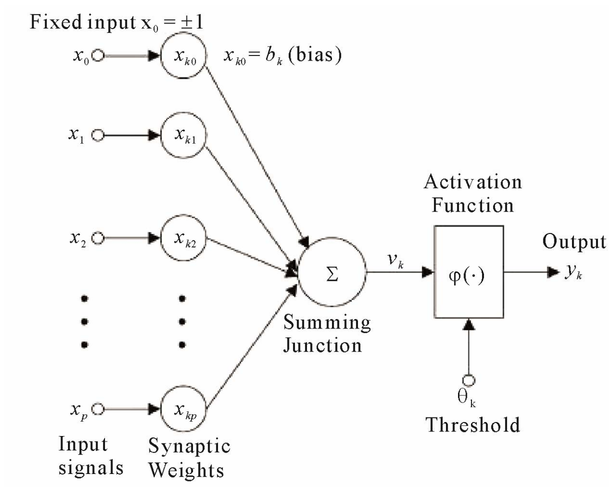

Figure 1. A schematic diagram of an artificial neural network.

1.3. Study Area

Mirik Lake (26˚54'N to 26.9˚N—latitude and 88˚10'E to 88.17˚E—longitude) is situated in a valley encircled by hill ridges with an extensive natural drainage network. This lake is located at an altitude of 1767 meters above the sea level. It is 49 km from Darjeeling and falls in the state of West Bengal in India. On the western side close to Mirik Lake flows the Mechi River, which demarcates the Indo-Nepal border. The climate is pleasant all year round with temperatures of a maximum 30˚C in the summer and a minimum of 2˚C in winter. Mirik Lake is surrounded by the Mirik bazaar, Thana-line, Krishnanagar, Pratapgaon, and Mahendragaon (wards nos. 2, 3, 5, 7 and 8 respectively). Its rich biodiversity, location on a migratory bird route, and vast areas of suitable habitats for multiple bird species make the lake important center for wildlife [46]. Mirik Lake as a whole contains multifarious features for boating, recreation, jogging, fairs, picnics and many other activities. The total of pollution load is drained from the surface runoff carrying the domestic and municipal sewage for the entire Mirik Lake. Some other sources of pollution, such as the outflow from hotels carrying waste, human excreta from poor sanitation, washing clothes, bathing, etc., are drained into the Lake. **Figures 2** and **3** show the location of sampling points on Mirik Lake and the land use map of Mirik Lake, respectively.

1.4. Objective

The present study investigates the influence of land use and the characteristics of the wetland bottom, on the water quality of wetlands. A neuro-genetic model was developed to estimate the interrelation between the former with the latter. The study can reveal the answers to question such as:

1) How does the quality parameter vary from the peripheral region to the central region of the wetlands? The

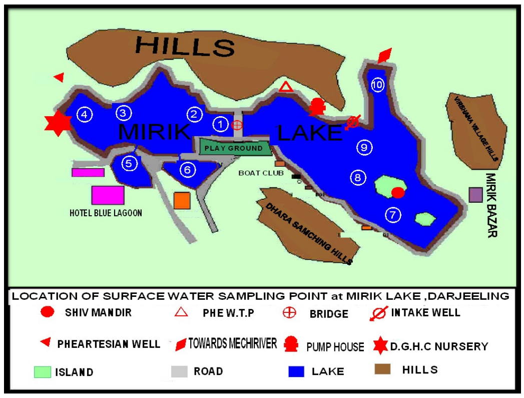

Figure 2. Location of sampling points in mirik lake.

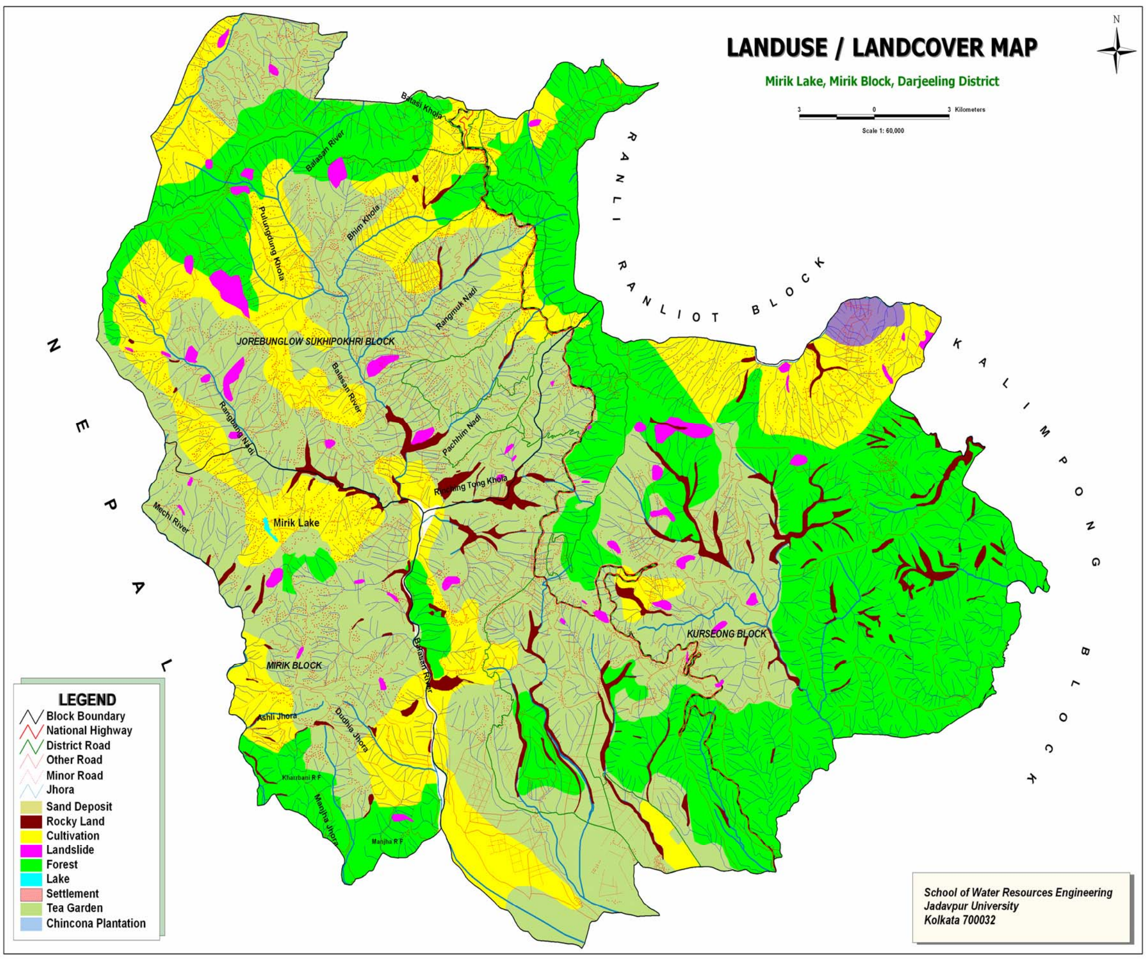

Figure 3. Land use map of mirik lake.

quality of the water in the peripheral region will be significantly influenced by the adjacent land use, but the water quality in the central region will be impacted by the floral population of the wetland.

2) What are the relationship between land use and the quality of stored water and why do these relationships exist?

3) What are the relationships that exists between aquatic plants and the quality parameters of the water and why do these relationships exist?

2. Methodology

2.1. Sample Collection and Analysis

During study period, the water-spread area of Mirik Lake was about 162,256 m^2, the width was 163 m, the depth was 1.65 m and the total volume of the water was about 2,677,224 m^3. The point sources of pollution of this lake are the four main drains for the domestic and municipal sewage. The non-point sources of pollutions include the outflow of hotels, clothes washing, bathing and surface run off from the surrounding areas.

Samples (**Figure 2**) were collected from all the sampling points on the same day at different times. The DO, pH, temperature and turbidity were measured on spot in the field with Rugged Field Kit HQ Series Portable Meters (HQd/IntelliCAL™-8505300-HACH). Samples were brought to Kolkata for further physico-chemical (BOD, COD and chloride) and bacteriological analysis at the School of Water Resources Engineering, Jadavpur University as per the standard method [30].

2.2. Selection of the Land Use Class

A land use map (**Figure 3**) of the study area was developed with the help of satellite imagery and a ground tooth sample survey. The major land uses within the 500 m diameter around the lake were identified as: Hill (H), Forest (F), Pond (P) and Road (R) as can be observed from **Figure 2**, which also showed the collection points for surface water.

2.3. Identification of Major Aquatic Species of the Lake

The aquatic species of the lake were identified from both the ground tooth survey and microscopic tests of the collected samples. The three major types of planktons identified were divided into the following classes: Rooted Floating Leaves or Shrubs (S), Submerged Plants or Shrubs (SS) (**Table 2**), and Clear Water (C), which represents water with no noticeable floral population.

The identified land use and plant classes along with the value of BOD, COD, DO and Cl parameters from the collected samples, a neural network model was developed with each quality parameter as output and land use, plant classes, distance from the wetland periphery and other 3 quality parameter as input. The other parameters were included as input to educate the model about the interrelationship if any in-between the parameters which also help to validate the model output.

2.4. Development of the Neuro-Genetic Models

2.4.1. Network Building Procedure

Selection of Network Topology

Neural networks can be of different types, like feed forward, radial basis function, time lag delay etc. The type of network is selected with respect to the knowledge of input and output parameters and their relationships. Once the type of network is selected, selecting the network topology is the next concern. A trial and error method is generally used for this purpose, but many studies now prefer the application of a genetic algorithm [47]. Genetic algorithms are search algorithms based on the mechanics of natural genetics and natural selection. The basic elements of natural genetics—reproduction, crossover, and mutation—are used in the genetic search procedure. A GA can be considered to consist of the following steps [48]:

1) Select an initial population of strings.
2) Evaluate the fitness of each string.
3) Select strings from the current population to mate.
4) Perform crossover (mating) for the selected strings.
5) Perform mutation for selected string elements.
6) Repeat steps 2) - 5) for the required number of generations.

The genetic algorithm is a robust method of searching for the optimum solution to complex problems, such as the selection of an optimal network topology, where it is difficult or impossible to test for optimality. The basics of GAs have already been discussed by many authors [47,49,50]. Hence the details of the basic procedures of GAs are not discussed in the present literature.

2.4.2. Training Phase

To encapsulate the desired input output relationship, the weights are adjusted and applied to the network until the desired error is achieved. This is called as "training the network".

2.4.3. Testing Phase

After training is completed, some portion of the available historical dataset is fed to the trained network and a known output is estimated out of them. The estimated values are compared with the target output to compute the MSE. If the value of MSE is less than 1%, then the network is said to be sufficiently trained and ready for estimation (see **Figure 4**). The dataset is also used for

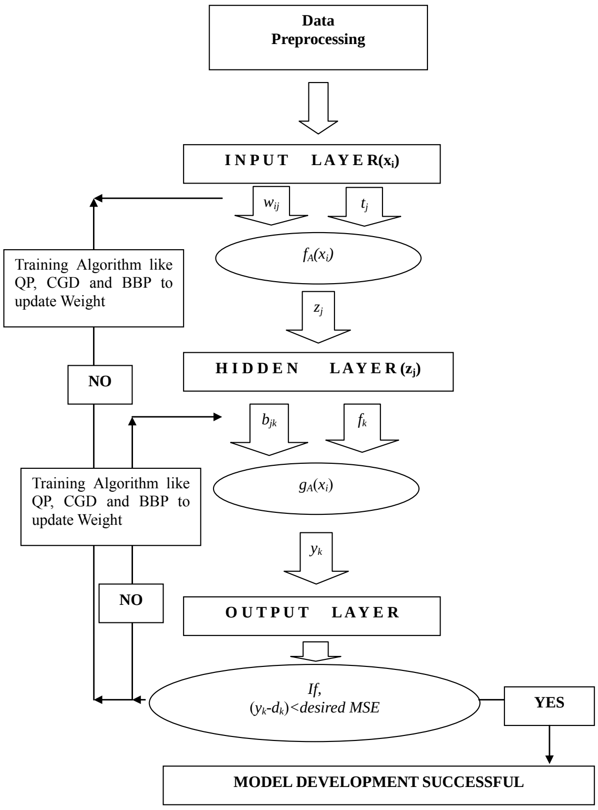

Figure 4. The basic methodology followed for the development of a Neural Network [40].

cross-validation to prevent over-training during the training phase [44].

3. Result & Discussion

In total four neuro-genetic models were prepared with one of four quality parameters as the output. The models were trained with the Quick Propagation and Conjugate Gradient Descent training algorithms, and the performance analysis of the results from the models revealed QP trained models for BOD and DO and CGD trained models for COD and Cl as the best trained neuro-genetic model (see **Table 3**). The estimation work was carried out with the help of the model trained with the selected algorithm.

3.1. Discussion

According to the **Tables 1**, **2** and **4** following observation and discussions are made to analyze the impact of land use and aquatic plants on the quality of wetland water.

According to the results (**Figure 5**), eight combinations of land use and types of wetland bottoms were identified. The results showed that the BOD is worst (*i.e.*, highest concentration) for the combination of road and submerged floral species and the combination of clear water and forest yielded the best concentration of the BOD (*i.e.*, lowest concentration). The combination of S and R showed a moderate concentration of BOD in the lake water. BOD concentrations of a lake are known to increase in the presence of organic content, which invites micro-bacteria. The bacteria, with the help of dissolved oxygen, decay the organic matter. The presence of organic content can thus reduce the DO of a lake. Lowering the DO concentration can severely impact the fish population and other floral and faunal colonies of the lake because these species depend on the lake DO for food production and respiratory activities. In the present investigation, the presence of submerged species in the water may increase the organic content of the water. The dead bodies of such species can increase the BOD concentration. The road adjacent to the lake will indicate heavy depositions of organic wastes in the pond from the incoming population, and because the lake is famous for eco-tourism, the influx of temporary population is very high. However, the absence of any biological species in the lake water and forest in the adjacent areas has reduced the deposition of organic wastes in the water. The forest had acted as a filter for removing organic matter, such as the dead bodies of animals and large trees from

Table 3. The specification used and results achieved from the neuro-genetic models developed for the present study.

Network	Input	Hidden Layer	Output	Training Algorithm	Training MSE	Testing MSE	MSE	r	SD
BOD	6.00	7.00	1.00	QP	0.05	0.08	0.06	0.87	0.96
BOD	6.00	3.00	1.00	CGD	1.21	1.56	1.02	0.76	0.87
COD	6.00	6	1.00	QP	7.87	7.68	5.67	0.76	0.88
COD	6.00	3.00	1.00	CGD	5.45	5.88	4.95	0.88	0.90
DO	6.00	3	1.00	QP	0.05	0.045	0.05	0.98	0.89
DO	6.00	3.00	1.00	CGD	0.07	0.08	0.08	0.76	1.20
Cl	6.00	4	1.00	QP	0.02	0.04	0.06	0.89	0.95
Cl	6.00	16.00	1.00	CGD	0.01	0.03	0.01	0.98	0.96

Table 4. The impact of characteristics of adjacent land use and wetland bed on the four quality parameters considered for the present investigation.

Combination	LUB	LUA	COD	Cl	BOD	DO
1	S	R	26.27	9.55	8.68	5.67
1	S	R	95.85	7.87	5.04	5.96
1	S	R	37.56	5.50	9.65	5.54
1	S	R	63.90	4.06	9.24	5.62
2	N	H	45.72	5.23	9.53	5.82
3	SS	H	18.83	19.53	11.05	5.22
3	SS	H	143.91	12.74	2.52	4.42
3	SS	H	55.11	8.51	12.02	4.52
3	SS	H	42.82	21.62	14.80	5.62
2	C	R	368.87	29.54	5.55	5.79
4	C	F	30.55	20.20	0.24	4.97
2	C	R	84.43	26.41	9.84	4.91
5	C	P	16.10	28.13	23.29	4.63
2	C	R	207.98	21.93	10.32	4.66
2	C	H	32.58	15.07	8.92	4.84
6	S	H	41.42	21.16	8.31	5.16
7	SS	H	30.24	22.54	8.52	4.93
7	SS	H	16.29	15.54	24.53	4.97
7	SS	R	56.94	26.65	21.74	4.59
8	SS	R	30.51	26.23	20.59	4.56

the surface runoff coming through the forest. Hence, the BOD of such areas were found to be lower than those of other areas that are widely visited by the tourist and locals who are economically dependent on the former for their sustainability, but the locals and tourists had also converted the lake into a basket for their waste materials.

The presence of non-biodegradable wastes can increase the COD concentration of the lake. According to the results (see **Figure 6**), the combination of clean water and road yielded the higher values of COD, and a combination of SS and H yielded the lower values of the parameter. The absence of shrubs or submerged species can reduce the BOD of a water body, but the presence of non-biodegradable wastes can increase the COD of the same. The presence of roads has only smoothed the path of such wastes with the surface runoff coming into the lake unhindered. That may be the reason for high concentrations of COD when the road is present within the 500 m of the lake. The results from the model also showed that the COD (mg/L) is lower (≤75 mg/L) whenever shrubs or submerged shrubs are present in the wetland bed, but the biochemical oxygen demands (BODs) of such areas are found to be more than 5 mg/L. The reason can be attributed to the presence of abiotic bacteria present in the plankton population of the lake. These bacteria produce their food with the help of oxygen bonded to metallic ions. That is why the CODs in such areas are lower because the chemical oxygen is not used for decaying inorganic wastes, but BOD is more than 5 mg/L because the decomposition of organic waste is done by the abiotic bacterial population.

In case of the DO (see **Figure 7**), the identified combinations yielded no noticeable differences, but the DO is found to be more for shrubs and road combination and less for submerged shrubs and hill combinations. The probable reason can be attributed to ecotourism events, such as boating and fishing, which are rampant in the lake and may aerate the lake water. Again, presence of shrubs can also maintain the oxygen content of the lake due to the ribosomal bacteria present in the roots, which release oxygen during nutrient uptake. The relative in crease of the DO in presence of shrubs may be the results of such nutrification procedures.

From the prediction of chlorine (see **Figure 8**) it can be observed that the concentration is higher for SS and R combinations, whereas the same is lower for S and R combination with respect to the other combinations identified. The presence of a road within 500 m of the lake can contribute to the increase in chlorine concentration. The lake, in case of the present investigation, is a popular place for eco-tourism and the generated organic as well as inorganic waste are deposited in the lake. The surface runoff from the adjacent high altitude land also brings dissolved chlorine due to the uncontrolled use of fertilizers in the adjacent floriculture. The unhindered surface (due to the road area) from these areas along with deposition of wastes by the tourists can cumulatively influence the increase in chlorine concentration of the lake. However, because water plants are popular for their chlorine uptake, the presence of shrubs has reduced the chlorine concentration and absence of the same has allowed the concentration to rise. The toxic byproducts generated from a submerged algal population can increase chlorine content, so the presence of submerged planktons and the absence of shrubs may have allowed the chlorine concentration to rise.

From **Table 4** and the discussions above, it can be observed that SS is an influential factor, which may impact the quality of water, because most of the cases the presence of submerged shrubs had caused the differences in the concentration of water quality parameters of the lake. The road and hills, present in the area within 500 m of

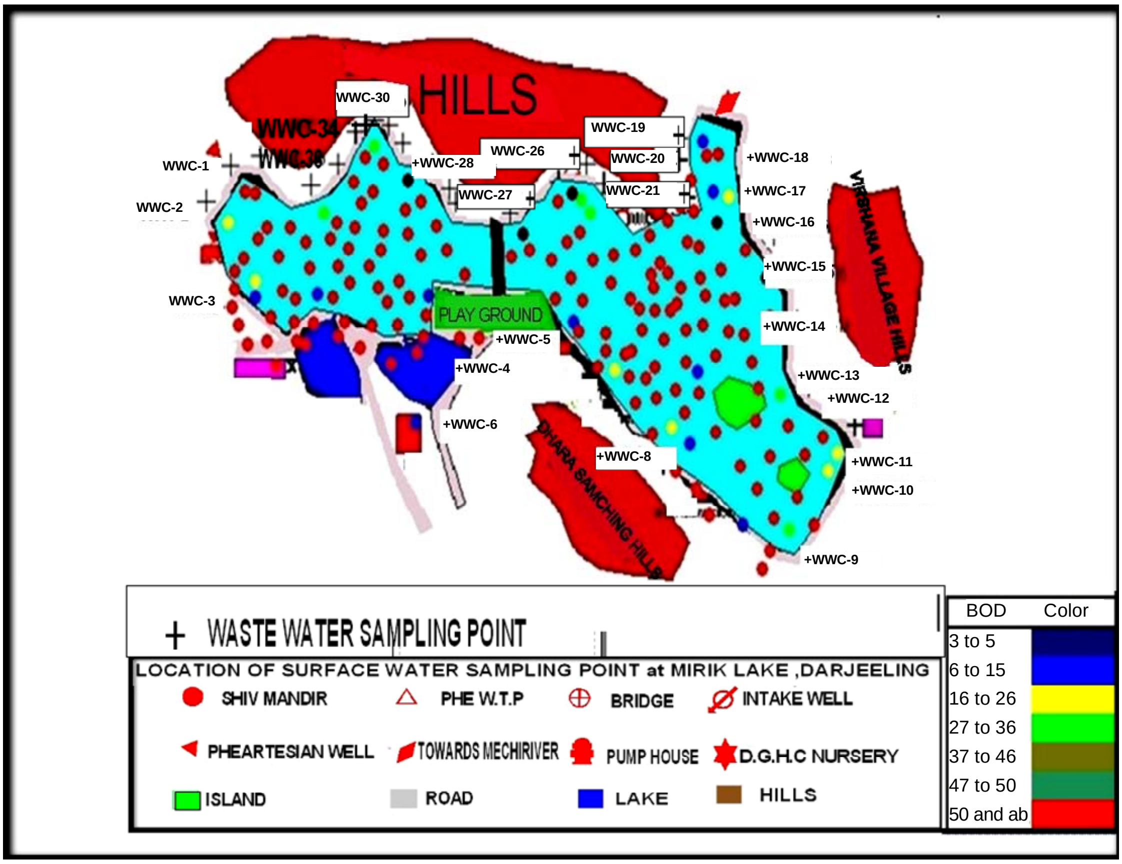

Figure 5. Thematic map of the BOD concentration generated from the identified relationships between the land use, wetland bed and the quality parameter.

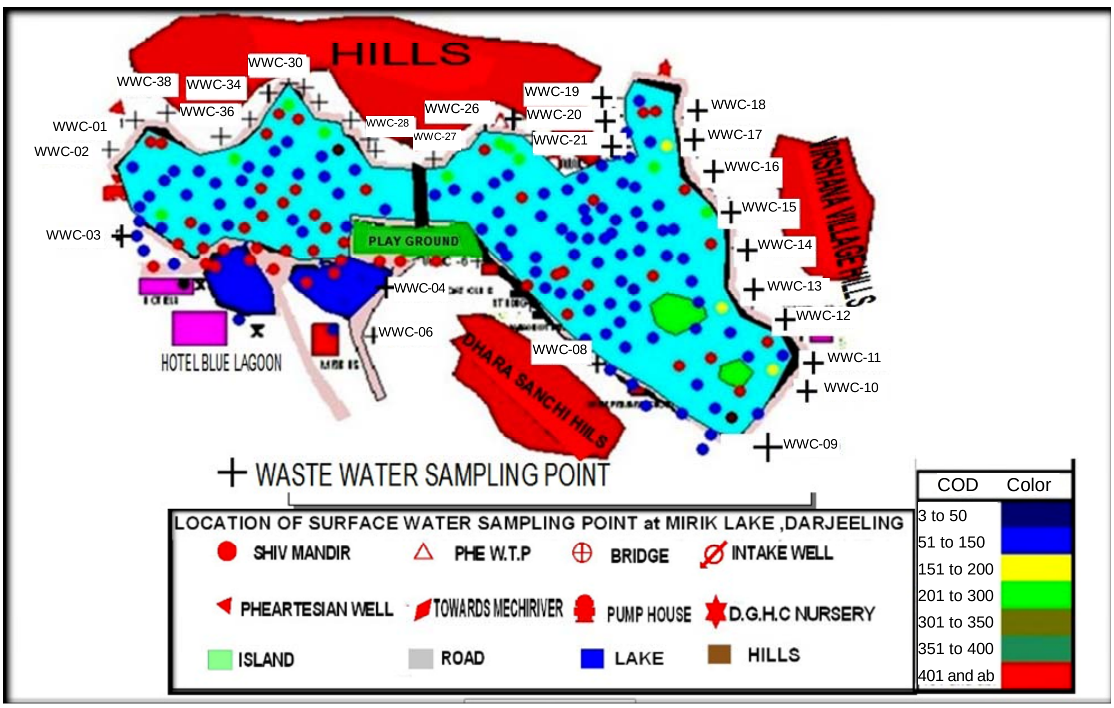

Figure 6. Thematic map of the COD concentration generated from the identified relationships between the land use, wetland bed and the quality parameter.

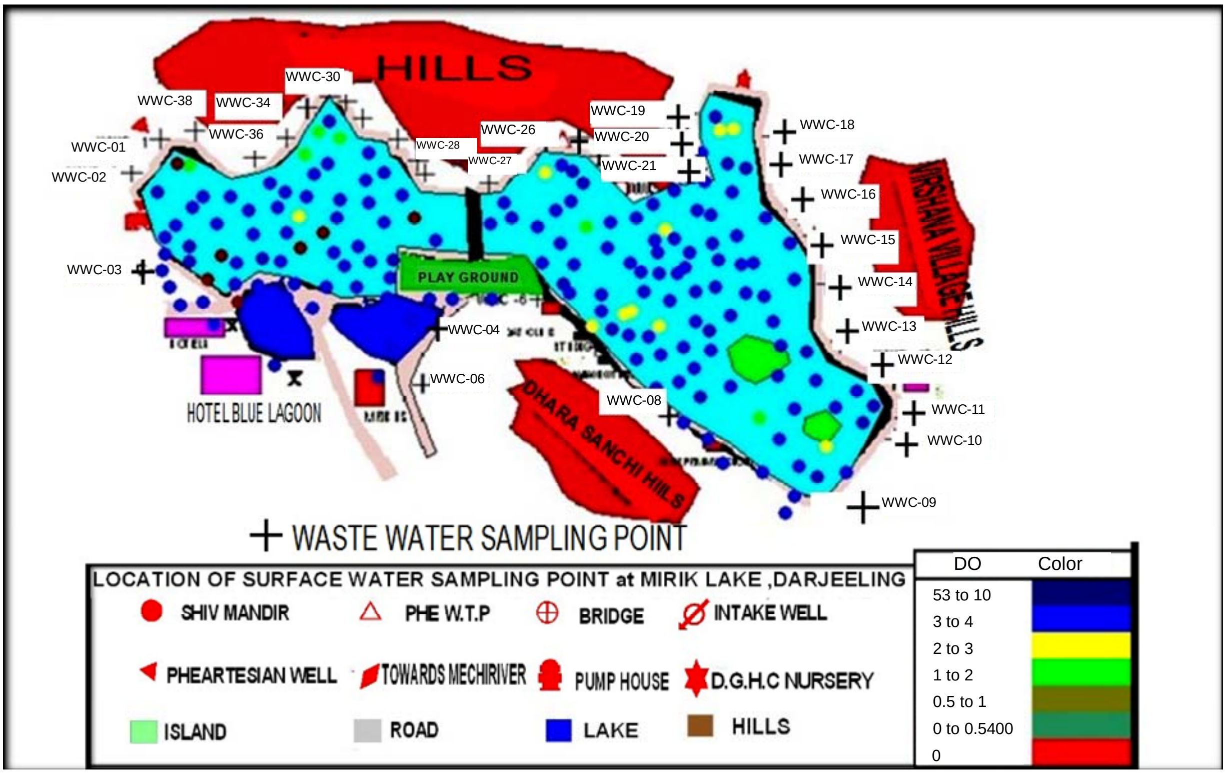

Figure 7. Thematic map of the DO concentration generated from the identified relationships between the land use, wetland bed and the quality parameter.

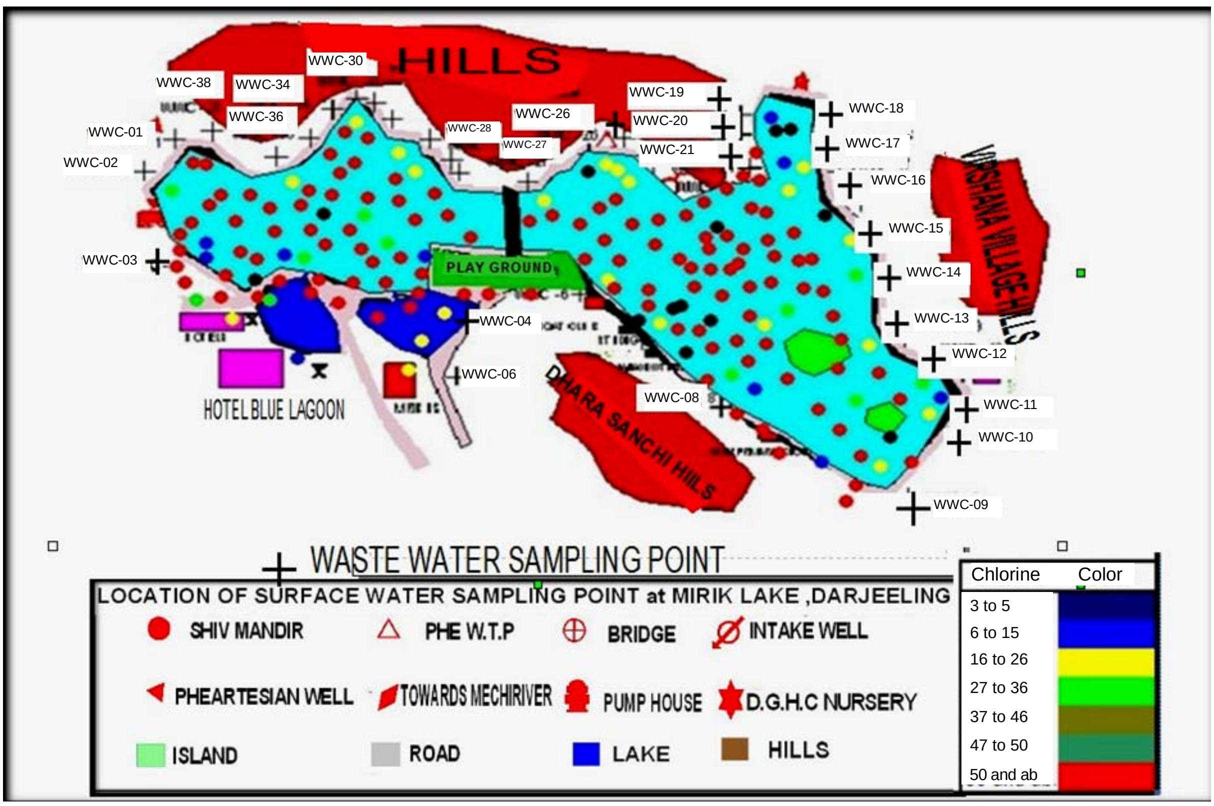

Figure 8. Thematic map of the Cl concentration generated from the identified relationships between the land use, wetland bed and the quality parameter.

the lake are also identified as an influencing factor on the quality of wetland water.

4. Conclusion

The present study investigates the relationship between water quality parameters and adjacent land use and the aquatic plants of the wetland with the help of neuro-genetic models. The model results showed that the submerged shrubs along with road and hills within the 500 m of the lake have a quantifiable relationship with the water quality of the lake. The DO was found to be least source of problems, and the BOD was found to be a highly correlated parameter with the inputs among the considered four parameters, which the study has assumed to be representative of overall quality of the wetland. The ecotourism activities, which are common in and around the lake due to the geo-morphology and biodiversity of the region was also found to be affecting the quality of the lake water. That is why the anthropogenic impacts coming from both tourist and local population on the quality of water may be considered as the next objective for the overall economic and environmental sustainability of the wetlands, which is treated as the major source of income of the local population. The identification of the correlation can be performed for other lakes also to obtain a better conclusion and generalization of the observations. The neuro-genetic models were found to be suitable for the identification of relationships, which is eminent from the low MSE achieved by all the models trained with different training algorithms.

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Journal of Water Resource and Protection, 2012, 4, 590-596
doi:10.4236/jwarp.2012.48068 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Nitrogen Use Efficiency under Different Field Treatments on Maize Fields in Central China: A Lysimeter and ^{15}N Study

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Received May 7, 2012; revised June 17, 2012; accepted June 23, 2012

ABSTRACT

Nitrogen loss from farmland has caused serious problems all over the world. This field study assessed Nitrogen Use Efficiency[1] (NUE) and biomass yield under four different field treatments in the Hubei Province, in central China. Results show that 1) in these four treatments, the maize monoculture plots have the highest rate of fertilizer N losses (69.12%), and the lowest (32.45%) is treated by surface rice straw mulch of maize intercrop with peanut; 2) compared with monoculture, polyculture plots have 36.9 $kg \cdot ha^{-1}$ and 26.57 $kg \cdot ha^{-1}$ more nitrogen absorption in the mulched and un-mulched plots respectively, however, polyculture has a lesser effect on NUE; 3) surface straw mulch is an effective way to keep nitrogen in the soil (0 - 100 cm), however it may decrease dry matter yield in monoculture plots; 4) maize intercrop with peanut and surface mulch can keep 47.63% of the fertilizer N in the soil profiles (0 - 100 cm), which is the highest among these four treatments.

Keywords: Monoculture; Surface Mulch; Nitrogen Use Efficiency; Leaching; Biomass Yield

1. Introduction

Nitrogen is one of the essential elements for plant growth, as it is not only promotes plant growth but also acts as a building block for protein. In order to increase yield, fertilizer consumption has continued to increase across the world since the 19th century. The global production of fertilizer has increased from 27.4 million tons in 1960 to 143.6 million tons in 1990, and it will rise further to 208 million tons in 2020 [1]. Because of low NUE, the more fertilizer N applied, the more nitrogen was lost. Only 30% - 35% of the fertilizer N was taken up by plants and about 20% - 50% went away through leaching and runoff [2-4]. Nitrogen lost from farmland is the main source of non-point source pollution for water systems, causing problems of groundwater nitrate pollution, surface water eutrophication, and natural ecological degradation.

Researches into nitrogen losses from agricultural activities commenced several decades ago, founding a consensus that nitrogen losses from agricultural land is the main source of water NO_3^- contamination around the globe [5]. Nitrogen loss reduction strategies such as manure fertilizer [6,7], fertilizer application methods [8], environmental policies [9,10], surface mulching [11], tillage/irrigation skills [12,13] (Meek, *et al*., 1995; Turpin, *et al*., 1998) and proper intercropping system [14] have been well documented in existing researches. However, most of the studies were focused on nitrogen losses or NUE, with less research being conducted on nitrogen losses reduction with the yield consideration [15]. Nitrogen pollution mitigation strategies without yield consideration cannot be implemented in China, because most farmers small-scale and therefore pursue high yields to support their families.

In this paper, we used on-site lysimeters and a stable isotope ^{15}N urea to compare the nitrogen distribution, NUE and biomass yield of four different field treatments in central China. We first analyzed nitrogen distribution and yield under different treatments, and then examined the ^{15}N rate in the soil and crops to determine NUE. Finally, we discussed the field treatments which may improve NUE and reduce nitrogen losses without sacrificing the yield.

2. Materials and Methods

2.1. Site Description

Lysimeters and a ^{15}N enriched urea were used in this

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[1]Nitrogen Use Efficiency in this research is defined as the part of the applied fertilizer nitrogen which is found in the plant.

experiment, established in Zhijiang City, in the western part of Hubei Province (central China) (N43.715635, E87.251374). The region has a moist monsoon climate with a mean annual temperature of 16.5˚C, and a mean annual rainfall of 1032.7 mm. The soil in the experiment field is yellow-brown, and the properties of the surface soil (0 - 20 cm) are listed in **Table 1**.

2.2. Treatments

The experiment was a split-plot factorial design with two factors and three replicates. Three lysimeters are located in each of the plots, (with the plots being 5 m wide ∗ 8 m long), and the lysimeter is 1.5 m wide ∗ 1.5 m long ∗ 1.3 m deep, leaving an edge of 0.3 m lysimeter above the soil surface when put into the field. The four treatments are: (1) maize monoculture (C); (2) maize intercrop with peanut (C + P); (3) maize with rice straw mulch (C + M); (4) maize intercrop with peanut and rice straw mulch (C + P + M). There are two rows of peanut between two rows of maize. In the monoculture plots, the plant density is 36,000 maize·ha^{-1}; in the polyculture plots, the plant density is 17000 maize·ha^{-1} and 180000 peanuts·ha^{-1}. Only urea is applied to the maize, and no nitrogen fertilizer is applied to peanut. 276 kg N·ha^{-1}, 252 Kg K·ha^{-1} and 126 kg P·ha^{-1} were applied in the monoculture plots; 130 Kg N·ha^{-1}, 252 Kg K·ha^{-1} and 126 kg P·ha^{-1} were applied in the polyculture plots. According to the farmers' conventional methods, the urea were applied twice, with 111 Kg N·ha^{-1} (monoculture) and 52 Kg N·ha^{-1} (polyculture) at planting as a basic fertilizer, and with 165 Kg N·ha^{-1} (monoculture) and 78 Kg N·ha^{-1} (polyculture) as a top-dressing when the maize plants reach the stage of two fully expanded leaves. All of the potassium and phosphorous was applied at once in the first time. The basic fertilizer was applied on 5 May, 2008, at the same time of transplant maize and sowing peanuts; and the top-dress was applied on 31 May, 2008. All of the crops in the lysimeter received ^{15}N enriched urea, and ordinary P and K fertilizers. The abundance of the ^{15}N urea is 5.02%.

2.3. Sample Collection and Lab Analysis

In order to determine the nitrogen assimilated by the crops, all of the crops were harvested including roots (0 - 20 cm). Crop samples were separated to grain and stem. Subsequently, all of the samples were dried at 70˚C until constant weight, and then crushed to powder until passing a 0.15 mm sieve, waiting for Total Nitrogen Concentration (TNC) and ^{15}N abundance analysis.

The drainage water sample of each lysimeter was collected whenever drainage occured, stored with dark glass bottles in the refrigerator at 4˚C, and then returned to the laboratory for TNC analysis. Unfortunately, ^{15}N abundance in leached water hadn't been analyzed; therefore, fertilizer N deficit in this research includes gaseous and water losses.

After the crops were harvested in August, soil samples in each plot were collected from a depth of 0 - 20, 20 - 40, 40 - 60, 60 - 80 and 80 - 100 cm. The mass of TNC and fertilizer utilization was calculated after considering the bulk density of different soil layers.

2.4. Methodology in Lab

1) Water samples: filtered and sent to the lab for TNC analysis on an Alpkem Flow Solution IV auto-analyzer.

2) Plant tissues: TNC in grain and stem of the sub-samples were determined by the micro-Kjeldahl method by digesting the sample in H_2SO_4-H_2O_2 solution. The crop samples that waited for ^{15}N were solute as the TNC method, and the solute samples were analyzed by using isotope mass spectrometer detector (ANCA-SL/20-20).

3) Soil samples: 10 g of the sub-samples were placed in a 100 ml 2 N KCl, shaken for 1min and allowed to equilibrate for 18-24 hrs. Supernatant was removed and stored at 4˚C. The TNC in the supernatant was measured colorimetrically on the Lachate auto-analysis system. The ^{15}N in the supernatant was determined by an isotope mass spectrometer detector (ANCA-SL/20-20).

3. Results and Discussion

3.1. Nitrogen in the Soil

TNC in the soil layers were varied among different treatments. **Figure 1** shows that in terms of the TNC change trend in the soil of 0 - 100 cm, there were steady decreases in the two plots which were treated by monoculture; however, it decreased slowly in the two plots treated by polyculture, especially in the plot of C + P + M which increased in the layer of 0 - 60 cm and then decreased sharply. The highest TNC in the surface layer (0 - 20 cm) is the plot of C + M which reached up to 1514 kg/ha, the other three plots were approximately in the range of 900 kg·ha^{-1}.

Straw mulch cannot only keep nitrogen in the surface

Table 1. Soil properties before experiment.

pH	Organic Matter (g·Kg^{-1})	CEC (mmol·kg^{-1})	Available N (mg·Kg^{-1})	Extractable P (mg·Kg^{-1})	Exchangeable K (mg·Kg^{-1})	Total N (g·Kg^{-1})	Total P (g·Kg^{-1})	Total K (g·Kg^{-1})
6.27	9.49	10.2	9.67	3.0	72.3	0.21	0.16	8.65

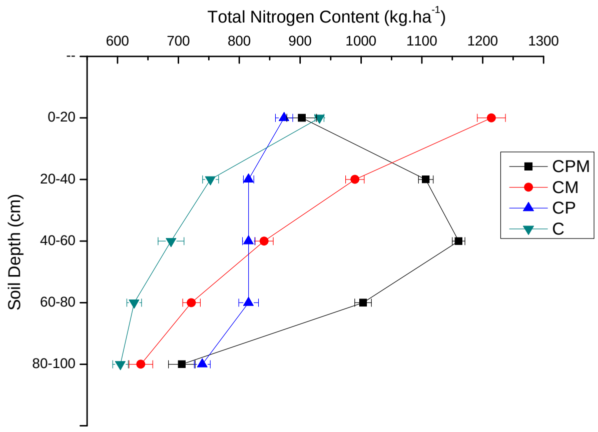

Figure 1. Total nitrogen retention in the soil (0 - 100 cm) after harvest.

layer of the soil, but also can improve the yield and water use efficiency [16]. Alexandra found that a mulch-based cropping system could increase TNC in the surface soil layer (0 - 30 cm) in the long-term. Similar results were found in this experiment, with the TNC of the surface soil layer (0 - 60 cm) in the mulched plots being higher than the un-mulched ones. However, there were no significant differences in the deeper soil layer (80 - 100 cm) among these four treatments. Conversely, the plots treated by C + M had higher TNC in the soil layer of 0 - 20 cm. This may be due to the straw being decomposed 3 months after mulching; however, maize cannot utilize surface nitrogen because of its deeper root and less rainfall at that time.

In summary, mulch and polyculture are the two treatments keeping nitrogen in the surface soil layer (0 - 60 cm) which provide more nutrients for the crops of next season. However, nitrogen accumulation in the soil is regarded as a potential danger for the water system, because it is leached out when the rainy season arrives. Its termed as a "memory effect" in [17]. Therefore, C + P + M is suitable for intensive agriculture, because it can provide more nutrients for the following season with less fertilizer N consumption.

3.2. Nitrogen in the Leachate

With regards to TNC in the leachate, there were no obvious differences between the four treatments, **Figure 2**. After basic fertilizer was applied, there was no drainage water in the lysimeter until 6 June. At beginning, the TNC in the leachate from the two plots treated by mulch were a little lower than the other two plots.

During the whole cropping period, the peak of the TNC in the leachate occurred three months after the basic fertilizer was applied, and reached up to more than 7 $kg \cdot ha^{-1}$.

There are two reasons which might explain this phenomenon. Firstly, plants do not consume too much nitrogen after the growing period, therefore the nitrogen in the root zone will be leached. Secondly, as [18] described, the period which is most prone to leaching is autumn, because during that time evaporation decreases and soil moisture increases, soil microbial activities increase, and there is an increased mineralization of organic nitrogen, which cause more nitrogen to be leached.

Two months after the basic fertilizer was applied (30 June), the TNC in the leachate was at the bottom. We consider this to be because two months after maize being planted is the fast growing period, with nitrogen being rapidly taken up by the crops, and the amount of the fertilizer nitrogen leached during this season is normally low [19].

3.3. Biomass Yield and Nitrogen Absorption

The crops' nitrogen absorption in the plots which were treated by polyculture was much higher than in the monoculture plots. The highest nitrogen absorption by the crops was the treatment of C + P + M, reaching up to 124 kg/ha; the lowest was C + M, which only recorded 87.6 kg/ha. As for monoculture, un-mulched plots had higher nitrogen absorption than mulched plots, which is similar to the results of other research (Wang Wei-Ming, 1986). The main reason is that straw has a high C/N content which may cause nitrogen immobilization. Therefore available nitrogen in the plots of C + M is not sufficient for the crops' growth. However, the intercropping plots had the opposite results; with the reason being that the nitrogen fixed by the peanut is not **Figure 3** shows that the crops' nitrogen absorption in the plots which were treated by polyculture were much higher than in the monoculture plots. The highest nitrogen absorption by the crops was the treatment of C + P + M, reaching up to

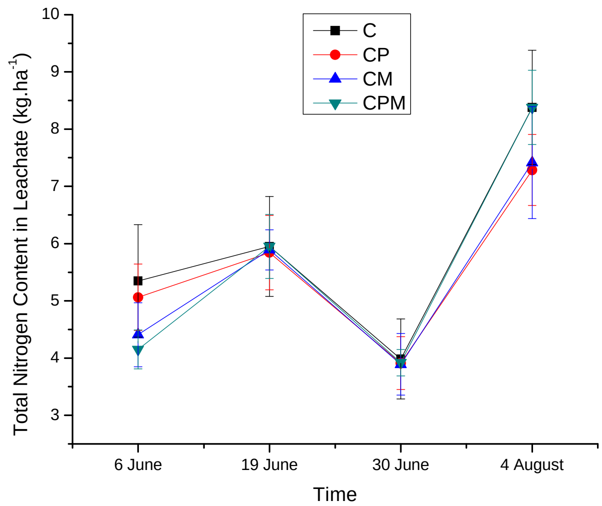

Figure 2. Nitrogen leaching from farmland during grow season.

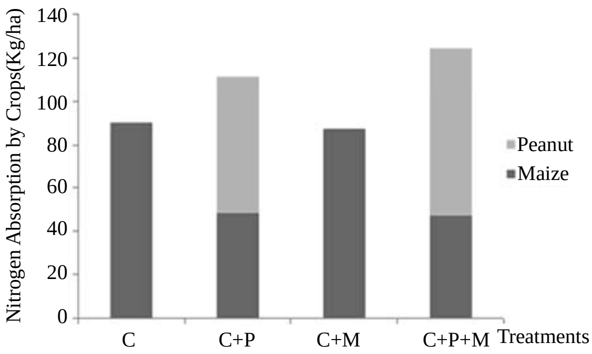

Figure 3. Nitrogen absorption by crops after harvest.

124 kg/ha; the lowest was C + M, which only recorded 87.6 kg/ha. As for monoculture, un-mulched plots had a higher nitrogen absorption than mulched plots, which is similar to the results of other research (Wang Wei-Ming, 1986). The main reason is that straw has a high C/N content which may cause nitrogen immobilization, therefore available nitrogen in the plots of C + M is not sufficient for the crops' growth. However, the intercropping plots had the opposite results; with the reason being that the nitrogen fixed by the peanut is not only used by the crops but also by the microorganism. Therefore the number of microorganisms in the soil explodes in a short time, and the microorganism can decompose the mulched straw which will provide more nitrogen resources for the crops that may contribute to the yield enhancement by intercropping [20].

Compared to monoculture, the intercropping system contributes greatly to crop production through its effective utilization of resources [21,22]. This research produced the same results, with crops absorbing more nitrogen in plots which were treated by polyculture. This is because legumes can fix nitrogen from the air and pass it to the cereals which are intercropped with them [23-25]. However, if it is not handled properly, polyculture will fail to work better than monoculture. For example, when maize is intercropped with ryegrass, they not only showed weaker growth but also took up smaller amounts of nitrogen than plant maize alone [26].

4. Nitrogen Recovery

Fertilizer N utilization by crops and retention in the soil were calculated as Equations (1)-(6):

$$\%\ \text{utilization of added fertiliser} = \frac{\text{Amount of nutrient in the plant derived from the fertiliser}}{\text{Amount of nutrient applied as fertiliser}} \times 100 \quad (1)$$

$$\%\ \text{Ndff} = \frac{\text{atom}\ \%\ ^{15}\text{N excess}_{(\text{plant/soil/water})}}{\text{atom}\ \%\ ^{15}\text{N excess}_{(\text{Fertilizer})}} \times 100 \quad (2)$$

$$\text{DM yield}(\text{kg/ha}) = \text{FW}\ (\text{kg}) \times \frac{10000\left(\text{m}^2/\text{ha}\right)}{\text{area harvested}\ \left(\text{m}^2\right)} \times \frac{\text{SDW}(\text{kg})}{\text{SFW}(\text{kg})} \quad (3)$$

$$\text{N yield}(\text{kg/ha}) = \text{DM yield}(\text{kg/ha}) \times \frac{\%\text{N}}{100} \quad (4)$$

$$\text{Fertilizer N yield}(\text{kg/ha}) = \text{N yield}(\text{kg/ha}) \times \frac{\%\text{Ndff}}{100} \quad (5)$$

$$\%\ \text{fertilizer N utilization} = \frac{\text{Fertilizer N yield}}{\text{Rate of N application}} \times 100 \quad (6)$$

where:

Ndff—Fraction of N in the plant derived from the ^{15}N labeled fertilizer.

FW—Fresh weight per area harvested.

SDW—Subsample dry weight.

SFW—Subsample fresh weight.

DM—Dry matter Yield.

4.1. Fertilizer N Utilized by Crops

Considering fertilizer N utilization by crops, C + P had the highest NUE (reaching up to 24.38%), while maize monoculture had the lowest NUE of only 18.36%. Because competition exists between the two crops which are planted together, the intercropping system has efficiency more efficient use of natural resources [27-29].

In the two plots which were treated by monoculture, straw mulch increased the NUE, however the plots treated by polyculture displayed the opposite trend. In the polyculture plots, mulched plots had a higher nitrogen absorption yet lower NUE. This may be because peanuts can fix the nitrogen, therefore the system has enough nitrogen resources, and mulched straw can be decomposed fast and provides more nitrogen resources for the crops (which can cause lower NUE). In the monoculture plots, surface mulch can reduce the fertilizer nitrogen percolation and volatilization, which may improve NUE; therefore, mulched crops have higher NUE.

4.2. Fertilizer N Retention in the Soil

When maize is intercropped with legume crops, nitrogen content in the soil profiles will improve significantly [30]. In our experiment, 47.63% of the fertilizer N remained in the soil (0 - 100 cm) after harvest in the plots of C + P + M, which was the highest amongst these four treatments. However, maize monoculture plots have the lowest fertilizer N soil retention (only 12.52%). This is because legume crops can improve soil fertility through biological nitrogen (N) fixation [31].

Straw mulch can increase the soil fertilizer N retention, NO_3^- content and improve soil fertility after harvest [32]. Results show that the fertilizer N soil retention in the polyculture plots which were treated by mulch and unmulch were 47.634% and 30.69% respectively; in the monoculture plots, the figures were 23.06% and 12.52% respectively.

Table 2. Dry matter yield and nitrogen absorption by crops under different field treatments.

Treatments	Fertilization (kg/ha)			Dry Matter (kg/ha)		Nitrogen Absorption (kg N/ha)
	N	P	K	Biomass	Grain	
C						
Maize	276	126	252	16253.3 ± 150.3	6540 ± 62.3	90.3 ± 7.5
C + P						
Maize	130	126	252	4933.3 ± 73.4	1951.11 ± 13.5	48.5 ± 3.6
Peanut	0			6997.0 ± 38.1	1680.88 ± 19.1	62.9 ± 5.8
C + M						
Maize	276	126	252	15573.3 ± 128.8	5720 ± 41.6	87.6 ± 5.2
C + P + M						
Maize	130	126	252	5195.6 ± 61.2	2044.44 ± 18.3	47.5 ± 3.6
Peanut	0			4786.6 ± 37.4	1344.3 ± 15.9	77.1 ± 5.4

Table 3. Nitrogen fertilization utilization among different treatments.

Treatments	Total N (kg/ha)	Fertilizer N (kg/ha)	Fertilizer utilization (%)
C			
Crop	90.3 ± 7.5	50.67 ± 6.2	18.36
Soil	3644.48 ± 161.5	34.55 ± 4.8	12.52
Water	22.93 ± 8.1	-	-
Deficit			69.12
C+M			
Crop	87.61 ± 5.2	65.2 ± 4.5	23.62
Soil	4804.8 ± 159.6	63.64 ± 8.7	23.06
Water	22.25 ± 5.2	-	-
Deficit			53.32
C+P			
Crop	111.53 ± 9.4	31.7 ± 2.1	24.38
Soil	4058.88 ± 173.8	39.9 ± 5.7	30.69
Water	22.01 ± 4.5	-	-
Deficit			44.93
C+P+M			
Crop	124.51 ± 9.0	25.9 ± 1.9	19.92
Soil	4878.72 ± 168.2	61.92 ± 7.1	47.63
Water	22.33 ± 3.2	-	-
Deficit			32.45

4.3. Fertilizer N Losses

Results show that polyculture is one of the most effective ways to reduce fertilizer N losses. The fertilizer losses in the plots treated by C + P + M and C + P were 32.45% and 44.93% respectively. However, in the plots treated by C and C + M they were 69.12% and 53.32%.

Surface straw mulch can reduce the fertilizer N losses effectively, and in our research it reduced 15.8% and 12.48% of fertilize N losses in the monoculture and polyculture plots respectively. This is because surface straw mulch can reduce soil evaporation and retain soil moisture, which may increase the yield and reduce nutrient losses [33-35].

5. Conclusions

According to the results and discussions above, we can confidently draw the following conclusions:

Maize intercrop with peanut is an effective way to reduce fertilizer N losses, increasing the nitrogen absorption by crops and fertilizer N retention in the soil profiles (0 - 100 cm); however, it has a lesser effect on NUE.

Compared with un-mulched plots, surface rice straw mulch can reduce nitrogen losses and keep nitrogen in the root zone area (0 - 100 cm), however, it should be used in intercropping systems because it may sacrifice the crop dry matter yield in the first season when used in maize monoculture.

6. Acknowledgements

This study was supported by Key Project of Nation Spark Program of China (201176GA0009) and National People's Livelihood Science and Technology Plan of China (2011MSB05007).

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Journal of Water Resource and Protection, **2012, 4, 597-604**
doi:10.4236/jwarp.2012.48069 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Development of El-Salam Canal Automation System

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Received February 24, 2012; revised March 26, 2012; accepted April 7, 2012

ABSTRACT

In Egypt irrigation water is becoming more scarcer with the continuously increasing demand for agriculture, domestic and industrial purposes. To face this increasing irrigation demand, the available water supply in Egypt is supplemented by the reuse of agricultural drainage water as in El-Salam Canal that do not satisfy water quality standards defined for the canal. This paper introduces an automation system for El-Salam Canal to control the flow of the fresh water and drainage water supplied to the canal. This automatic control system (ACS) is able to process data of various flows and water quality data along the canal. This control system is represented by a canal computer model. This system computes the required control actions at the Damietta branch and the feeding drains. It is also able to generate optimum solutions for the canal to satisfy the pre-defined canal conditions and standards.

Keywords: Water Quality; Automatic Control; Modeling

1. Introduction

As water is becoming more and more a scarce resource all over the world, proper management of the available water is essential. For an optimal use of the available water resources, water management strategies have to be developed. A water management strategy is based on a water control system. The two main factors that determine the designated water use are the water quality and water quantity of a water system. Controlling the quality and quantity of a water system is done using monitoring devices, water gates, pump stations, power stations and other operational devices. There are different types of controlling a water system. However, the use of automatic control has lately proven to have more advantages over other types. Automatic control provides accuracy, reliability, time-saving and man-power saving. It also enhances flexibility and saves water and improves production.

Many researches have been conducted for implementtation of automatic control water systems. [1] studied the real-time control of combined surface water quantity and quality for polder flushing. [2] studied the Elements of a decision support system for real-time management of dissolved oxygen in the San Joaquin River Deep Water Ship Channel. In Thailand, on the Kamphaengsaen Irrigation Canal, the canal's automation system has been developed and tested during October 2006 to July 2008. The canal automation system consists of the master station and six remote terminal units (RTU) which communicate by VHF radio. The six RTUs installed in the canal irrigation system are for monitoring and controlling of water levels and discharges in the canal system, monitoring rainfall, air temperature and relative humidity. The system has provided flexible, accurate and reliable control of irrigation water supply [3]. In Arizona USA, on the Salt River Project Canal system an automatic control system was proposed. This system automates and enhances functions already performed by operators. Some of these functions are control of water levels and flow control at check structures. The proposed system consists of three separate controllers with a configuration that makes control actions computed independently of gate hydraulics. The controllers are centrally operated, that is monitoring and determining control actions is done from a remote site. The control system has proven to be a stable and robust system [4]. In Australia, on the Coleambally Canal Network, an automation system has been introduced, with the objective of reducing the operating cost of the canal system, reducing conveyance losses and improving the ability of the supply system to respond to irrigation demands. There is an ability to remotely monitor and regulate the main canal which results in a much improved standard of service to the secondary canal off-takes. Gates are being automated and a software system controls the opening and closing of the gates automatically. The control system assists irrigators to improve the efficiency of water use [5].

This study focuses on introducing El-Salam Canal control system that consists mainly of an automatic monitoring system and an automatic control system which is represented by a computer control model based on a data driven model.

2. Study Area Description

El-Salam Canal is located in the North East of Egypt where it supplies water for the reclamation of new lands in that part of the country. These areas are originally parts of the sedimentary formation of the ancient Nile branches in that area. The canal intake is on the right bank of Damietta Branch at Km 219, 3.0 Km upstream the Faraskur Dam. The canal passes through five governorates: Damietta, Dakahliya, Sharkiya, PortSaid and North Sinai [6], the total length of the canal is about 277 Km and is divided into two main parts. The first part is West of Suez Canal, it is about 86 Km long and the second part lies east of Suez Canal and is about 191 Km long. The western part of the canal is known as El-Salam Canal. It starts from the intake at Damietta Branch (Nile River) runs in a south-eastern direction and crosses the Suez Canal through a siphon, it continues after the siphon and the eastern part of the canal is known as El-Sheikh Gaber Canal. A layout of El-Salam Canal is shown in **Figure 1**. El-Salam Canal was designed to supply the irrigation water to a total area of 620,000 feddans consisting of 220 thousand feddans on the western side of the Suez Canal and 400 thousand feddans east of the Suez Canal in Sinai. The canal was planned to convey a discharge of 4.45 billion m^3/year. About 2.2 billion m^3/year would be fresh water supplied from the Nile and transferred through the canal at its intake. And about 2.25 billion m^3/year is to be supplied from two drains called Bahr Hadous and Lower Serw drains. The water quality represented by salinity was also a concern when designing the canal.

Salinity should not exceed 1250 ppm generally in the canal. Many structures are constructed along El-Salam Canal. The first group of these structures is for water regulation purposes, consisting of pump stations and regulators. The second group of structures is crossing structures such as siphons and bridges.

Some of the objectives and benefits that are gained from implementing El-Salam Canal are: redistributing population in Egypt, protecting the eastern borders of the country, strengthening the Egyptian agricultural policy through increasing the cultivated areas and agricultural yield, increasing agricultural and national production and thus increasing exporting vegetables and fruits while decreasing food import, benefiting and making good use of agricultural drainage water as an important water resource, creating work opportunities for the youth and establishing tourism, industrial and mining projects.

Therefore, careful investigation and prediction of the quality of water throughout the canal is crucial. Many studies have been carried for assessment of the water quality of Bahr Hadous and El-Serw drains, [7-11], also many studies have been conducted about the agriculture development of El-Salam water [12-16], and few studies were conducted to study the water quality along El-Salam

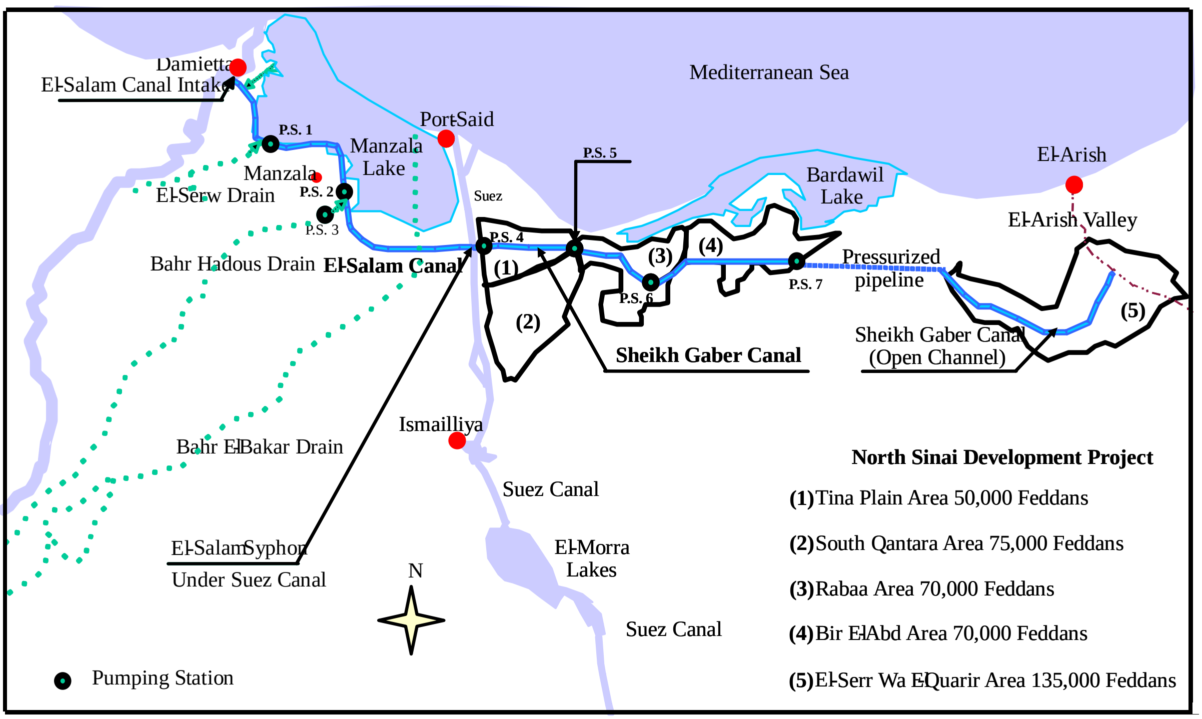

Figure 1. Layout of El-Salam Canal project.

Canal, [17-22] developed a decision support system (DSS) to choose the required treatment option of discharging drains in order to satisfy with these guidelines but little attention has been for real time operational water quality management of the canal [23,24].

3. Computer-Aided Control System for El-Salam Canal

The Control System on El-Salam Canal integrates the water quality monitoring and the water quality control policy using:

- An automatic monitoring system (AMS), which is capable of collecting data of different flows and water quality along the canal.
- An automatic control system (ACS), which is able to process data of various flows and water quality data along the canal. This control system is represented by a computer model designed for the canal.

This computer model is able to generate optimum solutions for the canal to satisfy the pre-defined canal conditions and standards. The model can also compute the required control actions at the Damietta branch and the feeding drains which supply the canal with its water. It calculates the gate opening required for each mixing drain.

3.1. The Automatic Monitoring System (AMS)

The type of automatic monitoring system used consists of a Data Acquisition System (DAS) which runs a data software collection platform (DCP). This DAS includes at each local station:

a) A Data Collection Unit (DCU)
b) A Data Terminal Unit (DTU)
c) Computer Control Model

The DCU collects data from sensors and is triggered by the DTU, whereas the DTU is the part that triggers the DCU and sends data to the computer control model at the main station [7,8]. The communication equipment is installed at each DTU and at the main station. The communication system also supports voice communication between any two stations. The facilities of the voice communication system include telephone, earpiece and mouthpiece. To fulfill web communication, a web-enabled software is introduced to the control system at the main station to support remote monitoring and viewing of databases for station details, historical and actual data through the internet. In case of failure of the automatic system that sends the control actions from the main station to all the DTUs of all stations, the data communication system delivers the control actions to the concerned stations in the form of messages. These messages are displayed on the DTU for the managing of the station manager and the operators. Upon the reception of a message, alerting devices like a horn and a flashing light are automatically activated through digital signals delivered to the DTU. All electrical devices are connected with cables to deliver power and to transport signals and data. Cable guidance tubes, ducts and similar connections are used to give the cables proper protection.

3.2. Description of the Automatic Real-Time Control System (ARTCS)

- The supply, transport and distribution of the irrigation water are managed through real-time control of the structures on El-Salam Canal. The structures which we consider in this study are:
- The head regulator at Damietta Branch admitting fresh water from the Nile.
- The regulators at the Lower Serw drain admitting drainage water from the agricultural drain.
- Pump station No. 3 lifting water from Bahr Hadous drain to El-Salam Canal.

3.2.1. Automatic Real-Time Control System Features

The ARTCS system is based on:

- Full utilization of the available fresh Nile water with a water quantity control at the rest of the intakes to El-Salam Canal.
- Presence of instantaneous information available on the actual flow of the drains and of Damietta Branch feeding El-Salam Canal.
- Presence of instantaneous information available on the salinity of the drains and of Damietta Branch feeding El-Salam Canal.
- The difference between the actual value (measured) and the setpoint (desired output response) is checked every suggested period (e.g. 30 minutes) and control actions are calculated by the controller. Those actions are automatically communicated and act on the actuators that execute the control actions physically causing the operation of the gates and pump stations as desired.

Thus the automatic real-time control system fulfills the following functions:

- Receiving the measured data once every 30 minutes.
- Processing data and comparing it with setpoint values
- Computing required actions by pump stations and gates.
- Communicating these actions to the needed gates and pump stations and operating them as desired.

3.2.2. Control Method Description

The computer model is installed at the main station. It includes the software that receives the monitored data from the DTU and makes all the necessary computations (processing of data). It then gives an output of control actions that are sent back to the DTUs of all stations .In the computer control model, the control method that is applied is called the Master-Slave controller. The Master controller determines the flows that need to be applied at the control structures (Damietta Branch, El-Serw drain and Bahr Hadous drain), while the Slave controller of each structure converts the flow to a local setting of the structure. As the Slave controller receives information from the Master controller about the flow change that the concerned structure has to implement, it converts this flow change to a change in the opening height of the gates or in a change of the pump flow by the following relationship (Equation (1)):

$$U = f(Q) \tag{1}$$

where:

U: structure setting (gate opening or pump flow)

Q: flow through the structure

Slave controllers use upstream and downstream water levels (h) around the structure in this formula. A detailed explanation of this formula is given earlier in chapter three.

3.3. The Automatic Control System (ACS)

The type of control system used is the "multivariable closed-loop water management control system with disturbance and feed forward monitoring". This control system is a combination of feedback control and feed forward control methods. Parts of the automatic control system are shown in **Figure 2** [8].

The computer control model represents the automatic control system used. This computer model is based on a data driven model. The data measured along El-Salam Canal over the years 2006 to 2008 are being used in this model.

3.3.1. Mathematical Background of the Computer Control Model

The basic equations governing El-Salam Canal are :

- **Mass Balance Equations: Equations (2) and (3)**

$$Q_t = Q_{dam} + Q_{serw} + Q_{hadous} \tag{2}$$

$$Q_t * TDS_t = Q_{dam} * TDS_{dam} + Q_{serw} * TDS_{serw} + Q_{hadous} * TDS_{hadous} \tag{3}$$

- **Data Driven Equations: Equations (4) and (5)**

$$Q_{serw} / Q_{hadous} = R \tag{4}$$

$$OMR = (Q_{serw} + Q_{hadous}) / Q_{dam} \tag{5}$$

where:

Q_t = output discharge of El-Salam Canal (million m^3/day)

TDS_t = salinity at the output discharge of El-Salam Canal (ppm)

Q_{dam} = flow of Nile water at Damietta Intake (million m^3/day)

TDS_{dam} = salinity of Nile water at Damietta Intake (ppm)

Q_{serw} = discharge of El-Serw drain (million m^3/day)

TDS_{serw} = salinity of El-Serw drain (ppm)

Q_{hadous} = discharge of Bahr Hadous drain (million m^3/day)

TDS_{hadous} = salinity of Bahr Hadous drain (ppm)

R = measured ratio between discharge of El-Serw drain and discharge of Bahr Hadous drain

OMR = optimum mixing ratio of fresh water and drainage water

- **Flow-Gate Equation: Equation (6)**

From Bernoulli equation the following flow-gate Equation (6) is derived:

$$Q = c_d \cdot A\sqrt{2 \cdot g \cdot (h_1 - h_2)} \tag{6}$$

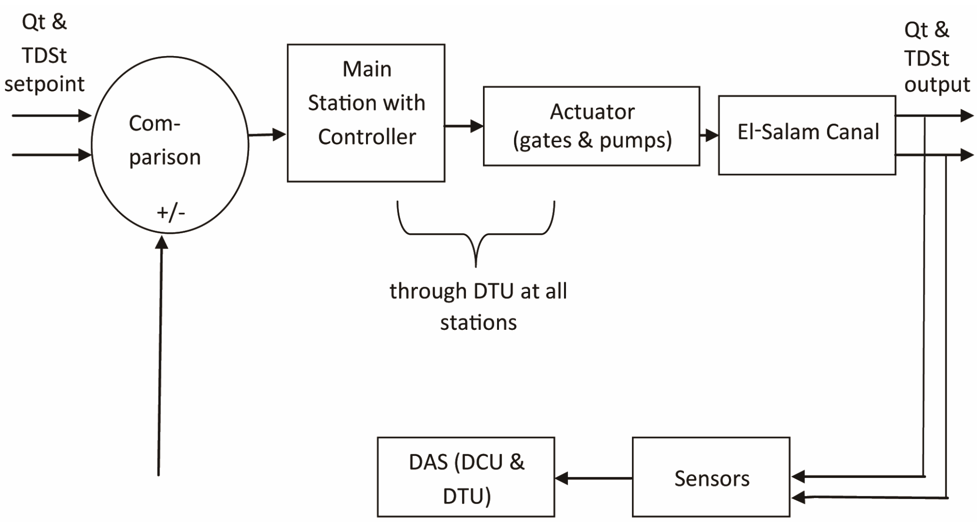

Figure 2. Design of the suggested automatic monitoring and control system for El-Salam Canal.

with

$$A = W_g \cdot Go$$

where:

Q = Discharge through the gated structure (m^3/s)

c_d = Overall discharge coefficient

A = Wetted area (m^2)

Wg = Gate width (m)

Go = Gate Opening height (m)

g = Gravity acceleration (m/s^2)

h_1 = Upstream water level (m)

h_2 = Downstream water level (m)

Constants: c_d = 0.6 – 0.65, W_g = 25 m for Damietta intake & 12 m for El-Serw drain, g = 9.81.

- **Flow-Pump Equation: Equation (7)**

$$NOP = Q/COP \tag{7}$$

where:

NOP = No. of Pumps

Q = Discharge needed to be pumped (m^3/s)

COP = Capacity of Pump (m^3/s)

Constant: COP = 16.5

It has been found from the data measured over the years 2006 to 2008, that the best scenario to be used to satisfy the specified conditions for El-Salam Canal is fully utilizing the available fresh Nile water (Damietta Branch) together with the optimum discharge of the available drains feeding El-Salam Canal (El-Serw drain and Bahr Hadous drain). Both fresh and drainage waters are mixed with an optimum mixing ratio. It has also been concluded that if the available fresh water (Damietta Branch) is greater or equal to half the required discharge of El-Salam Canal, then both fresh and drainage waters are mixed with mixing ratio 1:1 as designed and in that case this would be the optimum mixing ratio.

To satisfy the quantity and quality standards defined for El-Salam Canal, we have to calculate an optimum value of the drains discharges and an optimum mixing ratio between fresh water and drainage water. To do so, Equations (1)-(4) are solved in a numerical method. After the optimum values are calculated, control actions are computed using Equations (5) and (6).

4. Automatic Control System Implementation

In order to represent the optimum values of the feeding drains discharge, the optimum mixing ratios and the suitable control actions which satisfy the standards defined for El-Salam Canal, the model is run under different input discharges and different values of input water quality parameter (TDS) from Damietta Branch, El-Serw drain and Bahr Hadous drain. Data obtained through the years 2006 to 2008 represent the different scenarios that are chosen by the model.

4.1. Scenarios Analysis

Input values of discharge and TDS at the Damietta intake, El-Serw drain and Bahr Hadous drain are shown in **Figure 3**, and input values of upstream and downstream water levels at Damietta intake and El-Serw drain are shown in **Figure 4**. Iinput values of constants are shown in **Figure 5**. The input values are used by the model to define the control actions of water levels of the drains

Figure 3. Screen displaying the input discharge and TDS at feeding points along El-Salam Canal year 2007.

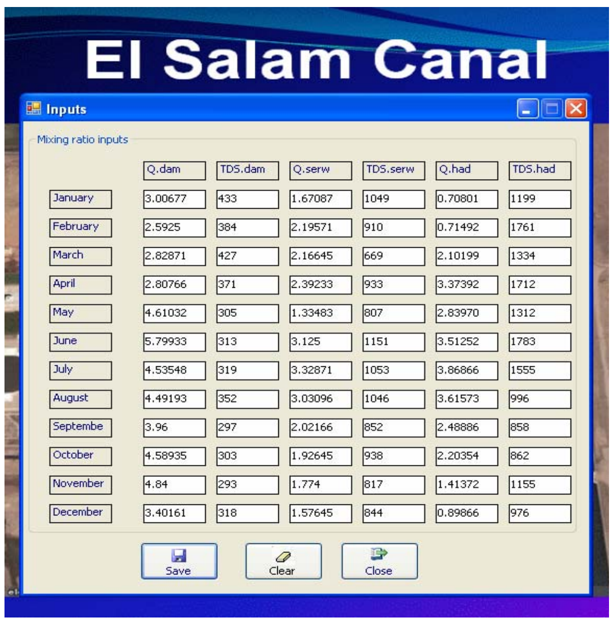

Figure 4. Screen displaying the input values of levels upstream and downstream water along El-Salam Canal year 2007.

Figure 5. Screen displaying the input values of pumps constants.

discharging into the canal and to calculate optimum values of drains discharges at the feeding points, an optimum mixing ratio the output discharge of El-Salam Canal together with the salinity at the output discharge of the canal.

The results of running different scenarios by the implemented computer control model are shown in **Table 1**. Output results of all scenarios presented in this study are displayed for certain months chosen as an example (February 2006, May 2006, July 2006, June 2008 and one assumed month). The table shows the control actions taken at Damietta Branch, El-Serw Drain and Bahr Hadous Drain concerning the gate opening and number of pumps are under different scenarios.

In **Figure 6**, the results of a run of the model for the selected month January 2007 chosen as an example are displayed. Values entered shown in **Figure 6** are used to compute the control actions that are required at the Damietta intake and the feeding drains. In **Figure 7**, the calculated control actions for the selected month January 2007 chosen as an example are displayed.

4.2. Analysis of Scenarios Outputs

In scenario 1 (June 2008), it is concluded that when the available fresh water (Damietta Branch) is greater or equal to half the required discharge of El-Salam Canal and the salinity at the output discharge of El-Salam Canal is within canal's standards, then the optimum mixing ratio between fresh and drainage waters will be 1:1 as designed. This will increase the discharge of El-Salam Canal to the required discharge (improve) and will maintain the salinity within the canal's standards 1).

In scenario 2 (January 2007), salinity at the output discharge of El-Salam Canal and the required discharge of the canal are within the canal's standards, thus the optimum mixing ratio between fresh and drainage waters will continue to be as measured.

In scenario 3 (July 2006), it is concluded that salinity at the output discharge of El-Salam Canal is within canal's standards and the output discharge of El-Salam Canal is increased to the required discharge (improve).

Table 1. Measured and calculated Data (GO & No. of Pumps) under different scenarios.

Chosen months	June 2008	Jan. 2007	July 2006	Feb. 2006	March 2007	Nov. 2007
Scenario name	Scen. 1	Scen. 2	Scen. 3	Scen. 4	Scen. 5	Scen. 6
GOdam[a] (original) meter	2.75	1.73	1.09	0.54	2.34	0.27
GOdam (calculated) meter	2.55	1.73	1.09	0.54	2.34	1.39
GOserw[b] (original) meter	0.34	2.4	1.08	0.61	1.34	1
GOserw[b] (calculated) meter	0.41	2.4	1.62	1.6	1.04	1.39
FChadous[c] (original) No. of pumps	3	2	2	1	2	1
PChadous[c] (original) No. of pumps	0.17	0.95	0.43	0.3	0.46	0.91
FChadous[d] (calculated) No. of pumps	3	2	3	3	1	2
PChadous[d] (calculated) No. of pumps	0.79	0.95	0.64	0.4	0.91	0.68
TDS total (measured)	841	994	1016	975	1622	1473
TDS total (calculated)	871	994	1142	975	1250	1099
Mixing ratio (measured)	0.77	1.54	1.2	1.14	3.73	3.73
Mixing ratio (calculated)	1	1.54	1.8	2.99	3.12	1

Where: [a]GOdam = gate opening at Damietta Branch; [b]GOserw = gate opening at El-Serw Drain; [c]FChadous = full capacity of pumps at Bahr Hadous Drain; [d]PChadous = partial capacity of pumps at Bahr Hadous Drain.

Figure 6. Screen displaying the original and calculated control actions at the feeding points along El-Salam Canal for January 2007.

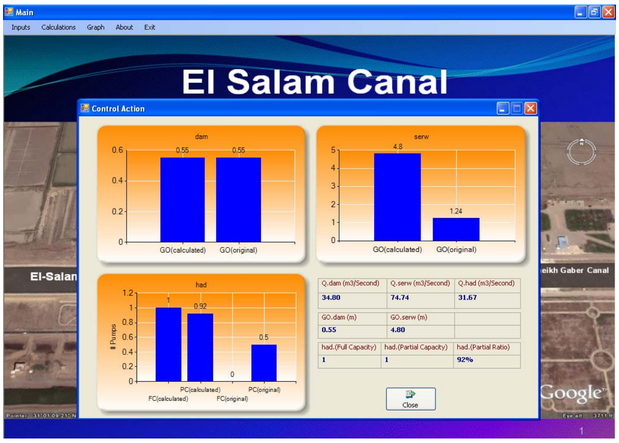

Figure 7. Screen displaying the original and calculated control actions at the feeding points along El-Salam Canal for January 2007.

In scenario 4 (February 2006), it is concluded that salinity at the output discharge of El-Salam Canal is decreased (improve) and the output discharge of El-Salam Canal is increased although not reaching the required discharge (improve).

In scenario 5 (March 2007), it is concluded that salinity at the output discharge of El-Salam Canal is decreased to the standard value (improve) and the output discharge of El-Salam Canal does not increase but may decrease, thus sacrifice with the discharge for the sake of the improved salinity of the canal.

In scenario 6 (November 2007), it is concluded that salinity at the output discharge of El-Salam Canal is decreased to the standard value (improve) and the output discharge of El-Salam Canal does not increase but may decrease, thus sacrifice with the discharge for the sake of the improved salinity of the canal.

In all cases, control actions are taken at the Damietta Branch, El-Serw drain and Bahr Hadous drain to fulfill all scenarios. On El-Salam Canal the gated intakes are at Damietta Branch and at El-Serw drain. The pumped intake is at Bahr Hadous drain. Thus the gated intakes use Equation (5) to calculate the control action needed (gate opening height) and the pumped intake uses Equation (6) to calculate the control action needed (no. of pump units required to operate).

5. Conclusion

Based on the results of this work, the following may be concluded that the computer-aided control system proposed in this paper could successfully monitor and control the flow of the fresh and drainage waters supplied to El-Salam Canal allowing variable mixing ratios. Also, mixing the fresh and drainage waters at the designed ratio 1:1 does not improve the value of the total output discharge except when using fresh water as half the required discharge of El-Salam Canal. Finally, fully utilizeing the available fresh water together with optimum discharge of drainage water has improved the total output discharge of El-Salam Canal and the salinity at the output discharge of the canal.

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Journal of Water Resource and Protection, 2012, 4, 605-615
doi:10.4236/jwarp.2012.48070 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Assessment of the Impact of Solid Waste Dumpsites on Some Surface Water Systems in the Accra Metropolitan Area, Ghana

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Received April 25, 2012; revised May 28, 2012; accepted June 7, 2012

ABSTRACT

Water samples from four water bodies that flow through some solid waste dump sites in the Accra metropolitan area of Ghana were analysed over a period of six months for Cd, Pb, Cu, Zn and Mn contents; coliform bacteria and helminth eggs. Other water quality parameters such as BOD, DO, suspended solids and turbidity were also assessed. Cd, Pb, Zn, Mn, and Cu were determined using flame atomic absorption spectrometry (FAAS). Faecal coliforms, total coliforms and helminth eggs were determined by the membrane filtration (MF) method. The water samples contain various levels of Cd, Pb and Mn; Zn and Cu levels were low and found to be below the detection levels of the instrument in most cases. Helminth egg counts in water samples were high; an indication that the water bodies were polluted with pathogens. It has been observed that the major sources of pollutants into the water bodies were organic waste as well as coliform bacteria derived from these waste dumps. The elevated levels of bacteria make the water bodies unsafe for both primary and secondary contacts.

Keywords: Landfills; Solid Waste; Dumpsite; Water Bodies

1. Introduction

Solid Waste Management (SWM) is a complex issue throughout the world. In developed countries the issues of SWM (collection, transportation and disposal) are well understood, accepted and workable. However, solid waste management is one of the many problems confronting many developing countries and recent events in major urban centres have shown that the problem of waste management has become too complex to handle and has seen dwindling efforts of city authorities, federal governments, state and professionals alike in addressing the issue [1].

N'dow, (1996) [2] pointed out that by the year 2000, half of humanity will be living in urban areas where most economic activities will take place and where most pollutants will be generated and natural resources consumed. The problem of waste in urban cities of Africa can be better understood in the light of rapid urbanization and for the first time in the history of mankind, we are witnessing an unprecedented phenomenon in the development of places of habitat making the balance of human settlement patterns shift from more people inhabiting rural areas to more people living in cities [3,4]. This is especially so in developing countries such as Ghana, Nigeria, Kenya and Mauritania. Whilst urbanization is not a new phenomenon in Africa, the current rate of uncontrolled and unplanned urbanization in Africa has given rise to a huge amount of liquid and solid waste being produced. So much is generated that these wastes have long outstripped the capacity of city authorities to collect and dispose of them safely and efficiently.

Based on an estimated population of 23 million and an average daily waste generation of 0.4 kg per person, Ghana generates annually about 3.0 million tons of solid waste [5]. The high population and its associated increase in urbanization and economic activities within the Accra Metropolis have made the impact of the society's solid waste very noticeable. The urban areas of the metropolis produce about 760,000 tons of Municipal Solid Waste (MSW) per year or approximately 2000 metric tonnes per day [5]. According to the Environmental Protection Agency (EPA) report, by 2025, this figure is expected to increase to 1.8 million tons per year or 4000 metric tons per day. The Accra Metropolitan Assembly (AMA) is solely responsible for municipal solid waste management in Accra and is able to collect through its private partners

1500 metric tons of municipal solid waste daily representing about 75% of solid waste generated. The remainder ends up at community dumps in open spaces, in water bodies, beaches and storm drainage channels. Only a small fraction of solid waste generated in the Accra metropolis is recycled mostly by the informal sector without any support from the authorities. This indicates an overwhelming dependence on landfillng as waste disposal option, the least preferred waste management option on the waste management hierarchy [6].

The design and optimization of solid waste management technologies and practices that aim at maximizing the yield of valuable products from waste as well as minimizing the environmental effects have little or no consideration in the African region. In the major cities of Ghana (Accra, Kumasi, Takoradi and Tamale) open dumps were the means of solid waste disposal. It was under the World Bank's Urban Environmental Sanitation Project that Ghana developed plans to build her first sanitary landfills in these four major cities [7]. The problem of solid waste management in the Accra metropolis has been characterized by single and ad hoc solutions such as mobilizing people to collect waste and de-silt choked gutters after a flood disaster or for an occasion, temporal allocation of waste contracts and dumping or building a central solid waste composting site.

It is a common site seeing water bodies flowing through most of these solid waste dump sites. Four prominent water bodies which are found flowing through some of these solid waste dump sites have been studied in order to ascertain the effects of the dump sites on water quality of these river sources. It is a known fact that virtually all water pollutants are hazardous to humans as well as lesser species. For example, sodium is known to cause cardiovascular disease while nitrates are involved in blood disorders. Mercury and lead are also widely known to cause nervous disorders. Some other contaminants are carcinogens while others for example, DDT is known to be toxic to humans and can also alter chromosomes. Others such as, PCBs cause liver and nerve damage, skin eruptions, vomiting, fever, diarrhea, and fetal abnormalities. These known effects therefore support the need to assess the effects of these dumpsites on the water quality of these water resources which are widely in use by the communities leaving around them.

2. Materials and Methods

2.1. Profile of the Study Area

Accra, the capital city of Ghana, is located at the south-eastern part of Ghana, and stretches between longitudes 5˚33' to 5˚55' north and latitudes 0˚15' to 0˚25' west. Accra ranges about 20 m above sea level on the average. The landscape is low-lying with few short irregular hills and depressions in some parts of the city. The map of the study area is presented in **Figure 1**.

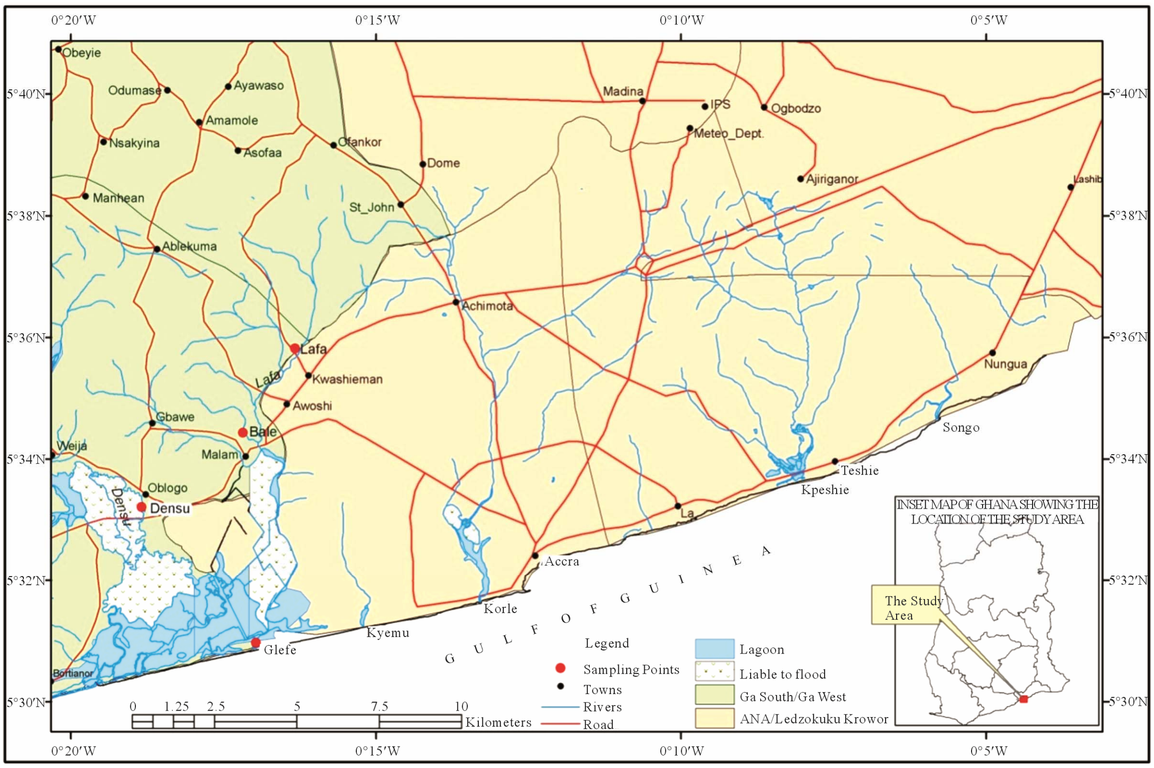

Figure 1. Map of the study area showing the sampling points.

Accra has four major drainage catchment systems. The Densu River and Sakumo Lagoon catchment drains settlements like, Dansoman, Kwashieman, McCarthy Hill and Awoshie areas. The Korle-Chemu catchment basin covers an area of 250 km^2. The Odaw River is the main stream in this system with Nima, Onyasia, Dakobi and Ado as tributaries. The Kpeshie catchment drainage basin covers an area of 110 km^2 and drains settlements like Cantonments, Osu, Labadi and Burma camp. The Songo-Mokwe catchment covers about 50 km^2, and drains Teshie.

2.2. Vegetation and Soils

The vegetation consists mainly of coastal-savanna grasslands, shrubs and some few mangroves in isolated areas. Market gardening is practiced in few places particularly along major waterways where irrigation is possible to support all-year round farming. Vegetables like pepper, okra, cabbage, lettuce, onion and cereals like maize are the main crops cultivated [8].

The geological formations consist mainly of the Precambrian Dahomeyan schist, granodiorites, granites gneiss and amphibolites and the Precambrian Togo series. The underground water table ranges between 4.80 metres to 70 metres. The main soil types include; Drift materials from wind-blown erosion, alluvial and marine mottled clays, residual clays and gravels from weathered quartzite, gneiss and schist rocks and lateritic sandy clay soils.

2.3. Collection of Water Samples

Sampling of water from the study area was done over a period of six months (June-November, 2009). Wet season samples were obtained in June, July and August while dry season samples were collected in September, October and November. The locations of the sampling sites were established using a Garmin 45 Ground Positioning System (GPS). The geographical locations, site elevations and types of samples collected from each sampling site are presented on **Table 1**.

At each sampling point, two sets of water samples were collected into separate pre-cleaned 1 L polyethylene bottles. 2.0 mL of concentrated HNO_3 was added to one of the bottles. The acidified sample was used for elemental analysis [9]. The non-acidified sample was analyzed for biological characteristics. Collected samples were stored in a cooler containing ice cubes, and later transported to the laboratory at the Department of Chemistry, University of Ghana, Legon, for analysis. At the laboratory, samples were stored in refrigerators at 4˚C until analysis.

Table 1. Surface water bodies sampled and their locations.

Surface Water Bodies	Sample Site Locations
Lafa	05º35.821′N; 00º16.301′W
Bale	05º34.372′N; 00º17.338′W
Densu (Oblogo)	05º33.493′N; 00º18.813′W
Gbegbe Lagoon (Glefe)	05º40.355′N; 00º09.594′W

Source: field survey, 2009.

2.4. Apparatus and AAS Measurement Conditions

An atomic absorption spectrometer (Analyst 400, Perkin Elmer) was used for the determination of the concentrations of Cd, Pb, Zn, Mn and Cu. Boosted Cd, Pb, Zn, Mn and Cu hollow cathode Superlamps (Photron, Australia) were employed as radiation sources. The operating conditions of the spectrometer for the determination of Cd, Pb, Zn, Mn and Cu are presented in **Table 2**.

2.5. Chemicals, Reagents, and Standards

HNO_3 (Merck, Germany); H_2O_2 (30%, Merck, Germany), were used for mineralization of the samples. Standard stock metal solutions were prepared from Cd stock standard solution (1000 mg/L in 2.0% HNO_3, TraceCERT®, Fluka, Switzerland), Pb stock standard solution (1000 mg/L in 2.0% HNO_3, TraceCERT®, Fluka, Switzerland), Mn stock standard solution (1000 mg/L in 2.0% HNO_3, TraceCERT®, Fluka, Switzerland), Cu stock standard solution (1000 mg/L in 2.0% HNO_3, TraceCERT®, Fluka, Switzerland), and Zn stock standard solution (999 mg/L in 1.4% HNO_3, Teknolab AB, Sweden) respectively.

For all dilutions, demineralized redistilled water was utilized. Calibration curves were developed by using calibrants prepared by appropriate dilution of the 1.0 $g \cdot L^{-1}$

Table 2. The AAS operating parameters for the five elements determined.

Element	Wavelength (nm)	slit width (nm)	lamp current (mA)	Flame: Fuel (Flow rate: 2 $L \cdot min^{-1}$)	Flame: Oxidant (Flow rate: 13.5 $L \cdot min^{-1}$)
Cd	228.8	0.5	4	Acetylene	Air
Pb	**217.0**	**1.0**	5	Acetylene	Air
Zn	213.9	**1.0**	5	Acetylene	Air
Mn	279.5	0.2	5	Acetylene	Air
Cu	324.8	0.5	4	Acetylene	Air

Detection limits for the five elements: Cd: 2.0 µg/l; Pb: 3.0 µg/l; Zn: 2.0 µg/l; Mn: 3.0 µg/l; Cu: 4.0 µg/l.

stock solutions to the required concentration with 2.0% HNO_3. The working standard metal solutions were prepared daily.

2.6. Cd, Pb, Zn, Mn, and Cu Measurements by AAS

Determination of metals in the acidified filtered (0.45 µm Millipore filter) water samples were carried out in accordance with standard methods [10,11]. The concentrations of Cd, Pb, Zn, Mn and Cu in the samples were respectively estimated by comparison with either the respective calibration curve or by the standard addition technique.

2.7. Physical and Chemical Measurements

Temperature, pH and dissolved oxygen were measured in-situ and recorded at the sampling sites. Nitrates, phosphates and physical parameters such as Biochemical Oxygen Demand (BOD_5), turbidity and suspended solids were also determined using standard methods [12].

2.8. Determination of Biological Characteristics

Total coliforms and Faecal coliforms were determined by membrane filtration method using M-Endo-Agar at 37˚C and on MFC Agar at 44˚C ± 0.5˚C for 48 hours, respectively.

All species of helminth eggs in water samples were quantified using the concentration method [13]. The identities of the specific helminth eggs were established using the World Health Organization (WHO) bench aid for the diagnosis of intestinal parasites [14].

3. Results and Discussions

3.1. Physical Parameters

The physical parameters of water quality can be broken down into many topics and one needs to take into consideration the nature of the physical parameters of the ecosystem surrounding a water source to be able to understand the physical appearance of water. Physical parameters which usually determine water quality are con sidered below.

3.1.1. Temperature of Water

Temperature affects sediment and microbial growth among other characteristics of water and it is also a known fact that the rate at which chemical reactions occur increase with increasing temperature and the rate of biochemical reactions usually double for every 10.0˚C rise in temperature. Physically, less oxygen can dissolve in warm water than in cold water. This is because increased temperature decreases the solubility of gases in water. Increased temperature increases respiration leading to increased oxygen consumption and increased decomposition of organic matter [15]. It is for these reasons that the temperatures of the water samples were determined for the river systems. The mean seasonal water temperature ranged from 27.4˚C at Glefe to 31.1˚C at Lafa in the wet season, **Table 3** and 27.4˚C at Bale to 28.7˚C at Glefe in the dry season, **Table 3**.

Since water temperature affects the concentration of biological, physical, and chemical constituents of water, the relatively high temperatures recorded would speed up the decomposition of organic matter in the water. Hence, population of bacteria and phytoplankton would double in warm weather in a very short time [16].

3.1.2. pH of Water

pH is important in water quality assessment as it influences many biological and chemical processes within a water body [16]. The pH values recorded were slightly alkaline with little variations among the study sites. The seasonal mean values ranged from 7.64 at Lafa to 8.06 at Glefe in the wet season, **Table 2** and 7.60 at Lafa to 7.74 at Bale in the dry season, **Table 4**.

The mean values fell within the WHO acceptable limits of 6.5 - 8.5 except at Glefe. The high pH value at Glefe is probably due to the direct disposal of refuse into the lagoon and also to sea water intrusion. However, most of the sampled sites had pH values slightly higher than natural background level of 7 for tropical surface water.

It is a known fact that variations in pH affect chemical and biological processes in water and low pH increases the availability of metals and other toxins for intake by aquatic life. On the other hand, the slightly high alkaline pH values recorded at the study sites would tend to decrease the availability of metals and other toxins for in-

Table 3. Physico-chemical characteristics of water samples for rainy season.

	Sampling sites/Mean levels of parameters			
Parameters	Bale	Densu	Lafa	Glefe
Temp. (˚C)	27.4	27.7	28.2	28.7
pH	7.7	7.8	7.6	7.6
Dissolved Oxygen (mg/l)	6.0	5.5	4.8	4.2
Biological Oxygen Demand (mg/l)	3.6	4.3	3.4	3.1
Suspended Solids (mg/l)	50.0	47.3	48.3	51.7
Turbidity (n.t.u)	33.6	39.9	32.0	43.4
Nitrate (mg/l)	2.1	2.2	2.2	2.0
Phosphate (mg/l)	0.4	0.2	0.2	0.3

Table 4. Physico-chemical characteristics of water samples for dry season.

Sampling sites/Mean levels of parameters				
Parameters	Bale	Densu	Lafa	Glefe
Temperature (˚C)	28.2	31.1	28.4	27.4
pH	7.8	7.7	7.6	8.1
Dissolved oxygen (mg/l)	5.8	5.8	4.8	4.2
Biological oxygen demand (mg/l)	4.0	4.5	3.7	3.5
Suspended solids (mg/l)	43.7	42.3	33.3	46.0
Turbidity (NTU)	27.9	39.0	40.2	48.4
Nitrate (mg/l)	2.1	1.9	2.0	1.8
Phosphate (mg/l)	0.2	0.2	0.2	0.3

Source: field survey (2009).

take by aquatic life as well as plants. The high pH may be due to the presence of other pollutants introduced into the water. As most of the study sites are located near landfills/dumpsites.

3.1.3. Turbidity

Turbidity (a term that refers to the optical property that causes light to be scattered and absorbed rather than transmitted in a straight line through water) in water is caused by suspended and colloidal matter such as clay, silt, finely divided organic matter, plankton and other microscopic organisms.

The mean seasonal values for this parameter ranged from 27.9 NTU at Bale to 49.0 NTU at Glefe in the wet season as can be seen in **Table 2** and 32.0 NTU at Lafa to 43.4 NTU at Glefe in the dry season, **Table 3**. All the values recorded were higher than the WHO value of 5 NTU. The high turbidity value could be due to the siting of the landfills/dumpsites close to the water bodies. It could also be due to indiscriminate disposal of waste into the water bodies. **Figure 2** shows heaps of refuse disposed off into the water body at Glefe.

Another possible cause of high turbidity values may be the siltation of the Densu, the Lafa, Bale Rivers and the Gbegbe lagoon. Siltation of these rivers and the lagoon is one of the problems arising from the cultivation along the banks of the rivers and the lagoon. Most of the farms are situated very close to the banks of these water bodies and cultivation of the banks is intense especially during the dry season, when there is water scarcity. This therefore results in erosion. According to the EPA, (2002) [6], turbidity values between 0.0 - 5.0 NTU show no visible turbidity, no adverse aesthetic effects and no significant risk of infectious disease transmission. Values > 10 NTU have severe aesthetic effects and the water carries an associated risk of diseases due to infectious agents and chemicals absorbed onto particulate matter [6].

3.1.4. Suspended Solids

Suspended solids consist of materials originating from the surface of the catchment area, eroded from river banks or lake shores and suspended from the bed of the water body [16]. Suspended solids include tiny particles of silts and clays, living organisms (zooplankton, phyto plankton and bacterioplankton) and dead particulate organic matter [17]. The seasonal mean values for suspended solids ranged from 33.3 mg/l at Lafa to 4.60 mg/l at Glefe in the wet season, **Table 3** and 47.3 mg/l at Densu to 51.7 mg/l at Glefe in the dry season, **Table 3**.

The suspended solids values recorded were generally high. The extremely high values recorded at all the sampling locations could be due to the large quantity of decomposing matter as all the sites have landfills/dumpsites located near them. At Glefe, as evidenced in **Figure 2**, aquatic microphytes are threatening to take over the lagoon.

According to Lester and Birkett, (1999) [18], suspended solid values of less than 25 mg/l have no harmful effect on fisheries as indicated in **Table 5**.

One direct effect of suspended solids is the influence on the turbidity of the receiving water body. This in turn reduces the amount of light that can penetrate the water and therefore will tend to reduce photosynthesis. Moreover, this could affect the recreational use of the water body. Suspended solids may also exhibit an effect if they settle out of suspension. Deposition of solids can change the characteristics of the riverbed, which will in turn affect plant and animal growth and fish breeding. Suspended solids generally cause damage to fish gills affecting their oxygen consumption and ultimately causing death at high concentrations. There was a defined trend in seasonal variations as dry season values were higher than the wet season values.

3.2. Chemical Parameters

Chemical characteristics of water can affect aesthetic qualities such as how water looks, smells, and tastes. This can also affect its toxicity and whether or not the water is safe to use. Since the chemical quality of water is important to the health of humans as well as the plants and animals that live in and around streams, it is neces-

Table 5. The effects of suspended solids on fisheries.

Suspended Solids (mg/l) Effects
<25 No harmful effect
25-80 Some possible reduction in yield
80-400 Good fisheries unlikely
>40 Very poor or non-existent

Source: lester and Birkett (1999).

sary to assess the chemical attributes of water. It is in light of these facts that the following chemical parameters have been determined for the water systems.

3.2.1. Dissolved Oxygen

The amount of oxygen dissolved in water depends on the rate of aeration from the atmosphere, temperature, air pressure and salinity. While the actual amount of oxygen that can be dissolved in water depends on the relative rates of respiration by all organisms and of photosynthesis by plants, oxygen levels are actually low where organic matter accumulates because aerobic decomposers require and consume oxygen. The mean seasonal values dissolved oxygen values of the river systems ranged from 4.7 mg/l at Glefe to 5.8 mg/l at Bale and Densu in the wet season, **Table 3** but ranged from 4.2 mg/l at Glefe to 6.0 mg/l at Bale and Lafa during the dry season as is observed in **Table 4**. The DO values recorded at the locations compared with the natural background level of 7.0 mg/l were generally low. This low values give an indication of pollution at all the sampling sites especially at Glefe and Lafa. The major possible causes of the pollution would include contamination by leachates from the landfill sites and indiscriminate defaecation and dumping of refuse along the banks and into the water bodies. The influence of other human activities such as farming at the river banks, fishing, washing and bathing in the river cannot be ruled out.

According to Cunningham and Saigo, (1997) [19], the addition of certain organic materials to water stimulates oxygen consumption by decomposers. The dissolved oxygen falls as decomposers metabolize waste materials. Water with less than 2.0 mg/l will only support detritus feeders, decomposers and worms. The optimal DO concentration for growth of fisheries is 5.00 - 8.00 mg/l. The sites that fell within this range are Bale and Densu where some kind of fishing is done. All the other sites except Glefe which fell in the range lethal for tilapia had concentration for which growth of tilapia will be impaired [6].

3.2.2. Biochemical Oxygen Demand (BOD)

Biochemical Oxygen Demand (BOD) is used as an index for determining the amount of decomposing organic materials as well as the rate of biological activities in water. This is because oxygen is required for respiration by micro-organisms involved in the decomposition of organic materials. Thus high concentration of BOD indicates the presence of organic effluent and hence oxygen-requiring micro-organisms. Mean seasonal BOD for the water systems ranged from a minimum of 3.1 mg/l at Glefe to a maximum of 4.3 mg/l at Densu in the wet season as in **Table 2** and 3.5 mg/l at Glefe to 4.5 mg/l at Densu during the dry season, **Table 4**.

Indiscriminate defaecation and refuse disposal was observed at all the sampling sites. The slightly high BOD values may be attributed to the discharge of organic waste into water bodies resulting in the uptake of DO in the oxidative breakdown of these wastes [20]. The nearness of the sampling locations to landfill/dumpsites is a factor promoting the loading of the water bodies with organic matter hence, the high BOD values.

The implication of high BOD in surface water could also mean that the oxygen present in the water will be used for decomposition of the pollutants, and thus, is not available for aquatic life anymore. The natural background level for freshwater ranges from 1.0 to 3.0 mg/l. The BOD of a river must generally not exceed 4.0 mg/l. This would reduce DO from saturating to 5.0 - 6.0 mg/l which is still capable of supporting aquatic life.

3.3. Nutrients

Nutrients mainly refer to inorganic matter from runoffs, landfills, livestock operation and crop lands, etc. The two primary nutrients of concern are usually phosphorus and nitrogen.

3.3.1. Nitrate

Nitrogen which usually exists in water bodies as nitrate is a key ingredient in fertilizers. It generally becomes a pollutant in saltwater or brackish estuarine systems where nitrogen is a limiting nutrient. Excess amounts of bioavailable nitrogen in marine systems lead to eutrophication and algae blooms.

It is with regards to the key role nitrates play in water quality determination that its assessment has been undertaken in this study. As can be seen from **Table 3**, the mean seasonal values for the compound ranged from 2.0 mg/l at Glefe to 2.2 mg/l at Densu and Bale in the wet season and 1.8 mg/l at Glefe to 2.1 mg/l at Bale in the dry season, **Table 4**. All the sites registered nitrate values higher than the natural background level of 0.23 mg/l. The nitrate concentrations were however lower than the WHO limit of 10.0 mg/l. The presence of nitrate may be the result of waste being disposed off at the landfills/dumpsites. Thus, contamination of the water bodies with chemicals from the landfills/dumpsites is very likely to occur. This is because wastes from agro-based Industries which may contain nitrates are not segregated before disposal and are likely to find their way into the river systems in runoffs or leachate emanating from the landfills. It could also be attributed to run-offs from farms along the banks of the rivers which may contain organic fertilizers. There was a slight seasonal variation as the wet season values were higher than the dry season values.

Nitrates are the most common form of nitrogen found in natural waters with enough dissolved oxygen. The natural background levels of nitrate may come from rocks, land drainage and plant and animal matter. Extremely high concentration of nitrate is toxic. However, the values recorded for all the sampling sites do not exceed the WHO limit value of 10.0 mg/l [21].

Invariably, nitrate is seldom abundant in natural surface water because it is incorporated into cells and chemically reduced by microbes and converted into atmospheric nitrogen [16]. This phenomenon may account for the low concentration of nitrate in surface waters.

3.3.2. Phosphate

Phosphorus is a nutrient that occurs in many forms that are bioavailable and phosphate is one such form of its existence. It is a main ingredient in many fertilizers used for agriculture as well as on residential and commercial properties, and may become a limiting nutrient in freshwater systems. Phosphorus is most often transported to water bodies via soil erosion because many forms of phosphorus tend to be adsorbed to soil particles. Excess amounts of the element in aquatic systems (particularly freshwater lakes, reservoirs, and ponds) leads to proliferation of microscopic algae called phytoplankton.

Mean seasonal values of the element in this study ranged from 0.2 mg/l at Lafa and Densu to 0.4 mg/l at Bale in the wet season as can be seen in **Table 3**. The dry season mean values range from 0.2 mg/l at Bale, Lafa and Densu to 0.3 mg/l at Glefe in the dry season, **Table 4**. The phosphate concentrations were relatively high compared with the natural background level of 0.02 mg/l. With the exception of Bale, all the remaining sites Registered values not above the WHO limit of 0.3 mg/l. The high concentration may be due to the effect of seepage from the landfill/dumpsites into the water bodies. It can also be attributed to domestic waste water and agricultural run-offs. A high phosphate concentration is an indication of pollution. There was only minimal variation in the seasonal trend in this study.

Phosphorus is also an essential nutrient and can exist in water in both dissolved and particulate forms. It is vital to the production of living organisms in the aquatic environment. High phosphate concentration is responseble for the eutrophication of a water body as phosphorus is a limiting nutrient for algae growth. All polyphosphates are eventually hydrolysed to produce the ortho form and the rate of hydrolysis is increased by temperature, decreased pH and bacterial enzyme action [22].

3.4. Heavy Metals

Compounds including heavy metals like lead, mercury, zinc, and cadmium, and organics like polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs), fire retardants, and other substances are resistant to breakdown. These contaminants can come from a variety of sources including mining operations, vehicle emissions, fossil fuel combustion, urban runoff, Industrial operations and landfills.

These compounds can threaten the health of both humans and aquatic species while being resistant to environmental breakdown, thus allowing them to persist in the environment. These toxic chemicals could come from croplands, nurseries, orchards, building sites, gardens, lawns and landfills.

3.4.1. Lead (Pb)

Lead in the environment is mainly particulate bound with relatively low mobility and bioavailability. Lead does, in general, not bioaccumulate and there is no increase in concentration of the metal in food chains [23]. Lead is also not essential for plant and animal life. The mean values of the metal ranged from 32.0 μg/l at Densu to 44.0 μg/l at Bale in the wet season, **Table 6**. The dry season values range from 35.0 μg/l at Densu to 72.0 μg/l at Bale as seen in **Table 7**.

The presence of lead in the water may be due to the discharge of industrial effluents from petroleum production [15]. Lead may also come from lead-acid batteries,

Table 6. Levels of Zn, Cu, Pb Cd and Mn in surface water bodies for rainy season.

Sampling sites/Mean levels of metals (μg/l)				
Parameters	Bale	Densu	Lafa	Glefe
Zinc	bdl	bdl	bdl	bdl
Copper	bdl	bdl	bdl	bdl
Lead	44.0	32.0	35.0	40.0
Cadmium	4.0	5.0	10.3	9.0
Manganese	186.0	130.0	240.0	139.0

Source: field survey (2009); (bdl = below detection limit).

Table 7. Levels of Zn, Cu, Pb Cd and Mn in surface water bodies for dry season.

Sampling sites/Mean levels of metals (μg/l)				
Parameters	Bale	Densu	Lafa	Glefe
Zinc	bdl	bdl	bdl	bdl
Copper	bdl	5.0	bdl	6.0
Lead	72.0	35.0	41.0	43.0
Cadmium	8.0	7.0	17.0	11.2
Manganese	26.0	310.0	452.0	44.0

Source: field survey (2009); (bdl = below detection limit).

plastics and rubber remnants, lead foils such as bottle closures, used motor oils and discarded electronic gadgets including televisions, electronic calculators and stereos [22] where leachates from the waste dumpsites may find their way into the rivers. All lead values fell between 32.0 µg/l to 72.0 µg/l. There are no adverse effects of exposure to water at these concentrations. The recommended range for livestock is 0.0 - 100.0 µg/l. All the sites had concentrations below 100.0 µg/l and therefore there are no health risk concerns. However, all the sites were above the general upper limit of 30.0 µg/l for continuous exposure for fish.

3.4.2. Copper (Cu)

The mean seasonal values of copper in water systems during the study period were below detection limit of 4.0 µg/l at all the study sites during the wet season, **Table 6**. The dry season values ranged from below detection limit to 6.0 µg/l as captured in **Table 7**.

Water quality range for copper for which there is no health or aesthetic effect is 0.0 µg/l to 10.0 µg/l and all the sites fell within this range. All the sites where the water is used for irrigation also fell below the level for which copper is toxic to plants, 100.0 µg/l to 1000.0 µg/l. For fisheries, the level for which there are no adverse effects on early life stages of some species ranges from 2.0 µg/l to 60.0 µg/l, and all the sites fell below this range hence, copper levels in the river systems pose no threat to the environment and health.

3.4.3. Manganese (Mn)

Manganese occurs in surface waters that are low in oxygen and often does so with iron. When oxidized in aerobic waters, the oxide builds up in distribution causing severe discolouration at concentrations above 50.0 µg/l [21]. The mean seasonal concentrations ranged from 130.0 µg/l at Densu to 240. 0 µg/l at Lafa in the wet season, **Table 6**. The dry season values ranged from 26.0 µg/l at Bale to 452.0 µg/l at Lafa, **Table 7**.

The presence of manganese may be due to discharge from industrial facilities or as leachate from landfills [10]. The very high values of manganese may be as a result of pollution from manganese dioxide cells for which the nation has no controlled methods of disposal. The metal may also come from other sources such as domestic wastewater and sewage sludge disposal. There was no clearly defined trend in seasonal variations. All the sites registered mean values above the WHO limit of 10.0 µg/l.

3.4.4. Cadmium (Cd)

Cadmium is readily accumulated by many organisms, particularly by micro-organisms and mollusc where bio-concentration factors are in the order of thousands. Soil invertebrates also concentrate Cd markedly [24]. Chronic exposure to the metal produces a wide variety of acute and chronic effects in mammals similar to those seen in humans. Kidney damage and lung emphysema are the primary effects of high Cadmium in the body.

Mean values of the metal in this study ranged from 4.0 µg/l at Bale to a maximum of 10.3 µg/l at Lafa in the wet season, **Table 6**. The dry season values recorded ranged from 7.0 µg/l at Densu to 17.0 µg/l at Lafa, **Table 7**. There was a defined trend as the dry season values obtained were higher than the wet season values. Even though the values obtained are low, cadmium is known to be one of the most toxic elements with reported carcinogenic effects to humans [25]. High concentration of cadmium has been found to lead to chronic kidney dys function. Cadmium may bioaccumulate at all levels of aquatic and terrestrial food chains. Cadmium contaminations in surface water bodies could be attributed to the discharge of contaminants including nickel-cadmium batteries. Some other activities which may introduce cadmium into these environments include electroplating and plastic manufacture.

3.4.5. Zinc (Zn)

Zinc levels were below the detection limits in all the waters sampled at the various sites. It is not clear why the very low level of the metal in the rivers despite the recorded values of zinc in the leachates from the dumpsites located close to the rivers [26]. This will need further investigation to ascribe reasons to the very low values of zinc.

3.5. Bacteriological Parameters

Pathogens are bacteria and viruses that can be found in water and cause diseases in humans. Typically, pathogens cause disease when they are present in public drinking water supplies. Pathogens found in contaminated runoff may also contain parasitic worms (helminths). Coliform bacteria and faecal matter may also be detected in runoffs. These bacteria are a commonly used indicator of water pollution, but not an actual cause of disease.

3.5.1. Total Coliform (TC)

Total coliform gives a clear indication of the general sanitary condition of water since this group includes bacteria of faecal origin. However, many of the bacteria in this group may originate from growth in the aquatic environment. This is used to evaluate the general sanitary quality of drinking and related water use [6]. The mean total coliform population in this study varied between 6.0 × 104 cfu/100 ml at Densu and 94.0 × 104 cfu/100 ml at Lafa in the wet season. The dry season recorded a value of 2.3 ×104 cfu/100 ml at Densu to 118.0 × 104 cfu/100 ml at Glefe. There was a defined trend in the seasonal variations as the dry season values were generally higher than the wet season values as indicated in **Tables 8** and **9**.

The high concentration of TC could also be due to indiscriminate defaecation, sewage, land and urban run-off and domestic waste waters [16]. The presence of coliform group of organisms is an indication of faecal contamination. The high TC counts observed at all the sampled sites make the river systems unsuitable for both primary contact, such as swimming and secondary contact such as boating and fishing according to the World Health Organization (WHO) limit [27].

Comparison of TC counts in the various sampled sites with the natural background and WHO limit of 0.0 cfu/100 ml indicated gross contamination with bacteria at all the sites making the water unsafe for drinking by humans and livestock. According to UNICEF, (1999) [28], if water is found to contain faecal indicator bacteria, it is considered unsafe for human consumption.

3.5.2. Faecal Coliform (FC)

Bacteriological examinations of water samples are done to determine the sanitary quality and the degree of contamination with waste [12].

Faecal coliforms are bacteria that live in the digestive tract of warm-blooded animals. They are excreted in the solid wastes of humans and other mammals. Where faecal coliforms are present, disease-causing bacteria are usually also present. Untreated faecal materials that contain faecal coliforms add excess organic material to the water. The decay of these materials depletes the water of oxygen which may result in killing of fishes and other aquatic life [29].

The mean values in our study ranged from 0.4 × 104 cfu/100 ml at Densu to 8.4 × 104 cfu/100 ml at Lafa in the wet season and the dry season values ranged from 0.6 × 104 cfu/100 ml at Densu to 11.1 × 104 cfu/100 ml at Bale as represented in **Tables 7** and **8** respectively.

There was a clearly defined trend in the seasonal variations indicating higher values for the dry season than the wet season. The high counts of faecal coliforms may be due to run-offs from the municipal landfills and urban solid waste disposal sites which contain domestic animal and human faecal materials [16]. It may also be attributed to indiscriminate refuse disposal along the banks of the water bodies.

The faecal coliform density was calculated using the formula:

$$\text{Colonies/100 ml} = \frac{\text{Colonies counted} \times 100 \text{ ml}}{\text{sample volume (ml)}}$$

Table 8. Bacteriological parameters of water for the rainy season.

Sampling sites/Mean levels of metals (μg/l)				
Parameters	Bale	Densu	Lafa	Glefe
Feacal coliforms (CFU/100 ml)	5.9	0.4	8.4	1.9
Total coliforms (CFU/100 ml)	73.3	6.0	94.0	10.0
Helminth eggs/500 ml of water	2.3	1.5	1.3	2.7

Source: Field survey (2009).

Table 9. Bacteriological parameters of water for the dry season.

Sampling sites/Mean levels of biological parameters				
Parameters	Bale	Densu	Lafa	Glefe
Feacal coliforms (CFU/100 ml)	11.1	0.6	9.6	9.8
Total coliforms (CFU/100 ml)	94.7	2.3	38.0	118.0
Helminth eggs/500 ml of water	0.6	0.5	3.3	0.0

Source: field survey (2009).

3.5.3. Helminths

Helminth eggs including *Ascarislumbricoides and Strongyloidesstercoralis* were detected in the water samples. The helminth egg population ranged from 1 to 4 egg(s) 500 ml^{-1} in the water samples. The mean seasonal values are presented in **Tables 8** and **9**. The helminth egg in the samples might be due to the disposal of waste containing human and animal faecal materials at the disposal sites.

4. Conclusions

The study revealed that the major pollutants into the Densu, Lafa, Bale Rivers and the Gbegbe lagoon (Glefe) have been identified to be organic waste, total and faecal coliforms. The sources of these pollutants into these water bodies are through runoffs from the municipal landfills/dump sites and could also be attributed to indiscriminate defaecation and refuse disposal which had contributed to elevated levels of the pollutants. Also, dumping and farming along the banks of these water bodies had led to eroded materials accumulating in them. This has resulted in the occurrence of large quantities of suspended solids and ultimately high turbidities. The discharge of organic waste including human excreta, domestic and animal waste either directly or indirectly through runoffs, into the water systems has resulted in high BOD levels and subsequently, low levels of dissolved oxygen in the waters. The low level of dissolved oxygen recorded for the entire study period is an indication that the waters in the study area could not support life sufficiently.

The presence of the coliform group of organisms is an indication of faecal pollution. This is quite alarming considering the assertion by Pierce *et al.*, (1998) [15], that large numbers of coliforms in water is an indication of recent pollution by wastes from warm-blooded animals and therefore the water may contain pathogenic organisms. Even though the people in the study area do not depend solely on these water bodies as their sources of water supply, the spate of water shortages could turn the tide. The presence of these coliforms could be responseble for the transmission of infectious diseases which include typhoid fever, dysentery, salmonellosis, cholera and gastroenteritis [6] which have been reported in the Accra metropolis. Heavy metals such as Cd, Pb, Mn, Cu and Zn analyzed in the water samples recorded varying levels of the metals. Heavy metals of public concern like Pb and Cd fell between 4.0 µg/l to 100 µg/l and were below the WHO recommended levels. There is therefore no threat to life in relation to the levels of these metals detected in the water bodies.

Helminth eggs and especially those of the genus Ascaris and Strongyloides families are the most commonly found in the water samples from the study area. However, there is no evidence of significant pollution with helminth ova that might pose a threat to humans especially those who have direct contact with the water.

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Journal of Water Resource and Protection, 2012, 4, 616-621
doi:10.4236/jwarp.2012.48071 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Study about Top-Down and Bottom-Up Controls in Regulating the Phytoplankton Biomass in a Eutrophic Reservoir in Northeastern Brazil

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Received May 8, 2012; revised June 17, 2012; accepted June 26, 2012

ABSTRACT

This study aims to analyze the effects of nutrients and predation by zooplankton on phytoplankton biomass (chlorophyll *a*) in a eutrophic reservoir in Brazil (Apipucos Reservoir, State of Pernambuco), through experiments in microcosms. For this, samples of water were placed in 1 L Erlenmeyer flasks and kept for seven days. Treatments included the addition of nutrients (nitrogen combined with phosphorus and isolated additions of nitrogen and phosphorus), with presence and absence of zooplankton and a control which contained the reservoir water without any manipulation. The addition of nutrients did not stimulate phytoplankton growth. However, zooplankton significantly decreased phytoplankton biomass in the treatments it was added to ($p < 0.05$). The results of this study showed that for the reservoir studied, predation by zooplankton is the most significant factor in the regulation of phytoplankton, contradicting several studies which show that phytoplankton biomass is more strongly controlled by nutrients (bottom-up control) than by predation (top-down control).

Keywords: Chlorophyll *a*; Nutrients; Zooplankton; Tropical Regions

1. Introduction

Top-down (predation) and bottom-up (nutrients) controls have received considerable attention in recent years, mainly due to disputes about which one is more effective in regulating the phytoplankton community. The availability of nutrients has been considered the main regulatory force of phytoplankton structure and biomass [1,2], since the top-down effects, according to [3], are stronger at the top of the food web and weaken gradually toward the base. Thus, it is believed that the phytoplankton biomass is more strongly controlled by nutrients (bottom-up) than predation (top-down). However, studies have also shown that phytoplankton can be strongly regulated by zooplankton [4-8].

Many studies have demonstrated that the addition of nutrients, especially nitrogen (N) and phosphorus (P), can significantly increase the growth of phytoplankton, and influence the taxonomic composition and size class distribution of these organisms [2,9-12]. Zooplankton can have both direct and indirect effects on the phytoplankton community, through predation and by the recycling of nutrients, respectively [5]. Predation generally causes a decrease in phytoplankton biomass thus influencing primary productivity. However, it can also cause positive effects on phytoplankton, since it can stimulate the growth of inedible algae that have a greater competitive advantage for nutrients when the biomass of edible algae decreases [13].

Studies dealing with top-down and bottom-up controls on phytoplankton in reservoirs and lakes have been conducted in temperate regions, but are rare in tropical environments. Due to the need for broadening and deepening understanding of the structure and dynamics of phytoplankton in these regions, this study aims to analyze the effects of nutrients and predation exerted by zooplankton on the phytoplankton biomass (chlorophyll *a*) in a eutrophic reservoir in northeastern Brazil (Apipucos Reservoir, State of Pernambuco), through experiments in microcosms.

2. Materials and Methods

2.1. Study Areas and Sampling

The Apipucos Reservoir (8˚01'14"S e 34˚56'00"W) is a

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eutrophic ecosystem belonging to the Capibaribe river basin, the main river in the state of Pernambuco, Brazil (**Figure 1**). It has a total area of 2.9 km^2, a volume of 556,375 m^3 and an average depth of 2.5 m [14]. It has been built for flood containment and recreation, and consists of two interconnected subsystems. The occurrence of free-floating macrophytes is expressive, represented mainly by *Eichhornia crassipes* (Mart.) for its historical, cultural and environmental values; this water body is considered an "Environmental Protection Area". However, the organic (untreated sewage release) pollution input is high, resulting in elevated eutrophication levels in this system [15]. The annual cycle of the region is characterized by two seasons regulated by precipitation: a rainy season (March to August) and a dry season (September to February).

Water samples for the laboratory experiment were collected on the day that each experiment began: January 10, January 30 and February 27, 2012. Eight liters of water were collected from the surface layer at a single point, with the aid of a van Dorn bottle, of which seven liters were filtered through plankton net with mesh opening of 68 μm to remove most of the zooplanktonic organisms that could interfere with the experimental design. The filtered material was stored in plastic containers and taken to the laboratory where it was used in the experiment. Part of this water was destined for nutrient analysis.

The following abiotic variables were measured *in situ:* water temperature (˚C); and dissolved oxygen ($mg \cdot L^{-1}$), measured through a Schott oximeter, Handylab OX1; water transparency, measured with a Secchi disk; and light intensity (μmol photons m^{-2}, s^{-1}), measured with a photometer. The turbidity and pH values were obtained from measurements in the laboratory with a Hanna Instruments turbidimeter HI 93703 and a Digimed potentiometer, DMPH-2, respectively.

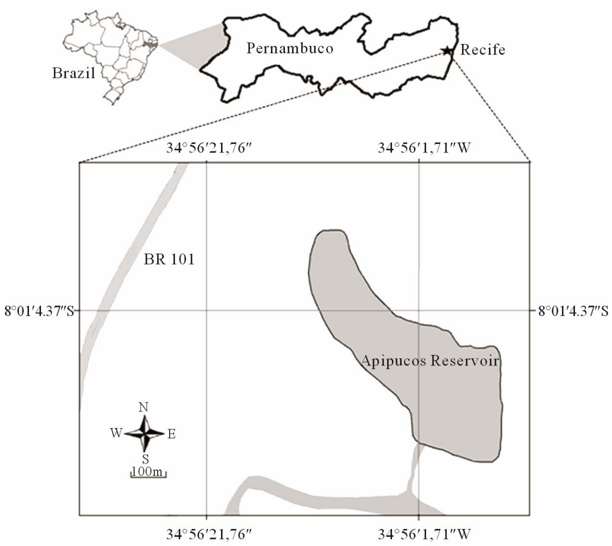

Figure 1. Geographical location of the Apipucos reservoir, Pernambuco, Brazil.

The concentrations of nitrate (μg $N\text{-}NO_3$ L^{-1}) and nitrite (μg $N\text{-}NO_2$ L^{-1}) were estimated according to the method described by [16]; and ammonia nitrogen ($N\text{-}NH_3 + N\text{-}NH_4^+$) [17], and total phosphorus (μg PT L^{-1}), according to the methodology in [18]. The total nitrogen (TN) was considered as the sum of the concentrations of nitrate, nitrite and ammonia nitrogen.

2.2. Experimental Design

Sub-samples of 700 mL were placed in 1 L Erlenmeyer flasks and kept in the laboratory for seven days under temperature conditions of 25˚C ± 1˚C, artificial light with a photoperiod of 12:12 (light and dark), and constant oxygenation.

Three experiments were performed, each one with treatments that included the addition of nutrients (nitrogen combined with phosphorus, nitrogen or phosphorus), with the presence and absence of zooplankton. Therefore, for the environment studied, seven treatments were tested (n = 3): two with the addition of nitrogen and phosphorus, with and without addition of zooplankton, treatments NP and NP + Z, respectively; two with the addition of nitrogen, with and without the addition of zooplankton, treatments N and N + Z, respectively; and two with the addition of phosphorus, with and without the addition of zooplankton, treatments P and P + Z, respectively, and the final one, water from the reservoir that had not been manipulated (Control).

For treatments with the addition of nutrients, the source of nitrogen was $NaNO_3$ and of phosphorus KH_2PO_4, and they were added once on the first day of the experiments (**Table 1**). Zooplanktons added to the treatments were collected in the reservoir using plankton net with mesh size of 68 μm. The initial density added to each treatment was twice the average recorded in the natural environment, based on the results of a recent study conducted in the studied reservoir [19].

2.3. Analysis of Phytoplankton

The phytoplankton biomass was estimated by the amount of chlorophyll *a* ($\mu g \cdot L^{-1}$), according to the methodology in [20] and, for this, 10 mL subsamples were removed from each treatment on the 1st, 3rd, 5th and 7th days of the experiment.

2.4. Statistical Analysis

Statistical differences between treatments were determined using a two-way analysis of variance (ANOVA), with a significance level of $p < 0.05$, using BioEstat. Version 5.0.

3. Results

3.1. Abiotic Variables and Chlorophyll *a*

The data of the physico-chemical variables, TN/TP ratio and chlorophyll *a* are shown in **Table 2**. The nutrient analysis in the Apipucos Reservoir showed high concentration of nitrogen in this environment and an average concentration of TN equal to 2229.07 ± 320.07 $\mu g \cdot L^{-1}$. Despite this, the TN/TP ratio was low (4.76 ± 0.98).

3.2. Bioassays

The addition of nutrients did not stimulate the growth of phytoplankton. At the end of the experiments, there were no significant differences between the concentration of chlorophyll *a* in the NP, N and P treatments and their respective control treatments ($p > 0.05$). Moreover, the zooplankton decreased significantly the phytoplankton biomass in those treatments to which it was added ($p < 0.05$). In the experiment with combined addition of nitrogen and phosphorus, the final chlorophyll *a* concentration was 886.53 $\mu g \cdot L^{-1}$ in the control, while in treatments NP + Z and Z it was 605.93 $\mu g \cdot L^{-1}$ and 540.87 $\mu g \cdot L^{-1}$, respectively. In the experiment with only the addition of nitrogen, the concentration was 1106.1 $\mu g \cdot L^{-1}$ in the control, 455.47 $\mu g \cdot L^{-1}$ in treatment N + Z and 345.67 $\mu g \cdot L^{-1}$ in treatment **Z**. In the experiment with phosphorus addition, the final concentration of chlorophyll was 947.5 $\mu g \cdot L^{-1}$ in the control, 427 $\mu g \cdot L^{-1}$ in treatment P + Z and 329.4 $\mu g \cdot L^{-1}$ in treatment Z (**Figure 2**).

Table 1. Concentration of nitrogen ($NaNO_3$) and phosphorus (KH_2PO_4) in the treatments with added nutrients.

Experiments		$NaNO_3$ ($\mu g \cdot L^{-1}$)	KH_2PO_4 ($\mu g \cdot L^{-1}$)
Nitrogen + Phosphorus	NP	4019.54	186
Nitrogen	N	15127.14	-
Phosphorus	P	-	186

Table 2. Physical-chemical variables, TN/TP ratio and chlorophyll *a* at the Apipucos Reservoir.

	Mean (± standard deviation)
Water temperature (˚C)	28.53 (±0.21)
Dissolved oxygen ($mg \cdot L^{-1}$)	3.67 (±1.69)
Depth (m)	1.20 (±0.15)
Transparency (m)	43.33 (±7.64)
Electrical conductivity ($\mu S \cdot cm^{-1}$)	992.33 (±288.12)
Light intensity (µmol photons m^{-2}, s^{-1})	139.25 (±81.97)
Turbidity (UNT)	34.14 (±6.27)
pH	7.10 (±0.16)
Nitrate ($\mu g \cdot L^{-1}$)	304.82 (±15.99)
Nitrite ($\mu g \cdot L^{-1}$)	106.26 (±2.85)
Ammonia nitrogen ($\mu g \cdot L^{-1}$)	1817.99 (±308.43)
Total nitrogen ($\mu g \cdot L^{-1}$)	2229.07 (±320.07)
Total phosphorus ($\mu g \cdot L^{-1}$)	472.95 (±34.10)
TN:TP ratio	4.76 (±0.98)
Chlorophyll *a* ($\mu g \cdot L^{-1}$)	432.69 ±136.04)

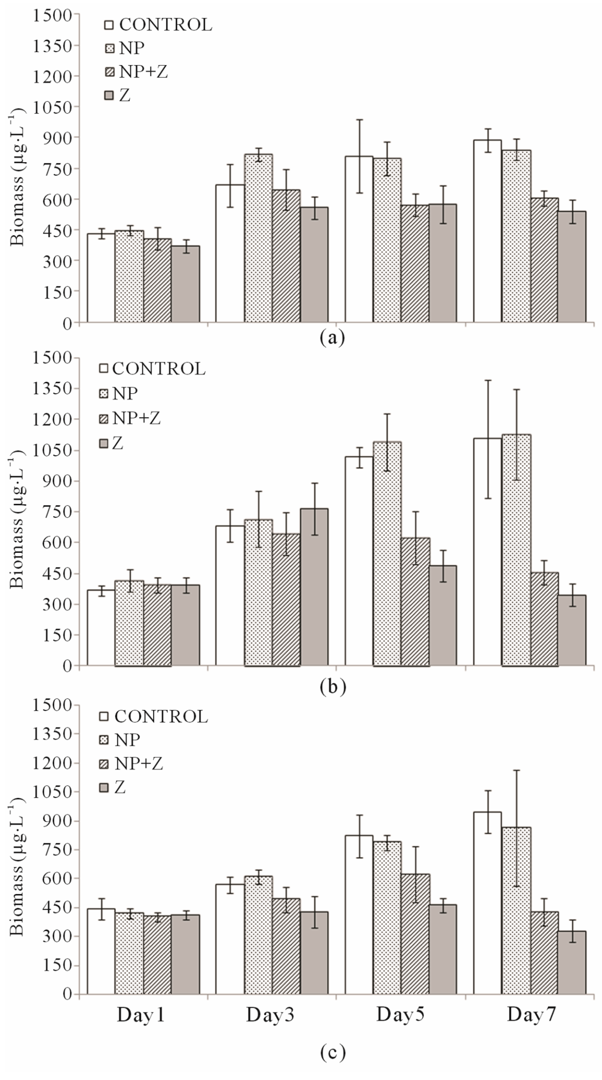

Figure 2. **Average concentration of phytoplankton biomass (±standard deviation) in the experiments with added (a) Nitrogen and phosphorus; (b) Nitrogen; and (c) Phosphorus. Treatment abbreviations are explained in Materials and Methods.**

4. Discussion

Studies show that phytoplankton biomass tends to increase with the addition of nutrients, especially nitrogen (N) and phosphorus (P), and decrease due to predation by zooplankton [10,21,22]. In this study, the addition of nutrients had no effect on the phytoplankton biomass, measured by the concentration of chlorophyll *a*.

Due to the low N:P ratio recorded in the Apipucos Reservoir, the addition of nitrogen, or nitrogen combined with phosphorus, was expected to stimulate the growth of phytoplankton. According to [23], lakes and reservoirs which have an N:P ratio lower than 20 are limited by nitrogen, those greater than 50 are limited by phosphorus, and those between 20 and 50 are co-limited by N and P. A similar pattern was also proposed by [24,25].

Traditionally, phosphorus has been identified as the main limiting factor for phytoplankton growth, at least in temperate regions, while nitrogen is frequently cited as limiting at low latitudes [26]. However, it has been difficult to establish the main limiting nutrient for phytoplankton, because this can vary between different environments and among algal species.

Dzialowski *et al*. (2005) [24], analyzing 19 reservoirs in Kansas, USA, observed that the addition of P stimulated growth of phytoplankton in only 8% of all experiments in bioassays. On the other hand, limitation by N (16%) and co-limitation by N and P (63%) was often observed. Moreover, just as in the present study, the authors observed that in 13% of the reservoirs, the addition of nutrients did not stimulate phytoplankton growth. A similar result was observed by [27], who analyzed five dams along the Manyame River (Harava, Seke, Chivero Lake, Manyame Lake and Bhiri), Mozambique. Nitrogen was the main limiting factor in the Harava, Seke and Manyame Lake dams, and phosphorus was a limiting factor in Bhiri, while no nutrient limited phytoplankton growth in Lake Chivero.

In both studies cited above, light was identified as the main limiting factor in the reservoirs where the addition of nutrients did not stimulate algal growth. The possibility of limitation by light is rarely addressed in bioassays for limiting nutrients in freshwater systems [28]; however there is frequent evidence of limitation by light for phytoplankton in lakes and reservoirs [28-30]. According to [31], when the phytoplankton is limited by light it may show little or no limitation by nutrients.

The availability of these two resources may vary spatially and temporally in most ecosystems [29-32]. According to [32], in periods of intense rainfall there is an excessive discharge of nutrients which can directly lessen the limitation of this feature. Moreover, there is an increase in the discharge and re-suspension of sediments, increasing the intensity of limitation by light, since less penetration of light into the water column can ultimately limit algal growth.

Regarding spatial variation, it is believed that the phytoplankton growing in shallow water sites near heavy water flow are exposed to increased sediment discharge and pulses of nutrients when compared to phytoplankton growing in deeper, thermally stratified areas closer to the dam. Hence, phytoplankton from shallow lotic areas may be more limited by light and less limited by nutrients than phytoplankton from deeper areas [32].

Due to the environmental conditions observed in the Apipucos Reservoir, the results of the experiments suggest a potential limitation by light, since this reservoir is made up of neighborhoods heavily populated by low-income populations, where sewage is discharged continually, either directly or indirectly [33]. Moreover, its waters are dark and turbid, indicating a high concentration of suspended sediment.

One should also consider the possibility of limitation by another nutrient or a seasonal variation in the limiting resource of the phytoplankton in the Apipucos Reservoir.

While the addition of nutrients had no effect on the phytoplankton biomass, predation exerted by zooplankton caused a significant reduction in the biomass. This result does not corroborate the hypothesis that top-down control exerted by zooplankton in tropical systems is weak when compared to that observed in temperate regions [34]. It is believed that the absence of large herbivorous zooplankton in the tropics, as well as the dominance of small-sized species, are the main reasons responsible for this difference [35,36].

Von Rückertand and Giani (2008) [34], while evaluating the effects of predation of three microcrustaceans *Daphnia laevis*, *Moina micrura* and *Thermocyclops decipiens* on the phytoplankton community in the Pampulha Reservoir (Minas Gerais, Brazil) found that zooplankton was inefficient in controlling phytoplankton. Low *et al*. (2010) [37], when analyzing phytoplankton and zooplankton populations in 12 tropical reservoirs, observed that the zooplankton exerted a top-down control on certain algal communities, such as green algae (Ankistrodesmus, Scenedesmus and Cosmarium) and phytoflagellates (*Peridinium*). Cyanobacteria, with the exception of *Planktothrix*, were not adversely affected by zooplankton.

For a better understanding of the interaction between phytoplankton and zooplankton communities in the Apipucos Reservoir, more information about the various factors that may affect the ability of zooplankton to prey on phytoplankton is necessary, namely, the taxonomic composition and biomass of the zooplankton, body size of the zooplankton and phytoplankton, zooplankton selectivity and efficiency, resilience of the phytoplankton, the relationship between the nutritional requirements of the zooplankton and the nutritional quality of the phytoplanktonic organism [38-44].

5. Conclusion

The results of this study showed that, for the reservoir studied, predation by zooplankton is the most significant factor in the regulation of phytoplankton, contradicting several studies which show that phytoplankton biomass is more strongly controlled by nutrients (bottom-up control) than by predation (top-down control).

6. Acknowledgements

This study was supported by grants from CNPq (Brazilian Council for Research and Development (302068/2011-2).

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Journal of Water Resource and Protection, 2012, 4, 622-633
doi:10.4236/jwarp.2012.48072 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Environmental Implications of the Discharge of Municipal Landfill Leachate into the Densu River and Surrounding Ramsar Wetland in the Accra Metropolis, Ghana

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Received May 24, 2012; revised June 27, 2012; accepted July 5, 2012

ABSTRACT

Investigations were conducted over a six-month period on leachate which continuously egresses from a "natural attenuation" landfill site into a fragile ecosystem in the Accra Metropolis, Ghana. Most physico-chemical, oxygen demand parameters and nutrient contents were within permissible limits but Total Dissolved Solids (1124 - 13200 mg/l), conductivity (7960 - 24890 µS/cm), Mn (0.12 - 0.94 mg/l), Ca^{2+} (160 - 356 mg/l) and, more especially chloride contents (1030 - 2967 mg/l) far exceeded respective World Health Organisation (WHO) limits for effluent discharge into the natural environment. Multivariate statistics using Principal Component Analysis (PCA) and Cluster Analysis (CA) suggest significant concentrations of Ca^{2+}, Cl^-, and to a lesser extent Zn, Cd, Mn and PO_4^{2-} relative to the river water samples. Because the landfill was abandoned recently (in 2009), degradation and other breakdown processes of waste material may only have just began, suggesting that the uncontrolled and continuous discharge of chloride and some heavy metal-laden leachate could, in the long-term, substantially impact negatively on the Ramsar Densu wetland and surrounding water bodies, soil and nearby marine ecosystem.

Keywords: Densu Wetland; Ghana; Landfill; Leachate

1. Introduction

Municipal solid waste landfills and the many harzardous materials or contaminant types they contain could reportedly have various adverse effects on environmental compartments including surface and groundwater resources, soils, fauna and flora as well as human health [1-7]. Such landfills often produce leachate, *i.e.* the liquid that usually drains from landfills due to infiltration by water and/or biogeochemical decomposition processes, which serves as an important point source of pollution in many environmental media around the world [8,9]. The constituents in leachate, some of which may be toxic, have often posed serious challenges in terms of cost of treatment, accumulation of metal or species, remediation and, in particular, possible eco-toxicological implications resulting from both short- and long-term exposure or bioaccumulation of leachate constituents.

In Ghana, municipal solid waste from households, commercial establishments and industries in the city with varied composition is commonly disposed of at open mainly un-engineered dump sites or, more frequently, abandoned quarry sites located in the city [10,11]. In the Accra metropolis, for example, such landfill sites receive the over 55% of all solid waste generated that the Metropolitan Assembly (AMA) collects [12]. The Oblogo landfill, one of many in the Accra Metropolis, is situated within an abandoned quarry hosted in well bedded rocks of the Togo Formation [13]. As a result of decomposition of waste, streams of untreated leachate continuously flow from the landfill into the surrounding environment [14]. In spite of the possible hazards presented by the apparently uncontrolled seepage and migration of leachate from many such un-engineered landfills throughout the country, very few studies have been undertaken, neither have effective mechanisms been put in place for leachate control or management. This paper presents data on leachate from the Oblogo landfill which continuously seeps and discharges into soils, river (Densu River), ecologically important Ramsar wetland and nearby marine environment in the Accra Metropolis, Ghana. The

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implications of the uncontrolled discharge of some constituents in the leachate are also briefly discussed.

2. Study Area

2.1. Location and Geographic Elements

The study area is located on approximately latitudes 5˚33'26"N and 5˚33'40"N and longitudes 0˚18'45"W and 0˚18'55"W in the Ga District in south-western Accra, Ghana (**Figure 1**). The landfill is situated in an area underlain by the Togo series of rocks which consist of bedded and interbedded sequences of quartzite, phyllite and schist [15]. The site covers an area of approximately 20,000 m^2 on the edge of a ridge about 200 m by road from Oblogo Township and approximately 1 km off the major Accra-Takoradi-Half Assini (Accra-Abidjan) highway.

The site lies in the coastal savannah zone and has mean annual rainfall of 800 mm [16]. The rainfall is seasonal with two peaks in June and September. According to Ghana Meteorological Agency, rainfall up to a maximum of about 200 mm can occur in one day and much of that could fall in about one or two hours. The highest mean monthly temperatures occur between March and April. Minimum and maximum daily temperatures range from 22.8˚C to 33.0˚C, respectively. The minimum yearly average is 24.2˚C with maximum yearly average of 31.0˚C. The highest monthly mean temperatures occur in April and the lowest in July. Mean relative humidity is high within a 24-hr period with relative humidity occurring in January and the highest in August.

The dominant vegetation is shrub and grassland. Thin grass and occasional patches of shrub characterise the landfill area. The vegetation grades gradually towards the Densu River into the surrounding wetland close to the coast. The wetland, a designated Ramsar site, is rich in various fish species and rare flora and fauna [17]. Residential buildings occur quite close to the landfill. Stone quarrying, fishing and subsistence farming are some economic activities undertaken by many people in the area. Others also undertake recycling and scavenging activities at or close to the landfill. Leachate from the landfill mainly flows into naturally created sumps where it is stored temporarily before flowing downslope through

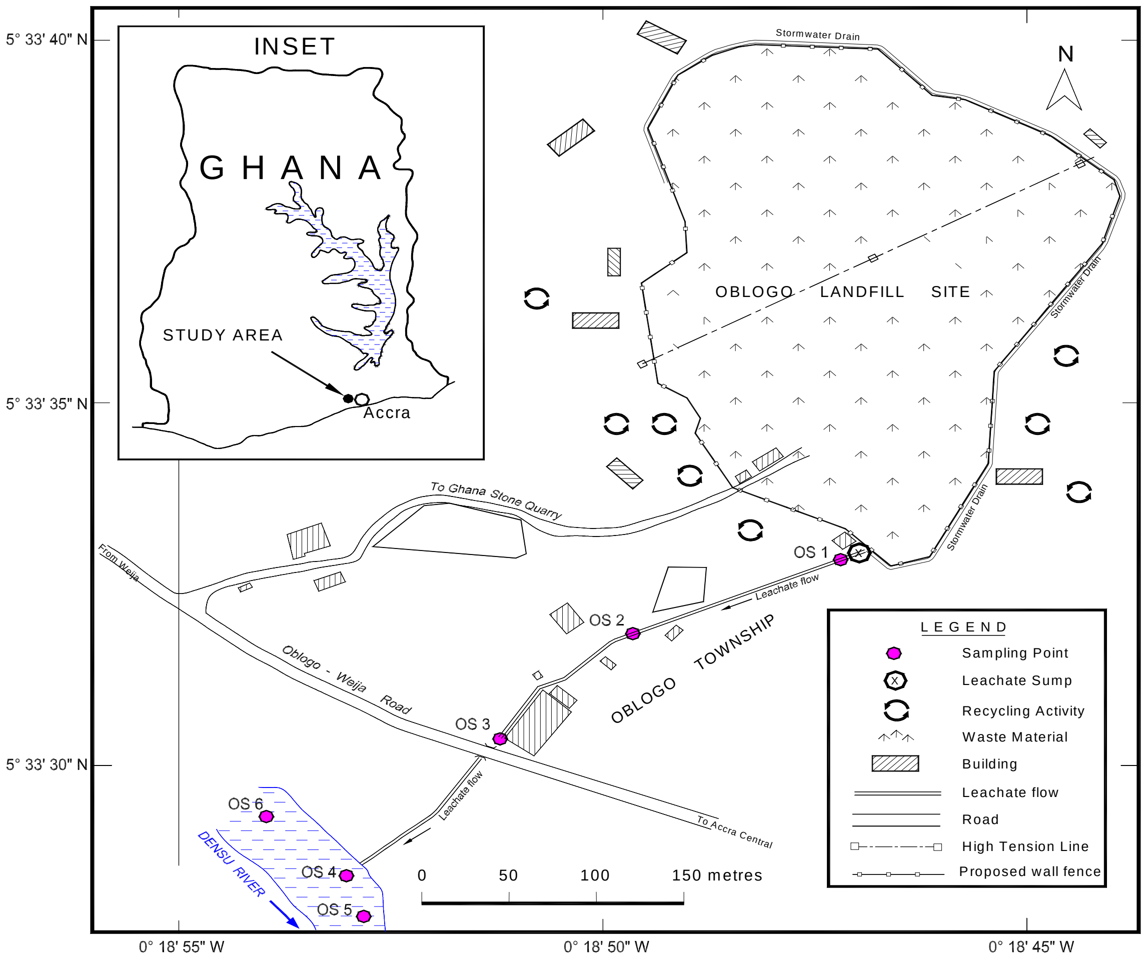

Figure 1. Study area with sampling locations.

Oblogo Township and parts of the wetland to join the Densu River about 250 m from the landfill. The river then flows less than a kilometre through the wetland into the Atlantic Ocean.

3. Methodology

3.1. Field Work

The study involved sampling and analysing leachate and river water along approximately 250 m from the landfill at an interval of about 100 m for six months. The location and description of sampling sites are given in **Table 1**. Sampling was done between January and June, 2004. A hand-held Global Positioning System (GPS) was used to locate sampling points. Samples were taken at varying but designated locations from the landfill site up to where leachate entered the Densu River through the wetland system (**Figure 1**). Samples were collected in the dry (January to March) and rainy (April to June) seasons once every month from six sampling points in accordance with protocols on sampling by APHA [18]. Most samples were collected in plastic bottles and labelled appropriately. Three samples were taken at each sample point, one in a 1.5-litre plastic bottle for physico-chemical analysis, another in a 100 ml plastic bottle acidified with nitric acid for mainly heavy metal contents and the third in a standard "ox top" bottle for oxygen demand parameters. Sample bottles were first rinsed with leachate or water before carefully dipping individual bottles in flowing leachate and water at the respective sampling points. These precautions were taken to reduce contamination. The collected samples were then kept in an ice chest in the field and later transferred into a refrigerator until analysis was done.

3.2. Analysis of Leachate and Water Samples

Analytical methods used for leachate and water samples varied depending on the parameters of interest. All field and laboratory determinations were done according to standard methods for the examination of waste and waste water [19]. For every sample, physico-chemical, nutrients and oxygen demand parameters were determined. Measurements of physical parameters were taken in situ by the use of a Water Quality Check U-10 instrument. Values of measured parameters were read from the digital display when the Checker U-10 was immersed in the respective samples.

Physico-chemical parameters were determined at the Water Research Institute (WRI) of the Council for Scientific and Industrial Research (CSIR, Ghana). Trace metals Fe, Mn, Zn and Cd were determined with Unicom Atomic Absorption Spectrometer (AAS). Samples were first treated with a mixture of concentrated nitric, sulphuric and perchloric acid in a digest and each sample solution aspirated into a flame and atomized. A light beam was then directed through the flame into a monochromator and onto a detector that measured the amount of light absorbed by the element in the flame. A blank sample (acidified) was also aspirated to set the automatic zero control. At least six standards were used for each element. Various samples were then aspirated individually and the respective concentrations obtained from the digital display. Concentration of sulphate in the samples was determined by the sulphate-turbidimetric method. To 100 ml of the sample, 5 ml of conditioning reagent (barium chloride) was added and stirred for about 60 seconds. The absorbance was read at 420 nm on a spectrometer. Concentration of sulphate was then calculated from standard calibration formula. Phosphate $\left(PO_4^{2-}\right)$ was also determined using the stannous chloride acid method (APHA, 2005).

Biochemical Oxygen Demand (BOD) was determined by diluting portions of the sample and incubating for 5 days at 20˚C. The BOD exerted over the 5 days deter-

Table 1. Location and description of leachate and stream or river water samples relative to landfill site, Accra, Ghana.

Sample No.*	Description of Sample Point	Sample Type**	GPS Location	~Distance (m) from landfill (Reference Pt.)
OS1	Naturally-created leachate sump	Landfill leachate	0˚18'44.8"W 5˚33'33.8"N	5
OS2	Artificial (dug) sump	Landfill leachate	0˚18'49.6"W 5˚33'32.6"N	100
OS3	Natural leachate sump	Landfill leachate	0˚18'52.8"W 5˚33'31.1"N	200
OS4	Leachate confluence with Densu River	River water	0˚18'52.3"W 5˚33'25.8"N	220
OS5	Slightly upstream of leachate confluence with Densu River	River water	0˚18'54.8"W 5˚33'25.1"N	230
OS6	Downstream of leachate confluence with Densu River	River water	0˚18'55.1"W 5˚33'28.1"N	250

*OS1 represents the first leachate sampling point at a sump topographically just below landfill; OS2 sample point along leachate flow path downslope or down gradient of OS1; OS3 is located along leachate flow path close to a major road linking Oblogo and Weija; OS4 is located in an area where the leachate empties into the Densu River; OS5 and OS6 are located downstream and upstream of OS4, respectively (**Figure 1**). **River water = sample taken from Densu River.

mined as follows:

Calculations

$$BOD_5 = BOD \times S_1 \times S_2$$

where

BOD_5 = BOD recorded on the fifth day from the Oxitop.

S_1 = Dilution factor.

S_2 = Factor dependent on total volume of diluted sample put in Oxitop bottle.

In determining the Chemical Oxygen Demand (COD), the sample was refluxed in concentrated sulphuric acid with a known excess of potassium dichromate ($K_2Cr_2O_7$) for two hours. After digestion, the remaining reduced $K_2Cr_2O_7$ was titrated with ferrous ammonium sulphate to determine the amount of $K_2Cr_2O_7$ consumed and the oxidizable matter calculated in terms of the oxygen equivalent.

Microsoft Excel (version 2007) was used to obtain correlation coefficients between measured physical-chemical and nutrient parameters for leachate and river water. In addition, the data were subjected to multivariate statistical analyses [20,21] involving Principal Component Analysis (PCA) and Cluster Analysis (CA) using SPSS (version 12.0).

4. Results

4.1. Physicochemical Data for Landfill Leachate

Data on parameters from leachate samples taken during the study are presented in **Table 2(a)**. pH values of leachate range from 6.6 close to the landfill (~5 m) in January to 7.9 (mean 7.4) in Aprilat a distance of 200 m from the landfill. Throughout the sampling period as well as outwards from the landfill, the pH of leachate thus remained fairly uniform. Temperature values also range from a minimum of 27.8˚C in April at distance 5 m to a maximum of 35.3˚C in January at the same sampling site, *i.e.* 5 m from the landfill. Even though minor differences occur up to about 100 m from the landfill, the values in general suggest not much change in temperature of leachate with respect to sampling period or distance from the landfill site. The lowest and highest conductivity values of 7960 and 24,890 µS/cm were obtained in leachate taken respectively in March (distance 5 m) and June (distance 100 m) from the landfill. Range of values for total dissolved solids (TDS), salinity and turbidity of leachate were as follows; TDS 1124 mg/l in April at about 100 m from the landfill to 13,200 mg/l also in April at about 100 m from the landfill; salinity 0.18% in June at 200 m from landfill to 2.02% in April at 5 m from landfill; turbidity 3.1 NTU in June at 200 m to 60.1 NTU in April at 100 m (**Table 2(a)**).

Fe concentrations in leachate ranged from 2.05 - 18.0 mg/l at 200 m and 5 m, respectively, from the landfill, the lowest value in April and the highest in January (**Table 2(a)**). Cadmium, Zinc and manganese also varied from 0 - 2.45 mg/l (distance 100 m and 5 m both in January), 0.02 - 0.28 mg/l (distance 100 m in February and 5 m in January) and 0.12 - 0.94 mg/l, respectively.

Both the minimum (0.12 mg/l) and maximum (0.94 mg/l) Mn values were obtained at more than one site (**Table 2(a)**). Calcium and chloride contents ranged from 160 - 356 mg/l (mean 276.7 mg/l) and 1030 - 2967 mg/l (mean 2291 mg/l), respectively, whilst total hardness also ranged from 104 to 1300 mg/l (mean 889.7 mg/l). The highest Ca^{2+} value was obtained in March in leachate sample taken about 5 m from the landfill and the lowest in January about 200 m from the landfill. Chloride in leachate (**Table 2(a)**), on the other hand, registered the highest and lowest values in June, the former nearer the landfill (distance 5 m) and the latter farther away (distance 200 m).

The nutrient contents of leachate, as given by concentrations of PO_4^{2-}-P, SO_4^{2-} and NO_3^--N (**Table 2(a)**), also showed variations with respect to distance from the landfill and sampling period. PO_4^{2-}-P contents ranged from 8.23 mg/l in April at about 200m from the landfill to 30 mg/l in January about 100 m from the landfill. The highest concentrations of SO_4^{2-} (68.3 mg/l) and NO_3^--N (41.52 mg/l) were both obtained in January at site 5 m from the landfill whilst the lowest (*i.e.* SO_4^{2-} 28.6 mg/l and NO_3^--N 1.03 mg/l) were also obtained in June with the SO_4^{2-} at 200 m and NO_3^--N at 100 m from the landfill. The oxygen demand parameters DO, BOD and COD also exhibited variations with respect to sampling site and period but were generally characterised by low values (**Table 2(a)**).

4.2. Physicochemical Data for River (Densu) Water

Table 2(b) gives data from water samples taken from the Densu River into which leachate egresses (see **Figure 1**). Except for Cd and Zn that were generally below detection, the data show perceptible variations with respect to site and period of sampling. pH ranged from 6.6 - 8.1 (mean 7.5), temperature 27.8˚C - 31.2˚C (mean 29.4), conductivity 610 - 1903 µS/cm, TDS 102 - 450 mg/l, salinity 0.01% - 0.13% and turbidity 2.0 - 45.1 NTU. Fe and Mn ranged from 0.12 - 1.23 mg/l and 0.12 - 0.92 mg/l, respectively. Calcium, chloride and total hardness also ranged from 23 - 70 mg/l (mean 36.1 mg/l), 59 - 105 mg/l (mean 81.8 mg/l) and 60 - 140 mg/l (mean 104.7 mg/l), respectively. Other variations were as follows; PO_4^{2-}-P 0.15 - 10 mg/l (mean 2.23 mg/l), SO_4^{2-} 16.1 - 33.8 mg/l (mean 25 mg/l), NO_3^--N (0.23 - 21.02 mg/l (mean 5.6 mg/l), DO 0.26 - 1.64 mg/l (mean 0.94 mg/l), BOD 0.03 - 1.04 mg/l (mean 0.20 mg/l) and COD 0.12 - 1.93 mg/l (mean 0.93 mg/l).

Table 2. (a) Physico-chemical data from landfill leachate from January to June, 2004, Accra, Ghana; (b) Physico-chemical data from River water from January to June, 2004, Accra, Ghana.

(a)

Month	Spl. Pt	pH	Temp (˚C)	Cond. $\times 10^3$ (μS/cm)	TDS $\times 10^3$ (mg/l)	Salinity (%)	Turb. (NTU)	Fe (mg/l)	Cd (mg/l)	Zn (mg/l)	Mn (mg/l)	Ca^{2+} (mg/l)	Cl^- (mg/l)	Total Hard (mg/l)	PO_4^{2-}-P (mg/l)	SO_4^{2-} (mg/l)	NO_3^--N (mg/l)	DO (mg/l)	BOD (mg/l)	COD (mg/l)
	OS1	6.61	35.3	21.61	10.80	1.31	41.0	18.00	2.45	0.28	0.12	241	2730	1000	22.80	68.30	41.52	0.63	0.81	1.28
Jan	OS2	7.59	28.3	24.82	7.58	0.88	36.2	14.00	0.00	0.18	0.15	168	1936	800	30.00	63.20	33.03	0.99	0.36	1.23
	OS3	7.68	28.3	23.13	9.01	0.95	31.2	9.78	0.05	0.12	0.14	160	1986	600	17.00	43.10	26.01	0.42	0.76	2.01
	OS1	7.57	32.3	24.28	12.46	1.52	55.0	10.20	1.23	0.07	0.19	348	2878	1200	18.60	63.70	15.23	0.51	0.23	1.32
Feb	OS2	7.61	29.4	15.12	8.97	1.12	48.1	10.10	0.01	0.02	0.35	326	2356	900	17.30	58.50	14.76	0.48	0.31	1.41
	OS3	7.54	28.7	16.83	9.66	0.98	21.1	5.87	0.08	0.09	0.12	189	1782	110	11.30	48.90	12.53	0.47	0.12	1.23
	OS1	7.63	32.7	7.96	10.79	1.51	56.1	8.78	0.98	0.05	0.23	356	2913	1100	18.90	65.10	5.02	0.13	0.21	1.04
Mar	OS2	7.65	29.5	8.23	8.62	1.01	43.0	5.89	0.21	0.06	0.32	342	2798	1000	14.90	56.30	4.89	0.52	0.42	0.43
	OS3	7.58	28.8	18.34	4.22	0.54	23.1	4.98	0.03	0.03	0.28	289	1189	900	11.80	44.00	2.43	0.39	0.81	1.03
	OS1	7.75	27.8	24.43	13.20	2.02	54.9	6.32	0.78	0.04	0.34	287	2889	1200	16.00	59.90	4.25	0.18	0.12	1.06
Apr	OS2	6.98	28.7	20.03	1.12	1.68	60.1	4.32	0.52	0.03	0.94	291	2098	1000	17.00	49.40	9.03	0.38	0.25	0.96
	OS3	7.87	29.6	18.34	9.08	0.64	56.1	2.05	0.43	0.02	0.12	321	1234	800	8.23	42.30	8.05	0.09	0.21	0.86
	OS1	7.65	29.8	24.28	11.22	1.98	48.9	5.32	0.46	0.03	0.72	342	2798	1300	16.50	52.20	3.23	1.02	1.03	1.89
May	OS2	7.36	27.8	21.74	11.88	1.23	50.9	4.82	0.62	0.02	0.45	329	2869	1000	20.00	49.00	9.23	0.48	0.93	1.07
	OS3	6.79	29.4	20.04	11.24	0.56	35.7	3.48	0.64	0.03	0.23	162	2691	104	9.46	32.80	8.24	0.28	0.96	0.89
	OS1	7.21	32.2	23.87	10.90	2.01	52.9	6.98	0.38	0.04	0.23	321	2967	1100	12.10	40.80	3.02	0.92	0.78	1.74
Jun	OS2	7.13	30.1	24.89	12.45	1.02	56.8	6.05	0.64	0.05	0.94	340	2106	1000	10.20	32.70	1.03	1.02	0.34	1.29
	OS3	7.77	28.8	20.65	1.26	0.18	3.1	5.89	0.84	0.07	0.67	169	1030	900	9.34	28.60	9.01	1.03	0.56	0.89
WHO limit		**6.5 - 8.5**	-	-	**1000**	-	**5**	**3**	**0.003**	**3**	**0.50**	**200**	**250**	**500**	-	**400**	**10**	-	-	

(b)

Month	Spl. Pt	pH	Temp (˚C)	Cond. $\times 10^3$ (μS/cm)	TDS (mg/l)	Salinity (%)	Turb. (NTU)	Fe (mg/l)	Cd (mg/l)	Zn (mg/l)	Mn (mg/l)	Ca^{2+} (mg/l)	Cl^- (mg/l)	Total Hard (mg/l)	PO_4^{2-}-P (mg/l)	SO_4^{2-} (mg/l)	NO_3^--N (mg/l)	DO (mg/l)	BOD (mg/l)	COD (mg/l)
	OS4	6.56	29.7	610	402	0.02	2.6	0.86	0.01	0.01	0.47	25	75	120	10.00	28.80	10.89	1.64	0.08	1.68
Jan	OS5	7.69	29.9	870	450	0.04	3.0	0.78	0.03	0.02	0.46	26	78	130	8.58	30.10	20.78	1.58	0.05	1.32
	OS6	7.84	30.1	790	305	0.03	2.7	0.87	0.02	0.03	0.51	28	73	140	5.02	29.30	10.41	1.61	0.06	1.93
	OS4	8.13	27.9	830	308	0.13	4.3	0.45	BD	BD	0.15	28	87	109	0.64	33.80	0.35	0.75	0.14	1.49
Feb	OS5	7.89	28.8	740	360	0.09	3.7	0.41	BD	BD	0.13	25	79	108	0.34	27.00	0.41	0.69	0.15	1.85
	OS6	7.97	29.5	630	415	0.08	4.1	0.32	BD	BD	0.18	25	85	108	0.15	27.00	0.95	0.73	0.11	1.34
	OS4	8.02	28.6	670	424	0.05	3.2	0.14	BD	BD	0.18	24	59	109	0.18	23.70	0.56	0.48	0.12	1.06
Mar	OS5	7.98	29.1	710	209	0.08	3.0	0.12	BD	BD	0.16	23	76	107	0.16	28.40	0.82	0.67	0.09	0.98
	OS6	6.99	28.4	740	322	0.07	2.3	0.13	BD	BD	0.12	24	74	108	0.19	29.70	0.68	0.63	0.07	1.06
	OS4	7.94	30.4	850	428	0.01	45.1	0.21	BD	BD	0.34	32	97	120	2.04	25.50	1.08	1.03	0.09	1.23
Apr	OS5	6.89	29.8	980	354	0.03	7.0	0.19	BD	BD	0.18	24	87	110	3.78	30.60	1.05	1.12	0.03	0.84
	OS6	7.58	28.5	780	352	0.02	7.1	0.19	BD	BD	0.92	28	59	100	1.78	29.40	0.96	0.94	0.06	0.67
	OS4	6.97	27.8	1080	122	0.08	4.0	0.12	BD	BD	0.46	62	89	100	1.05	18.90	0.23	0.26	0.03	0.21
May	OS5	7.02	30.4	690	328	0.02	4.7	0.17	BD	BD	0.28	59	97	102	1.89	19.10	0.34	0.96	1.04	0.12
	OS6	7.12	31.2	790	425	0.04	2.7	0.23	BD	BD	0.61	70	76	104	1.29	17.90	1.23	0.85	0.38	0.14
	OS4	6.69	28.4	1903	102	0.05	2.9	1.23	BD	BD	0.21	34	105	80	0.69	16.10	21.02	1.05	0.34	0.28
Jun	OS5	7.56	29.6	1480	403	0.03	2.0	1.02	BD	BD	0.16	42	100	70	1.05	17.20	20.03	1.01	0.45	0.31
	OS6	8.12	31.2	1263	324	0.04	2.7	0.23	BD	BD	0.61	70	76	60	1.29	18.20	8.56	0.85	0.38	0.14
WHO limit		**6.5 - 8.5**	-	-	**1000**	-	**5**	**3**	**0.003**	**3**	**0.50**	**200**	**250**	**500**	-	**400**	**10**	-	-	

BD: below detection. Distances of sample sites from landfill: OS4: 220 m; OS5: 230 m; OS6: 250 m.

4.3. Comparison of Data with WHO and UNEP Values

Compared to WHO [22] and WHO/UNEP [23] values, leachate and river water in the present study appear to have fairly high conductivity and, to some extent, high Mn, Ca and Cl contents. Leachate, however, registered total hardness values above WHO guideline values whereas corresponding river water values were below WHO values (see **Table 2**). Comparatively low values in river water than leachate probably reflect the extent of dilution in the river water compared to the narrower, low volume and channelized leachate.

4.4. Correlation Coefficients

Table 3 gives the correlation coefficients between measured parameters determined in leachate and river water samples. In leachate samples, strong to moderate positive correlations appear to exist mainly between NO_3^--N and Zn (0.96), NO_3^--N and Fe (0.85), NO_3^--N and PO_4^{2-}-P (0.67), Zn and Fe (0.86), Cl^- and each of TDS (0.69), salinity (0.74) and turbidity (0.60) and PO_4^{2-}-P and Fe (0.75). Ca^{2+} also correlates positively with turbidity (0.73) as are Cd and temperature (0.75) and turbidity and salinity (0.69). Similar positive correlations also exist between SO_4^{2-} and each of Fe (0.64) and PO_4^{2-}-P (0.78). Other relationships vary from weak to only slightly positive or negative (**Table 3**). The river water samples also show positive relationships between NO_3^--N and DO (0.61), SO_4^{2-} and COD (0.79), PO_4^{2-}-P and DO (0.85), total hardness and each of SO_4^{2-} (0.69) and COD (0.74), Fe and NO_3^--N (0.89), Fe and DO (0.64) and conductivity and NO_3^--N (0.65). Negative correlations are also shown by the pairs

Table 3. Correlation coefficients between measured parameters in landfill leachate (above) and river water samples (below), Accra, Ghana.

	pH	Temp. (˚C)	Cond. (mS/cm)	TDS (mg/l)	Salinity (%)	Turb. (NTU)	Fe (mg/l)	Cd (mg/l)	Zn (mg/l)	Mn (mg/l)	Ca^{2+} (mg/l)	Cl^- (mg/l)	Total Hard (mg/l)	PO_4^{2-}-P (mg/l)	SO_4^{2-} (mg/l)	NO_3^--N (mg/l)	DO (mg/l)	BOD (mg/l)	COD (mg/l)
pH		–0.45	–0.20	–0.09	–0.11	–0.16	–0.25	–0.52	–0.35	–0.15	0.13	–0.32	0.19	–0.08	0.05	–0.28	–0.10	–0.36	0.00
Temp. (˚C)	0.07		–0.11	0.30	0.29	0.26	0.55	0.75	0.46	–0.24	0.29	0.40	0.29	0.16	0.39	0.30	0.07	0.08	0.15
Cond. (mS/cm)	–0.24	–0.08		0.17	0.20	0.04	0.13	0.13	0.19	0.20	–0.25	–0.02	0.12	0.12	–0.21	0.22	0.47	0.28	0.54
TDS (mg/l)	0.30	0.50	–0.50		0.44	0.49	0.16	0.24	0.03	–0.32	0.32	0.69	0.09	0.12	0.29	–0.01	–0.11	0.05	0.29
Salinity (%)	0.32	*–0.61*	–0.09	–0.34		**0.69**	0.13	0.20	–0.08	0.14	0.53	**0.74**	*0.58*	0.31	0.49	–0.13	0.01	–0.07	0.41
Turb. (NTU)	0.20	0.21	–0.07	0.23	–0.33		–0.05	0.17	–0.26	0.17	0.73	*0.6*	0.51	0.20	0.37	–0.20	–0.26	–0.24	0.07
Fe (mg/l)	–0.22	–0.01	*0.56*	–0.10	–0.16	–0.21		0.47	**0.86**	–0.37	–0.20	0.24	0.16	**0.75**	*0.64*	**0.85**	0.25	0.00	0.32
Cd (mg/l)	–	-	-	-	-	-	-		0.52	–0.04	0.09	0.35	0.30	0.19	0.33	0.37	–0.01	0.10	–0.08
Zn (mg/l)	-	-	-	-	-	-	-	-		–0.37	–0.46	0.02	–0.08	0.58	0.43	**0.90**	0.26	0.08	0.17
Mn (mg/l)	–0.05	0.32	–0.06	0.10	–0.48	0.03	–0.06	-	-		0.26	–0.08	0.35	–0.20	–0.38	–0.45	0.41	0.03	–0.03
Ca^{2+}(mg/l)	–0.16	0.45	0.28	–0.16	–0.17	–0.08	–0.19	-	-	0.39		0.42	**0.71**	–0.04	0.29	–0.51	–0.12	–0.17	–0.07
Cl^-(mg/l)	–0.28	0.06	*0.59*	–0.32	0.00	0.28	0.37	-	–	–0.43	0.23		0.34	0.37	0.50	–0.02	–0.06	0.15	0.18
Total. Hard (mg/l)	0.04	–0.03	**–0.67**	0.29	–0.02	0.21	–0.04	-	-	0.01	–0.53	–0.30		0.30	0.39	–0.17	0.23	–0.03	0.12
PO_4^{2-}-P (mg/l)	–0.31	0.30	–0.18	0.32	–0.46	–0.02	0.43	-	-	0.34	–0.19	–0.13	0.50		0.78	*0.67*	0.14	–0.01	0.19
SO_4^{2-} (mg/l)	0.26	–0.27	*–0.61*	0.27	0.27	0.07	–0.14	-	-	–0.09	**–0.76**	–0.43	**0.69**	0.33		0.46	–0.23	–0.30	0.02
NO_3^--N (mg/l)	–0.18	0.14	*0.65*	–0.05	–0.31	–0.21	**0.89**	-	-	0.06	–0.03	0.32	–0.20	0.46	–0.27		0.11	0.07	0.24
DO (mg/l)	–0.19	0.46	0.02	0.32	–0.57	0.05	*0.64*	-	-	0.29	–0.24	0.04	0.40	**0.85**	0.26	*0.61*		0.31	0.37
BOD (mg/l)	–0.18	0.40	0.20	–0.02	–0.26	–0.13	0.04	-	-	–0.05	*0.59*	0.43	–0.44	–0.19	*–0.62*	0.09	–0.02		0.34
COD (mg/l)	0.33	–0.18	–0.53	0.32	0.24	0.11	0.17	-	-	–0.22	*–0.8*	–0.32	**0.74**	0.37	0.79	–0.07	0.36	–0.58	

Boldface = significance at 0.01 level; Italics = significance at 0.05 level.

Ca^{2+} and SO_4^{2-} (–0.76), Ca^{2+} and COD (–0.76), conductivity and total hardness (–0.67), conductivity and SO_4^{2-} (–0.61), temperature and salinity (–0.61) and SO_4^{2-} and BOD (–0.62). Other pairs of parameters seem to show little or no correlations with one another (**Table 3**).

4.5. Multivariate Principal Component (PCA) and Cluster Analyses (CA)

4.5.1. PCA and CA for Leachate

The dendrogram for leachate samples (**Figure 2**) suggests three distinct clusters. Cluster 1 consists of pH, Mn, BOD, DO, COD and electrical conductivity (EC). Cluster 2 comprises TDS, Cl, Salinity, total hardness, Ca, turbidity and Cd whilst cluster 3 is made up of temperature, SO_4^{2-}, PO_4^{3-}, Fe, NO_3^- and Zn.

Varimax rotation of the landfill leachate data is provided in **Table 4** and the scree plot in **Figure 3**. Principal component 1 (PC1), which explains ~23% of the variance, consists of TDS, salinity, turbidity, Ca, Cl^-, total hardness & SO_4^{2-}. High loading on Ca and total hardness suggests that calcium probably contributes mostly to the hardness of the leachate whereas SO_4^{2-} and Cl^- also suggest increased significance of agricultural and/or organically derived inputs. PC2 loading comprises Fe, Zn, PO_4^{3-}, SO_4^{2-} and NO_3^-. There is strong correlation between cluster 3 and PC2 suggesting that the presence of the metals Fe and Zn in leachate could have come from a common source in the landfill waste or material. PO_4^{3-}, SO_4^{2-} and NO_3^- are all likely sourced from agricultural or organic wastes in the landfill pile. PC3 is controlled by TDS, DO, BOD and COD whilst PC4 consists of pH, temperature and Cd. There is a negative loading of pH as compared to a positive loading of Cd. PC5 loading is made up of Mn, total hardness and DO.

4.5.2. PCA and CA for River (Densu) Water

The interrelationships between the various parameters measured for the river water samples are given by the dendrogram (**Figure 4**) three different clusters. Cluster 1 consists of mainly SO_4^{2-}, COD, total hardness, TDS, pH, turbidity and salinity. Cluster 2 comprises PO_4^{2-}, DO, temperature and Mn whilst cluster 3 is made up of Fe, NO_3^-, electrical conductivity, Cl, Ca, and BOD.

Table 5 and scree plot (**Figure 5**) suggest that up to ~89% of the original mean logs of the dataset is gathered in the first six components with Eigen values > 1. Principal Component 1 (PC1) gives ~27% of the variance and the parameters that are loaded in this component include electrical conductivity, Ca, total hardness, SO_4^{2-} and BOD. High loading of sulphate to water chemistry suggests contribution from agricultural activities such as use of fertilizers. PC2 consists of mainly electrical conductivity, Fe, NO_3^- and DO. High loadings of Fe may suggest dissolution of Fe-bearing bedrock in river water since the Densu River is known to drain iron-rich Birimianmetasedimentary and metavolcanic rocks in the eastern region of Ghana. Again, anthropogenic activities may not also be ignored as suggested by the high loading on nitrate. PC3 is loaded with temperature, TDS, salinity and DO whilst PC4, PC5 and PC6 are loaded with Cl^-, pH and turbidity, respectively.

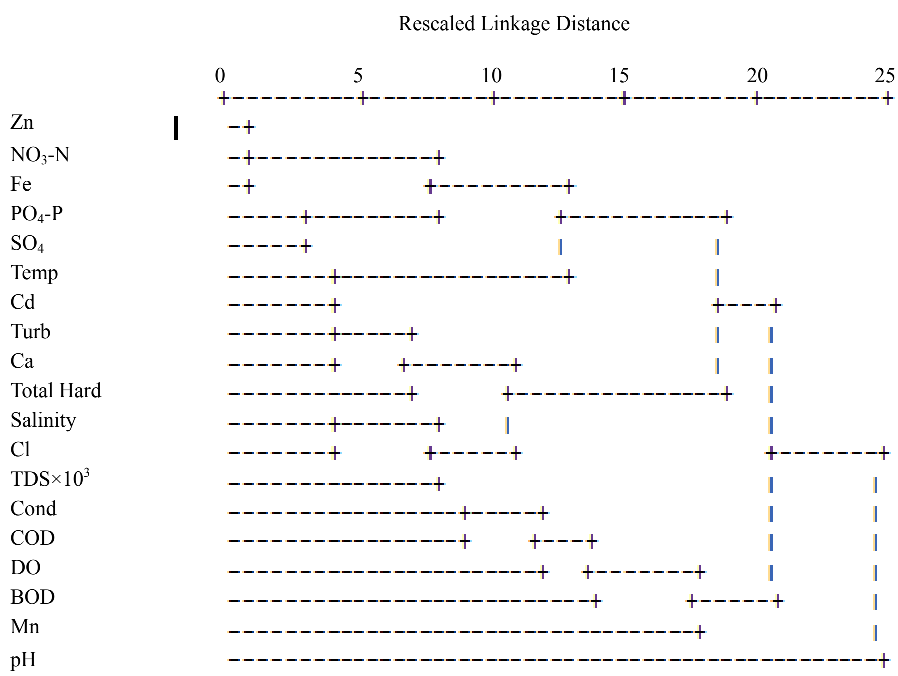

Figure 2. Dendrogram of landfill leachate parameters.

Table 4. Rotated component matrix for landfill leachate data.

	Component				
	1	2	3	4	5
pH	0.02	0.00	–0.22	**–0.83**	0.05
Temp	0.32	0.34	–0.09	**0.73**	–0.02
Cond	0.01	0.08	**0.80**	0.02	0.15
TDS	**0.62**	0.03	0.23	0.15	**–0.55**
Salinity	**0.87**	0.12	0.21	0.02	0.06
Turb	**0.86**	–0.07	–0.10	0.08	0.01
Fe	0.03	**0.92**	0.13	0.27	–0.03
Cd	0.19	0.32	–0.09	**0.81**	0.09
Zn	–0.27	**0.82**	0.14	0.39	–0.06
Mn	0.10	–0.42	0.17	0.04	**0.78**
Ca^{2+}	**0.81**	–0.23	–0.29	0.01	0.27
Cl^{-}	**0.78**	0.11	0.12	0.31	–0.26
Total Hard	**0.67**	0.17	–0.02	–0.01	**0.62**
PO_4-P	0.25	**0.85**	0.09	–0.07	0.05
SO_4^{2-}	**0.51**	**0.75**	–0.31	–0.02	–0.11
N-NO^{3-}	–0.28	**0.86**	0.16	0.22	–0.18
DO	–0.14	0.15	**0.62**	0.06	**0.56**
BOD	–0.13	–0.16	**0.60**	0.35	–0.07
COD	0.23	0.21	**0.79**	–0.14	-0.10
Eigenvalues	4.296	4.159	2.471	2.429	1.832
% of Total variance	22.61	21.89	13.00	12.78	9.64
Cumulative %	22.61	44.50	57.51	70.29	79.93

5. Discussion and Environmental Implications

The eight ionic species dominant in leachate, *i.e*. Ca, Cl, SO_4 (PC1), Fe, Zn, PO_4, SO_4, NO_3 (PC2), Cd (PC4) and Mn (PC5) (**Table 4**) suggest decomposition of landfill materials through a combination of physico-chemical (inorganic) and biological (organic) processes and subsequent release into the effluent discharge or leachate. The seasonally wet and dry climate, together with the generally heterogeneous, unsorted or mixed nature of refuse dumped at the landfill site, may have enhanced leaching of both organic and inorganic constituents of decomposing waste by percolating rain water. Tigme [14] characterized waste at the Oblogo landfill site into dominantly organic components (70%) followed successively by inert material (13%), plastics (9%), metal scraps (4%), paper (3%) and textile products (1%). The Togo host rocks [15] within which the landfill is situated is dominantly composed of quartzite and sandstone and may, hence, not contain significant amounts of ionic species such as observed in leachate, suggesting that most of these species were derived from refuse at the landfill. Though the proportion metals in the waste stream is low [14], the fairly significant presence of some metals in leachate may be an indication of the extent of decomposition of the metallic constitutents of the waste. The relatively high Ca, Cl and nutrient contents are likely reminiscent of decomposition from the high agricultural or organic inputs of waste.

In river water, Fe, Mn, SO_4 and NO_3 are the most sig-

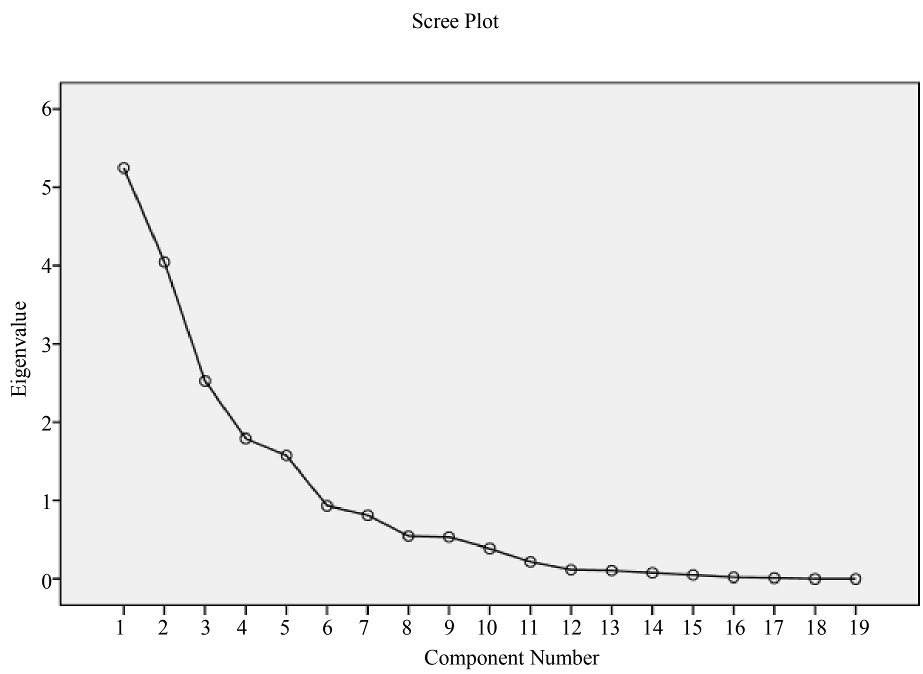

Figure 3. Scree plot of eigenvalues for landfill leachate.

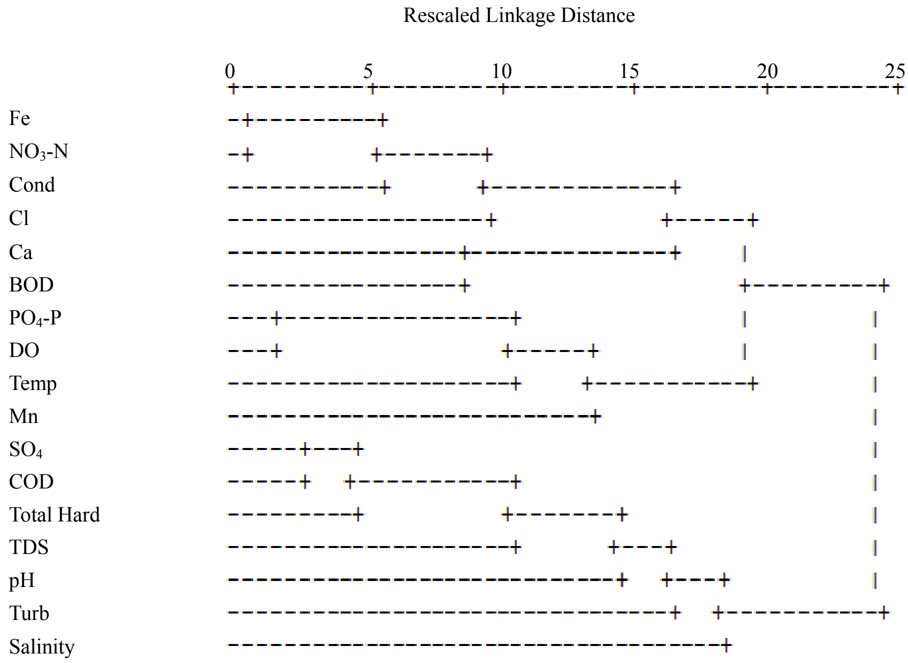

Figure 4. Dendrogram for river water data.

Table 5. Rotated component matrix for river water data.

	Component					
	1	2	3	4	5	6
pH	0.17	–0.14	0.12	0.00	**0.92**	0.06
Temp	–0.29	0.07	**0.85**	0.14	0.01	0.15
Cond	**–0.58**	**0.66**	–0.38	–0.08	0.01	0.17
TDS	0.29	–0.09	**0.77**	0.06	0.28	0.03
Salinity	0.18	–0.25	**–0.57**	–0.39	0.34	–0.37
Turb	0.11	–0.15	0.17	–0.03	0.09	**0.95**
Fe	0.06	**0.95**	0.00	–0.13	–0.11	–0.09
Mn	–0.16	0.02	0.20	**0.91**	–0.10	0.04
Ca^{2+}	**–0.85**	–0.16	0.19	0.18	–0.12	–0.04
Cl^-	–0.38	0.34	–0.07	**–0.65**	–0.24	0.41
Tot Hard	**0.83**	–0.15	0.22	0.04	–0.24	0.09
PO_4-P	0.45	0.45	0.43	0.32	–0.44	–0.05
SO_4^{2-}	**0.91**	–0.15	–0.02	0.05	0.08	–0.05
NO_3^--N	–0.12	**0.96**	0.08	0.03	–0.04	–0.07
DO	0.36	**0.65**	**0.53**	0.18	–0.29	0.05
BOD	**–0.69**	–0.03	0.43	–0.37	–0.19	–0.15
COD	**0.92**	0.08	0.09	–0.11	0.19	–0.03
Eigenvalue	4.664	3.188	2.619	1.753	1.558	1.324
% of Total Variance	27.43	18.76	15.41	10.31	9.16	7.79
Cumulative %	27.43	46.19	61.60	71.91	81.07	88.86

nificant species. Because the Densu River drains a significant portion of Fe and Mn-containing Birimian rocks [15], these two elements could have been sourced from the dominant underlying geological formation. Again, increased human activities such as use of fertilizers in subsistence agriculture in the Densu River catchment area may have contributed significant SO_4 and NO_3 contents to the river water.

Farquhar [1] who provided data on expected contaminant types and ranges of concentrations in leachate as function of refuse age noted chloride concentrations of 1000 - 3000 mg/l for landfills in age category 0 - 5 years. Because chloride contents obtained in leachate in this study agrees fairly well or falls within this range, it could reasonably be predicted that various physico-chemical and biological decomposition processes within the landfill may result in increased pollutant levels in leachate which would, in turn, be shed into the surrounding media for well some time before decreases in concentrations could be expected as the landfill ages [1]. As observed by Mizumura [24], chloride ion is non-reactive, non-sorptive and has no redox or precipitation. This suggests that much of the chloride in the leachate plume will find its way into the surrounding river and groundwater as well as soils.

Because the rocks in which the landfill is situated are highly bedded [15], the landfill not engineered [11] andno leachate collection systems in place, continuous discharge of leachate may pose serious threats to the surrounding soils, water bodies, the Densu Ramsar wetland area and also possibly on the health of people who

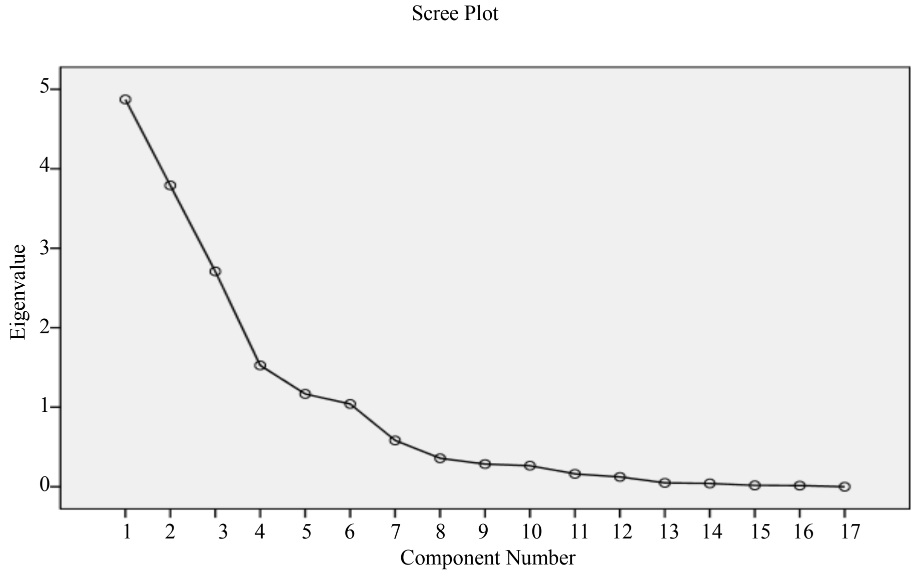

Figure 5. Scree plot of eigenvalues for river water.

depend a lot on the environmental resources of the area [25]. Furthermore, local communities and especially the urban poor who live around the landfill utilize water (from rivers, streams, shallow wells and boreholes) and soils for domestic and subsistence agriculture. Others also undertake fishing activities as a means of livelihood [26]. Food crops grown and the fish obtained from these areas mainly go to feed the urban population. Leachate also egresses through many low-income residential areas, presenting potential threats to the health of people especially children who constantly attend school in or play around such leachate contaminated areas.

Authors including Combs Jr. [27], Nordberg & Cherian [28], Frew [29] and Kurniawan [30] have pointed out adverse health effects of substances such as cadmium, chloride and zinc all of which occur in the leachate samples studied. As noted by Oteng-Yeboah [17], the wetland is known to be very rich in various species of fauna and flora and therefore deserves maximum protection, not the least from contamination through landfill leachate which could be controlled or managed. Loss of biodiversity in the internationally recognized Densu Wetland as a result of pollution from the landfill leachate could also not be entirely ruled out. Assessment techniques to provide information on early warning indicators of pollution in the wetland, as suggested by Van Dam *et al*. [31], could provide an important first step towards sustainable management of this ecologically important wetland and surrounding environment. Kao *et al*. [32] also suggested using network Geographic Information System (GIS) for the siting of landfills in order to reduce the potential for spread of infection through run off during rain as well as groundwater contamination.

Effects of landfill leachate on surrounding media, some resulting directly from pollutants in leachate or through bioaccumulation of leachate constituents in living organisms over time, have been documented by workers such as Schrab *et al*. [6], Stephens *et al*. [33], Kjeldsen *et al*. [4]. Kurniawan *et al*. [34-36] have worked extensively on recalcitrant contaminants in landfill leachate especially those that pose serious hazards not only to living organisms but also to public health in the long term. In Uganda, Nigeria and many other countries [8,9,37], the potential effect of leachate on surface and groundwater resources could be very significant. Rocks within which the landfill is located have a well bedded structure [38,39] and, in addition, typically weather into permeable sandy to silty soils. In addition, absence of bottom liners and artificially constructed drains to trap and channel leachate into channelized flow, respectively, likely promote increased infiltration of leachate into the surrounding environment.

Research to investigate the distribution and possible attenuation of hazardous substances in uncontrolled leachate from landfills [40], especially if done at many such landfills in the Accra Metropolis and throughout the country, could help provide invaluable data for remediation efforts. In addition, assessment of the spatial variability in leachate migration from landfills along the lines done by Kjeldsen [4] in Denmark could help identify plumes of pollution that may be contaminating various media around landfills. Finally, because chloride is non-reactive, non-sorptive and has no redox or precipitation, it is often used as a tracer element in leaching studies in soils [24,41,42]. Mizumura [24], in particular, investigated the influence of leachate plumes from sanitary landfills on groundwater by determining the concentration of chloride ion in the groundwater, soil water and river and observed that most of the leachate plume was discharged into a river whilst the remainder infiltrated into the ground through the weathered geological layer near the landfills. Such investigations may also be relevant in the present situation given that untreated leachate not only directly drains into the Densu River and adjoining Ramsar wetland (at a distance less than 0.3 km from the point or landfill source) but also egresses continuously through households, soils and possibly into the groundwater system in the area.

6. Conclusion

The current study has revealed fairly high levels of ionic constituents including Cl, SO_4, PO_4, NO_3 and moderate to high contents of Ca, Cd and Zn in leachate discharged without treatment from the Oblogo landfill site in Accra into the immediately surrounding environment, a situation which makes the area very vulnerable to pollution. The significantly high concentrations of chloride and, to some extent, other chemical species, present formidable challenges that may need to be addressed in order to minimize possible short- and long-term stresses on the immediate environment. It is suggested that simple but cost-effective techniques such as construction of manually excavated holes or ponds ("dug-outs") in the vicinity of the landfill to impound leachate for considerable period of time could provide a necessary first step towards facilitating natural breakdown or settling of some constituents in leachate. The "environmental cost" of any such initiative could, under the circumstances, be a much better option than the present indiscriminate and uncontrolled discharge of leachate into the immediate ecologically important Ramsar environment. Ultimately, the risks posed by possible organic contaminants, pathogenic microorganisms and other toxic substances that may additionally be present in leachate would have to be analyzed and/or monitored to also prevent or minimize their impact on the environment. In addition, it may be necessary to study the leachate migration patterns in tandem with leachate composition to gather information for future planning and remediation efforts.

7. Acknowledgements

Data used in the study was part of the M.Phil. Thesis work submitted by J. T. to the University of Ghana in partial fulfillment of the requirements for the award of the Master of Philosophy in Environmental Science Degree for which we are very grateful. The Environmental Protection Agency of Ghana (EPA Ghana), Accra Metropolitan Assembly (AMA), the Water Research Institute (WRI) of the Council for Scientific and Industrial Research (CSIR), many other organizations and people who contributed in diverse ways towards data collection, analysis, and interpretation as well as during preparation of the manuscript are all gratefully acknowledged. Finally, constructive criticisms by anonymous reviewers helped improve the quality of the paper considerably.

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Journal of Water Resource and Protection, 2012, 4, 634-637
doi:10.4236/jwarp.2012.48073 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Estimation of Water Environmental Capacity Considering Hydraulic Project Operation in the Xiangyang Reach of the Han River, Central China

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Received May 28, 2012; revised June 30, 2012; accepted July 8, 2012

ABSTRACT

Using the Xiangyang Reach of the Han River as an example, this paper evaluates the changes of water environmental capacity after the implementation of Cuijiaying Hydro-junction project. The allowable pollutant loads entering the Xiangyang Reach were estimated using two-dimensional steady state water quality model with different data sets. The water environmental capacity has declined in the reservoir area of the Cuijiaying Hydro-junction project during the low-flow period; it is appearing to increase slightly in the upper and lower stream of this reservoir. However, the state of flow may turn into the state of reservoirs flow in the reservoir area, and the changes of hydrological regime may cause the water flow and the nutrient contents suitable for the occurrence of ecological environment problems.

Keywords: Han River; Water Environmental Capacity; Cascade Development; Pollutant Loading

1. Introduction

The Han River is the longest tributary of the Yangtze River, with a watercourse length of 1577 km and a total drainage area of 159,000 km^2. The Han River Basin is one of the major agricultural production bases in China, and plays a significant role in regional economic development. Due to its plentiful water resources [1], the Basin cascade development is being implemented [2]. Meanwhile, changes of hydrological regime may inevitably impact the aquatic environment of the Han River, especially to the water environmental capacity. Moreover, the rapid urbanization and industrialization are beginning to affect water quality in this area. From 1992 to 2005, large-scale algal blooms had taken place five times in the middle and lower reaches of the Han River because of the ecological environment degradation [3].

Total amount control for water pollutants is one of the main methods to resolve water environmental problems in China [4,5]. Comprehensively understanding the water environmental capacity, namely the maximum allowable pollutant loads discharging into the water body, is a prerequisite to effective pollution control. With the aim of providing a valuable aid to the strategies of water pollution control, the influence of water conservancy projects on water environmental capacity is increasingly draw more attention in China [6-8].

The current study aims to estimate the water environmental capacity through a case study of the Xiangyang Reach of the Han River in Central China. Using two-dimensional steady state water quality model, we evaluated the variation of water environmental capacity considering the dam operation in this area.

2. Study Area

Xiangyang city is located in the middle reach of the Han River (**Figure 1**). With a total area of 19,727.68 km^2, it has a population of around 5,888,786 in 2009. The Han River runs through the city from north to south with a length of 195,000 m. There are five tributaries in the Xiangyang Reach: South River, North River, Xiaoqing River, Tangbai River, and Man River. The Cuijiaying Hydro-junction project, located in downstream of the Xiangyang urban area, is the third step of cascade development in the middle and lower reaches of the Han River [9]. Normal storage level of the project is 62.73 m. The project, which is mainly used for shipping and electricity generation, has been completed in July, 2010.

3. Methods

According to the actuality of total amount control for water pollutants in China [10], chemical oxygen demand (COD) and ammonia nitrogen (NH_3-N) were selected as the calculation factors in this study. We divided the Xiangyang Reach of the Han River into six computa-

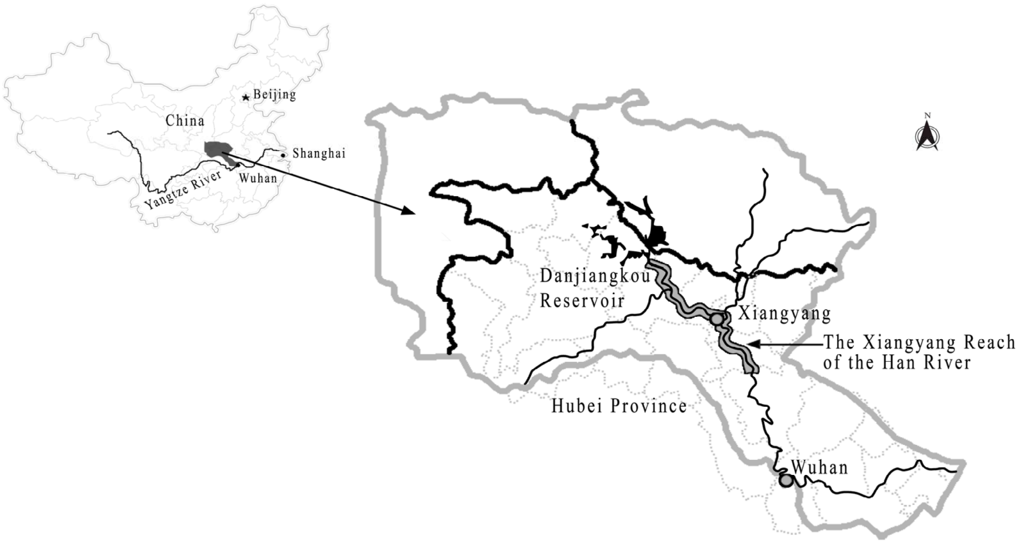

Figure 1. Study area.

tional units based on the water environmental function zone of Hubei Province [11], as shown in **Table 1**. The point source pollution data in 2007 for this study was obtained from the Xiangyang Environmental Monitoring Station.

As the river width of the Han River is usually in excess of 200 meters, it is necessary to consider the variation of pollutants in longitudinal dispersion and horizontal diffusion [10]. Two-dimensional water quality model, therefore, is suitable for calculation of water environmental capacity under given the unique hydrological conditions within the Xiangyang Reach.

Under steady flow conditions, fundamental equation of two-dimensional water quality model is written as:

$$E_x \frac{\partial^2 C}{\partial x} + E_y \frac{\partial^2 C}{\partial y} - u_x \frac{\partial C}{\partial x} - u_y \frac{\partial C}{\partial y} - kC = 0 \quad (1)$$

where u_x is the longitudinal velocity (m/s); u_y is the horizontal velocity (m/s); E_x is the longitudinal dispersion coefficient (m^2/s); E_y is the horizontal diffusion coefficient (m^2/s); C is the contaminant concentration (mg/L); K is the decay coefficient (1/d); x is the longitudinal coordinates (m); y is the horizontal coordinates (m).

Supposing u_y and E_x are approximate to zero under conditions of little changes in water depth, Equation (1) is converted to the following expression:

$$C(x,y) = \frac{M}{h\sqrt{4u\pi E_y}} \exp\left(-\frac{uy^2}{4E_y x}\right) \exp\left(-k\frac{x}{u}\right) + C_0 \exp\left(-k\frac{x}{u}\right) \quad (2)$$

where M is the pollutant emission rate (g/s); C_0 is the background pollutant concentration of the river (mg/L); C is the pollutant concentration (mg/L); K is the decay coefficient (1/d); x is the longitudinal coordinates (m); y is the horizontal coordinates (m); h is the average water depth (m); u is the average longitudinal velocity (m/s).

Assuming the degradation of background pollutant concentration is negligible, the model which can be applied to estimate water environmental capacity in the Xiangyang Reach gives the following formula:

$$W = \left(C(x,y) - C_0\right) H \sqrt{u\pi x E_y} \exp\left(\frac{y^2 u}{4E_y x} + k\frac{x}{u}\right) \quad (3)$$

where W is the water environmental capacity (t/a).

30Q10 is selected as the design flow (314 m^3/s) in the light of data from the Xiangyang hydrometric station, and the designed flow velocity is 0.31 m/s. The river depth and width are derived from our investigation (**Table 2**).

The horizontal diffusion coefficient (E_y) is usually determined by the empirical correlation or field survey method. In this study, we adopted the following empirical correlations [12]:

Table 1. Computational units in the Xiangyang Reach of the Han River.

Computational unit	Environmental functional category	Length (m)
Shenwan-Xianrendu	II	38.5
Xianrendu-Baijiawan	II	47.2
Baijiawan-Zhakou	III	14
Zhakou-Qianying	III	12
Qianying-Yujiahu	III	7.6
Yujiahu-Guo'an	II	53.2

Table 2. Average river width and depth of each computational unit in the Xiangyang Reach.

Computational unit	Sewage outlet and tributary	Width (m)	Depth (m)
Shenwan-Xianrendu	Jiangshan	180	2.8
	Damingqu	180	2.8
Xianrendu-Baijiawan	Jinhuan	235	2.7
Zhakou-Qianying	Xiaoqing River	323	2.42
	Tianjiu	160	3.3
	Nanqu	323	4.52
	Yuliangzhou	160	3.3
	Xiangyang power plant	390	3.35
Yujiahu-Guoan	Yakou	390	3.35

$$E_y = a_y \cdot H \cdot U^* \tag{4}$$

$$U^* = \sqrt{gHI} \tag{5}$$

where a_y is non-dimensional coefficient; H is average river depth (m); U^* is frictional velocity (m/s); g is gravity acceleration (m/s^2); I is gradient of the river (‰).

According to the Surface water environmental capacity of Hubei Province Technical Report, decay coefficient (K) is assigned values as follows: K_{COD} = 0.18d – 1， $K_{NH_3\text{-N}}$ = 0.08d – 1. The upper limit value of water quality standard is selected as the background pollutant concentration of the Xiangyang Reach.

4. Results and Discussion

4.1. Pollution in the Xiangyang Reach of the Han River

In 2007, the quantity of waste water entering the Xiangyang Reach was 130,414,000 t. The amount of COD was 38,741 t, and NH_3-N was 4418.43 t. **Table 3** shows the amount of pollutants entering the Xiangyang Reach from various pollution sources. Municipal domestic sewage was the primary point source pollution, accounting for nearly 64% of COD and 62% of NH_3-N, respecttively. **Table 4** shows the quantity of pollutants from point source pollution in each computational unit. The pollutant loading mainly originates from Xiangyang urban area with the highest population densities. The chemical fiber industry, textile industry, pharmaceutical industry, and chemical industry are four major industrial pollution sources, whose COD accounted for nearly 76% of industrial pollution.

4.2. Changes of the Hydrological Regime

The Cuijiaying Hydro-junction project has mainly exerted effects on the hydrological regime of the following two areas: the reservoir area and downstream of this project. The flow of reservoir area (backwater area) is mainly influenced by the upstream flow and runoff, and the designed flow of this area is 554 m^3/s. The water level increased from 2.66 m to 62.73 m, and the designed flow velocity is 0.138 m/s. The state of flow may turn into the state of reservoirs flow in the reservoir region, and the hydrological regime changes cause the water flow and the nutrient contents suitable for the occurrence of ecological environment problems.

Meanwhile, the minimum discharging downstream flow (490 m^3/s) is used for designed flow of downstream of the dam. The designed flow velocity is 0.57 m/s, and the water level has increased by 0.15 m.

Table 3. The quantity of pollutants from various pollution sources entering the Xiangyang Reach in 2007.

Pollution source	COD (t)	NH_3-N (t)
Municipal domestic sewage	21905.2	2267.7
Industrial sewage	12388.5	1418.2
Animal breeding	1758.2	353.6
Rural area sewage	1735.2	188.2
Farmland runoff	951.1	190.3
Urban runoff	2.7	0.3

Table 4. The amount of pollutants from point source pollution in each computational uint in 2007.

Computational unit	COD (t)	NH_3-N (t)
Shenwan-Xianrendu	6802.6	553.4
Xianrendu-Baijiawan	11640.6	1256.7
Zhakou-Qianying	10608.1	949
Qianying-Yujiahu	785.4	653.1
Yujiahu-Guo'an	4401	814.4

4.3. Water Environmental Capacity in the Xiangyang Reach of the Han River

Based on the specified water quality and hydrological patterns, the water environmental capacity in the Xiangyang Reach of the Han River was estimated. The results are shown in **Table 5**, which are the maximum allowable pollutants loading entering the river (there is no outfall in the unit of Baijiawan-Zhakou, so this unit does not need to calculate). The water environmental capacity after the implementation of the cascade project is shown in **Table 6**. The water environmental capacity has declined in the reservoir area (Zhakou-Qianying) during the low-flow

Table 5. Water environmental capacity in the Xiangyang Reach of the Han River.

Computational unit	COD (t/a)	NH_3-N (t/a)
Shenwan-Xianrendu	3614.1	576.34
Xianrendu-Baijiawan	1904.4	177.11
Zhakou-Qianying	44986.2	2278.76
Qianying-Yujiahu	3570.1	224.13
Yujiahu-Guo'an	9852.23	197.46

Table 6. Water environmental capacity in the Xiangyang Reach after the implementation of the Cuijiaying project.

Computational unit	COD (t/a)	NH_3-N (t/a)
Shenwan-Xianrendu	3728.3	592.51
Xianrendu-Baijiawan	1905.5	177.6
Zhakou-Qianying	42661.3	2112.2
Qianying-Yujiahu	3612.3	235.1
Yujiahu-Guo'an	10007.6	203.78

period, and it is appearing to increase slightly in the upper and lower stream (Shenwan-Baijiawan, Qianying-Guo'an) of this project.

5. Conclusion

Applying two-dimensional steady state water quality model, we estimated the water environmental capacity of the Xiangyang Reach. Owing to the influence of cascade development, there is a decline of water environmental capacity in the reservoir region, also a growth in the upper and lower stream of the dam. However, the flow rate has declined in the reservoir area, and what should be done is to look for the influence of the cascade development on the aquatic eco-environment. The reservoir area should be a priority region for pollution control. The case study of the Xiangyang Reach shows that municipal domestic sewage was one of the major point source pollution, which contributed the most COD load entering the Han River in 2007. Improvement of sewage treatment facilities should be considered by policy makers in this area.

6. Acknowledgements

This study was supported by the Environmental Protection S&T Program of Hubei Province. The authors thank the Hubei Provincial Environmental Protection Department and Xiangyang Environmental Protection Agency for providing necessary data.

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Journal of Water Resource and Protection, 2012, 4, 638-647
doi:10.4236/jwarp.2012.48074 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Influence of Temperature on Mutagenicity in Plants Exposed to Surface Disinfected Drinking Water

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Received May 31, 2012; revised July 2, 2012; accepted July 14, 2012

ABSTRACT

Disinfection of surface drinking water, particularly water chlorination, produces by-products with potential genotoxic and/or carcinogenic activity. A study carried out at a pilot plant for drinking water disinfection of lake water revealed mutagenic activity of three different disinfectants (sodium hypochlorite, chlorine dioxide and peracetic acid) in different seasons using *in situ* mutagenicity assays, both in animal (micronucleus test) and in plant organisms (anaphase chromosomal aberration and micronucleus tests). The effects of the disinfectants appeared to be modulated by the season of exposure. In this study, we tried to understand if (and to what extent) the temperature parameter could actually play an independent role in the registered seasonal variation of mutagenic effects, neglecting the variation of other parameters, e.g. physical conditions and chemical composition of the lake water. Therefore plants (*Allium cepa* for chromosomal aberration test and *Vicia faba* for micronucleus test) were exposed to the same disinfected lake-water samples at different temperatures (10˚C, 20˚C and 30˚C), according the ones registered during the *in situ* experiment. Long-term exposure at the temperatures of 20˚C (both *Vicia faba* and *Allium cepa*) and 30˚C (*Vicia faba* only) to disinfected waters induced clear mutagenic effects. These results show that temperature is an important variable which should be taken into account when *in situ* exposure of plants is planned for mutagenicity testing. Also, different plant systems clearly show specific temperature ranges suitable for their growth, thereby indicating the need for an accurate selection of the test organism for a specific experimental plan.

Keywords: Clastogenicity/Aneugenicity; *Allium cepa* Aberration Test; *Vicia faba* Micronucleus Test; Temperature Exposure; Water Disinfection

1. Introduction

Drinking water disinfection may produce toxic compounds, particularly when water is obtained from surface sources. Such disinfection by-products (DBP) are formed when disinfectants (e.g. chlorine, ozone, chlorine dioxide, or chloramines) react with naturally occurring organic matter, anthropogenic contaminants, bromide, and iodide. It has been demonstrated that chlorination, the most widely used method of disinfecting water, leads to the formation of numerous mutagenic and/or carcinogenic DBPs [1-4].

Despite the numerous laboratory evidences on the mutagenic or carcinogenic properties for many DBPs, epidemiologic studies to date have revealed only a modest association between DBP exposure and cancer in humans [4,5]. Interesting findings for their significant implications for cancer prevention come from a recent case-control study carried out in hospital which showed strong associations between DBP exposure and bladder cancer among individuals carrying inherited variants in three genes (GSTT1, GSTZ1, and CYP2E1) that code for key enzymes that metabolize DBPs [6].

As far as the mutagenic and genotoxic potential of DPBs is concerned, still incomplete information for many of them is available, particularly for the emerging ones, the levels of which are increased by alternative disinfectants that are being employed (primarily ozone or chloramines) compared to chlorination. However, many emerging DBPs are more genotoxic than some of the DBPs that are submitted to regulation [4,7]. Since the growing body of evidence about the adverse effects of DPBs produced by common disinfectants, alternative disinfection practices have been implemented and in some instances results from extracted organic material in

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drinking water showed to be less mutagenic than extracts from chlorinated water [4,8].

Nevertheless, the full toxicological effects of the complex mixtures of DBPs present in drinking water are largely unknown, except through epidemiologic studies [5], because the greatest majority of mutagenicity studies are carried out on drinking-water extracts or concentrates, largely in Salmonella [9]. In general, only a few of them involved drinking waters prepared by alternative disinfection methods, which showed that alternative disinfecttion methods may be considerably less mutagenic than chlorinated drinking water [10]. For this kind of studies an environmental biomonitoring approach can be adopted, in which *in situ* exposure of bioindicator organisms, principally aquatic animals (fish, mollusks) and plants [9,11,12] is performed.

Plants are unique in their ability to serve as *in situ* monitors for environmental genotoxins that exist either as single or complex forms [13] and to date new plant systems are being proposed for assessing the genotoxic potential on living organisms as a sensitive indicator of water quality complex-mixtures [14-16], besides the traditional plant systems routinely employed for cytogenetic damage. Plant bioassays can detect a wide range of genetic damage, including gene mutations and chromosomal aberrations; in particular, mitotic cells in meristems of plant roots are considered excellent experimental tool for assessing the cytogenetic damage induced by clastogenic and aneugenic environmental pollutants [13,17].

Cytogenetic tests have been successfully performed in plant systems for assessing genotoxic contamination of aquatic environments—especially polluted rivers [18,19] and soil [20-23], as well as for detection of drinking-water mutagenic potential [13,15,24-31]. Among plants, *Vicia faba* and *Allium cepa* are routinely used especially because of their sensitivity to a wide variety of mutagenic compounds [32] and to extremely low doses of X-rays [33]. Other factors, such as high percentage of dividing cells in root tips, uniform size of chromosomes, easy culturing in laboratory conditions and growth under *in situ* exposure, are further advantages which encourage the use of these systems [34].

Vicia faba and *Allium cepa* are widely used for the evaluation of genotoxic potential of disinfected wastewater, surface water, and drinking water [12,16,34-40].

A previous study was carried out to evaluate the genotoxic potential induced by surface water disinfected for drinking usage in root cells of plant organisms exposed *in situ* to unconcentrated water samples, using the *Allium cepa* anaphase chromosomal aberration test and the *Vicia faba* micronucleus test [11,29]. This approach allows waters to be tested without requiring extraction processes/concentration methods of water samples, so as to expose test organisms both in the laboratory and *in situ*. That study was performed at a pilot drinking water treatment plant, located near Lake Trasimeno (Central Italy), where lake water was experimentally disinfected using three alternative compounds: sodium hypochlorite (NaClO), chlorine dioxide (ClO_2) and peracetic acid (PAA). Results indicated that all the disinfection treatments, especially ClO_2 and NaClO, induce a clastogenic/aneugenic effect. All the plant tests gave overall similar results, yet a seasonal correlation was also observed, since a noticeable variability of the mutagenic effect was found among the different sampling months (October, February and June): while in *Allium cepa* raw water resulted to be weakly genotoxic in October and February, the *in situ* exposures carried out in October to disinfected waters exerted the strongest mutagenic effects both in *Vicia faba* and in *Allium cepa*, which were induced by all the compounds employed for drinking water treatment [11]. These differences were thought to be due to either temperature or chemical composition of disinfected water, being the two factors closely correlated.

To verify the first hypothesis, *i.e.*, the influence of temperature on the expression of mutagenic damage, the same samples of raw and NaClO-, ClO_2- and PAA-disinfected water were supplied to the plants (*Allium cepa* for chromosomal aberration test and *Vicia faba* for micronucleus test) at different temperatures (10˚C, 20˚C and 30˚C) in the present laboratory tests following the same protocol as that adopted for *in situ* exposures.

2. Materials and Methods

2.1. Lake Water Sampling and Treatment with Disinfectants

Water taken from Lake Trasimeno underwent sedimentation, filtration and acidification (H_2SO_4, pH 7.0), followed by disinfection with 3 mg/L of sodium hypochlorite (NaClO), chlorine dioxide (ClO_2) and peracetic acid (PAA). Water samples were taken in springtime and used immediately for plant exposures.

The disinfection treatments were as follows:

1) Chlorine dioxide (ClO_2): produced directly in treated water from an 8% $NaClO_2$ solution and a 10% HCl solution using an automatic generator (Tecme S.r.l., Gardolo di Trento, TN, Italy);

2) Sodiumhypochlorite (NaClO), (Solvay S.p.A., Rosignano, LI, Italy): supplied as a 14.5% - 15.5% solution via a membrane pump;

3) Peracetic acid (CH_3COO_2H), (Promox S.r.l., Leggiuno, VA, Italy): supplied as a 15% solution via a membrane pump.

2.2. Water Quality Measurements

Total Organic Carbon (TOC), Adsorbable Organic Halogens (AOX), UV absorbance at 254 nm and UV absorbance at 254 nm after filtration on a metallic filter at 0.45 µm (DUV) were measured in raw and treated water [29,41]. All these parameters indicate the total organic content that can react with disinfectants to produce potentially toxic and carcinogenic compounds.

2.3. *Allium cepa* Test

After root germination under controlled laboratory conditions (20˚C in mineral water) young bulbs of *Allium cepa* of equal size (2 - 2.5 cm in diameter) were exposed to disinfected and raw water for 24 and 72 hours at 10˚C, 20˚C and 30˚C. At the end of exposure roots were cut and fixed in 1:3 acetic acid-ethanol solution and stored in 70% ethanol. Clastogenic effects (chromatin bridges, fragments) and spindle disturbance (vagrants, c-mitosis, multipolar anaphases) were studied in root cells after Feulgen staining [42,43]. The mitotic index (MI) was also evaluated as a measure of cell division rate; MI values lower than 10/1000 was not considered as they are indicative of toxicity. At least 800 anaphase cells per experimental point (40 for each root, 20 slides) for anaphase aberrations and 5000 cells per experimental point (1000 for each root, 5 slides) for the mitotic index were analyzed. Root length was used as an index of toxicity, and modifications in root form (formation of tumours, hook roots, twisted roots) and root consistency were observed [44]. Statistical data analysis was performed by means of the χ^2 test. Raw and treated waters were compared to a negative control (mineral water stored in glass bottles). Maleic hydrazide (10 mg/L) was used as a positive control.

2.4. *Vicia faba* Micronucleus Test

The micronucleus test was performed in secondary root tips of *Vicia faba* [32] which have been exposed to raw and disinfected lake waters for 6 and 72 hours at 10˚C, 20˚C and 30˚C. After the initial 6-hour exposure to the disinfected waters, part of the seedlings were transferred in Hoagland's solution for the next 66 hours (recovery time) and maintained at the corresponding temperatures, according to the experimental protocol (10˚C, 20˚C and 30˚C). Hoagland's solution was also used for roots of the negative control group. After exposure, roots were fixed in 1:3 acetic acid/ethanol mixture and Feulgen stained. Root tips were cut and squashed onto microscope slides. Micronucleus frequency was studied in the proliferating tissue of each root, analysing 5×10^3 cells/root tip, 10 tips/experimental point of secondary roots (5×10^4 proliferating cells). This analysis was carried out after checking the mitotic index, which allowed the study of cell populations with comparable proliferation activity (data not shown). Statistical analysis of data was carried out through the Mann-Whitney test; pair-wise comparesons of micronucleus frequencies were carried out between root cells exposed to the differently-disinfected waters and either raw waters or Hoagland's solution. In addition, a comparison was made between data from 6-hour and 72-hour exposure to the same samples of water. Regression analyses were also performed between temperatures and micronucleus frequencies for each treatment and time of exposure. Maleic hydrazide (10 mg/L) was used as a positive control.

3. Results

3.1. Water Quality Measurements

The results of the physical and chemical analyses are set out in **Table 1**. A high concentration of TOC was observed in raw lake water (7.8 mg/L), similar to that for water disinfected with NaClO and ClO_2. The light increase registered in PAA treated water may be a consesquence of the carbon content of the compound itself. Organic carbon in water is composed of organic compounds in various oxidation states and TOC is the direct expression of total organic content. For disinfected waters, organic compounds may react with disinfectants to produce potentially toxic and carcinogenic compounds.

UV-absorbing organic constituents in a sample absorb UV light in proportion to their concentration. Samples are filtered to control variations in UV absorption caused by particles. UV_{254} absorbance is 0.074 abs/cm and 0.069 abs/cm in raw water and filtered raw water (DUV), respectively. The absorbance value of ClO_2-disinfected water was similar to that of raw water. NaClO- and

Table 1. Physical and chemical analyses of disinfected and raw lake water.

Parameters	Water samples			
	Raw water	ClO_2-treated	NaClO-treated	PAA-treated
$UV_{254\ nm}$ (abs/cm)	0.074	0.072	0.083	0.090
$DUV_{254\ nm}$ (abs/cm)	0.069	0.048	0.052	0.056
TOC (mg/L)	7.8	7.6	7.6	9.2
AOX (µg/L)	nd	22	166	16

nd = not detected.

PAA-treated waters showed higher values. Some organic compounds commonly found in surface water, such as lignin, tannin, humic and fulvic substances and various aromatic compounds, strongly absorb UV radiation; UV absorption is a useful surrogate measure of selected organic constituents in waters because a strong correlation may exist between UV absorption and organic carbon content and precursors of trihalomethanes and other disinfection by-products. The highest values of UV absorbance were found in NaClO and PAA disinfected waters suggesting these disinfectants induced DBPs. DUV values, *i.e*. measured in filtered water, were lower than UV ones in all samples, particularly in the treated waters suggesting the presence of compounds on particles in water that can absorbe UV. Analyses of detectable AOX concentrations were carried out only in disinfected samples. AOX is a measurement used to estimate the total quantity of dissolved halogenated organic material in a water sample. The presence of halogenated organic molecules is indicative of disinfection by-products. AOX concentration was significantly increased in NaClO-disinfected water (166 µg/L), whereas low values were found in ClO_2- and PAA-treated water (22 µg/L and 16 µg/L, respectively).

3.2. *Allium cepa* Test

The results of the *Allium cepa* root anaphase aberration test are set out in **Table 2** and in **Figure 1**.

The mutagenic potential was evaluated in proliferating cells in the meristematic tissue of Allium roots, analyzing both the chromosomal damage and the mitotic disturbance (*i.e*., clastogenic and aneugenic effects, respectively) induced in mitotic cells, through the chromosomal aberration test in anaphase, which are the first manifestation of an induced mutagenic damage and represent the "generating events" that give rise to micronucleus formation.

No data were obtained after long-term exposure at 10˚C because low temperature negatively influenced root growth. On the contrary, data obtained after the short-term exposure at the same temperature did not significantly differ from the raw water and negative control.

Table 2. Percent frequencies of anaphase aberrations (AA) and of mitotic index (MI) in *Allium cepa* root cells exposed to samples of raw and treated lake water at different temperatures for 24 or 72 hours; 800 anaphases (40 anaphases, 20 roots) and 5000 cells per experimental point were analyzed for AA and MI, respectively.

Exposure temperature	Water samples	Exposure time			
		24 hours		72 hours	
		AA (%)	MI (%)	AA (%)	MI (%)
10˚C	Negative control	2.8	9.2	nd	nd
	Raw water	2.7	8.0	nd	nd
	ClO_2-treated	3.5	9.7	nd	nd
	NaClO-treated	2.6	8.7	nd	nd
	PAA-treated	2.1	9.3	nd	nd
20˚C	Negative control	1.6	6.5	2.1	9.1
	Raw water	2.5	7.7	2.5	8.4
	ClO_2-treated	3.6^{**}	7.7	$4.7°^{**}$	8.2
	NaClO-treated	3.5^{*}	7.4	$4.3°^{*}$	8.2
	PAA-treated	2.7	8.2	3.7^{*}	8.0
30˚C	Negative control	4.0	7.9	4.0	7.9
	Raw water	4.3	9.1	5.1	9.8
	ClO_2-treated	5.6	8.1	3.4	7.8
	NaClO-treated	3.5	9.5	5.4	8.3
	PAA-treated	5.6	8.9	1.7	8.9

nd = not detectable (necrotic tissue); °Statistically significant vs raw water according to χ^2 test ($p < 0.05$); *Statistically significant vs negative control according to χ^2 test ($p < 0.05$); **Statistically significant vs negative control according to χ^2 test ($p < 0.01$); Positive control: maleic hydrazide (10 mg/L) for 6 hours, AA 11.4 (%).

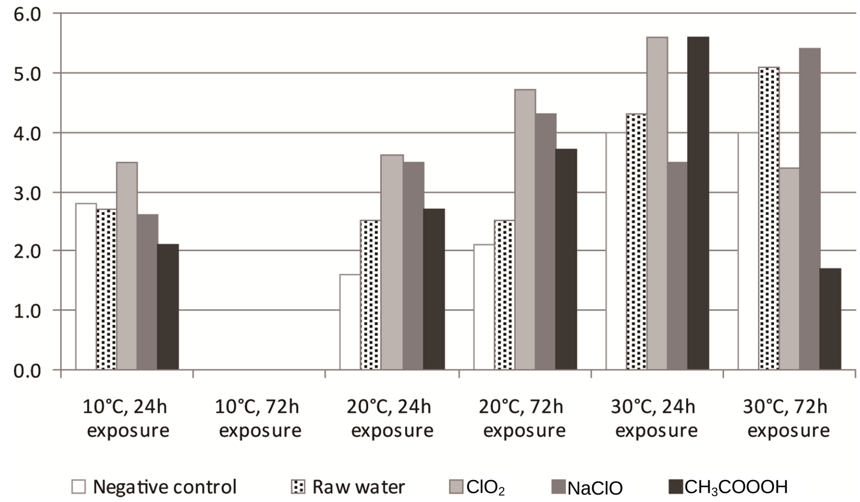

Figure 1. Percent frequencies of anaphase aberrations (AA) in *Allium cepa* root cells exposed to samples of raw and treated lake water at different temperatures for 24 or 72 hours.

ClO_2-, NaClO- and PAA-disinfected lake water showed mutagenic activity at 20˚C. This effect was detected for ClO_2- and NaClO-disinfected waters in comparison with the negative control treatment after exposures of 24 and 72 hours; such an effect was also shown in compareson to raw water after 72 hours of exposure. After 72 hours of exposure, PAA-disinfected lake water also showed a mutagenic effect in comparison to the negative control. The experiment carried out at 30˚C gave negative results for all the disinfectants and for all the exposure times.

3.3. *Vicia faba* Micronucleus Test

Unlike *Allium* roots, *Vicia faba* seedlings maintained at the temperature of 10˚C did not exhibit a considerable growth inhibition of secondary roots, even if a slight reduction of the mitotic indexes was observed in both control and treated root samples maintained at the low temperature for 72 hours.

The results of the micronucleus test in *Vicia faba* are set out in **Table 3** and visualized in **Figure 2**.

Significant increases of micronucleus frequencies appear to be induced by all disinfected-water under any experimental-exposure condition. Namely, NaClO- and ClO_2-disinfected lake waters showed a strong clastogenic/aneugenic activity at all the temperatures, whereas raw lake water showed no effect. This result is in agreement with the data from *in situ* exposure carried out in previous experiments at the disinfection treatment pilot plant [11]. In addition, the mutagenic effect increased with exposure time, mainly at 20˚C and 30˚C; a positive significant correlation between micronucleus frequentcies and temperature was also found after 72-hour exposure to NaClO- and ClO_2-disinfected lake waters (**Figure 3**).

4. Discussion

The present experimental design has been established taking into account the exposure conditions and positive response of the two plant test systems obtained from our previous *in situ* study, as well as their specific characteristics [11]. Namely, the temperatures were chosen so as to cover the whole range of those registered in the different seasons at which these plants had been exposed and responded, showing no growth inhibition of roots (data not shown). Different exposure protocols were established for the two plant systems: the 24 and 72 hours of continuous exposure were chosen for *Allium cepa*, on the basis of its relatively limited responsiveness observed after a short time exposure to disinfectants, especially if compared to that of *Vicia faba* which, on the contrary, showed consistent increases of micronucleus frequencies already after an exposure time of 6 hours. The short exposure times were long enough to hit the asynchronously proliferating cells in every phase of their cell cycle, so as to allow the possible induction of DNA damage (clastogenic mutagen) and/or impairment of mitotic process (aneugenic mutagen) in both test organisms. Also, the 72-hour fixation time was scheduled in order to allow a sufficiently large recovery time for the detection of possible S-dependent clastogenic effects, as well as to allow the occurrence of two or more mitotic rounds in order to detect micronuclei possibly deriving from mitotic disturbance.

Data from chemical analysis suggest the presence of DBP and their precursors in disinfected water, and in particular in NaClO-treated water where the AOX value is higher than water treated with other disinfectants.

The used bioindicators are sensitive to DBP but their response in terms of mutagenicity results to be modulated by the temperature of exposure.

Table 3. Mean frequency (±SE) of micronuclei/1000 cells in root tip cells of *Vicia faba* exposed to the same samples of raw and treated lake water at different temperatures for 6 hours (plus 66 h recovery) or 72 hours, evaluated in 10 root tips, 5000 cells per tip.

Exposure temperature	Disinfection treatments	MCN (‰) Exposure time 6 hours	72 hours
10˚C	Negative control		0.22 ± 0.06
	Raw water	0.28 ± 0.05	0.26 ± 0.08
	ClO_2-treated	0.42 ± 0.09	0.34 ± 0.09
	NaClO-treated	0.58 ± 0.11°**	0.66 ± 0.09°°**
	PAA-treated	0.50 ± 0.16	0.34 ± 0.07
20˚C	Negative control		0.16 ± 0.05
	Raw water	0.28 ± 0.10	0.44 ± 0.11
	ClO_2-treated	0.26 ± 0.08	0.62 ± 0.14**^
	NaClO-treated	0.40 ± 0.09	1.22 ± 0.41***^
	PAA-treated	0.32 ± 0.06	1.14 ± 0.36**^
30˚C	Negative control		0.24 ± 0.08
	Raw water	0.50 ± 0.13	0.46 ± 0.12
	ClO_2-treated	0.56 ± 0.14	1.48 ± 0.27°°***^^
	NaClO-treated	0.76 ± 0.11**	1.90 ± 0.37°°***^^
	PAA-treated	0.46 ± 0.09	0.72 ± 0.16*

°Statistically significant vs raw water, with the same exposure time, according to Mann-Whitney Test ($p < 0.05$); °°Statistically significant vs raw water, with the same exposure time, according to Mann-Whitney Test ($p < 0.01$); *Statistically significant vs negative control according to Mann-Whitney Test ($p < 0.05$); **Statistically significant vs negative control according to Mann-Whitney Test ($p < 0.01$); ***Statistically significant vs negative control according to Mann-Whitney Test ($p < 0.001$); ^Statistically significant vs the same treatment, 6-hour exposure, according to Mann-Whitney Test ($p < 0.05$); ^^Statistically significant vs the same treatment, 6-hour exposure, according to Mann-Whitney Test ($p < 0.01$); Positive control: maleic hydrazide (10 mg/L) for 6 hours, 31.7 ± 11.5 (‰).

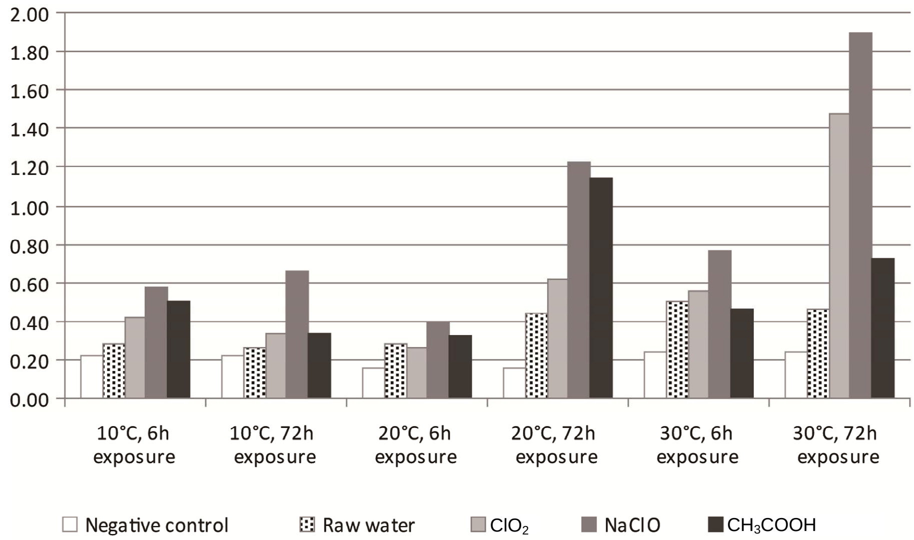

Figure 2. Graphical representation of mean micronucleus (MCN) frequency (‰) data from *Vicia faba* root tip cells (see Table 3) exposed to disinfected lake-water at different temperatures (10˚C, 20˚C and 30˚C) for either 6 h plus a 66 h recovery time (6 h exposure) or 72 h (72 h exposure).

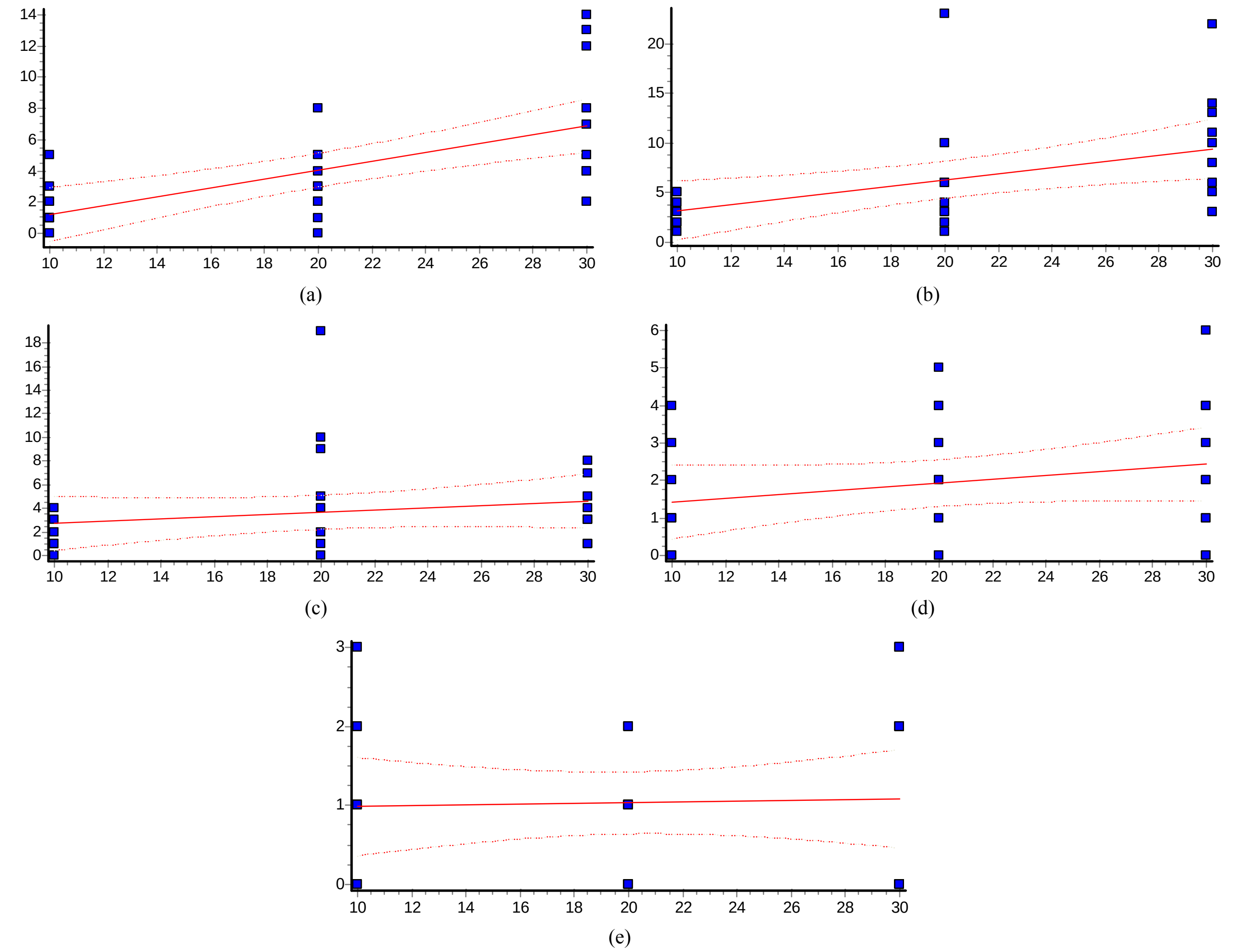

Figure 3. Regression analysis curves between temperature values (˚C, abscissa) and micronucleus frequencies (ordinate) obtained from 72 h exposure of secondary root tips of *Vicia faba* to samples of disinfected lake water (a)-(c), raw water (d) and negative control water (e). (a) ClO_2; (b) NaClO; (c) PAA; (d) Lake raw water; (e) Hoagland's solution.

The results of this study indicate that temperature may play an important role in mitotic cell cycle progression, at least in *Allium cepa* and *Vicia faba* root cells, and in modulating the expression of clastogenic/aneugenic damage.

Despite the positive results obtained in *Allium cepa* from the field study carried out at the Trasimeno water-treatment plant, where roots not only survived the 72-hour exposure at temperature as low as 10˚C, but also gave the most powerful mutagenicity data, in the present experiment the same temperature showed to be not permissive for root growth.

Indeed, mutagenic effects were only observed at 20˚C, which shows to be the optimal temperature for cell proliferation of this plant roots. In this organism, mitotic cycle duration has been determined by several authors, who showed that the optimal temperature for root growth is 20˚C ± 2˚C; under such conditions mitotic cycle duration is about 24 hours [42]. The observed variations reported in the present study could largely be explained by the different exposure temperatures, which appear to influence root growth [45].

Treatments of *Allium cepa* with NaClO- and ClO_2-disinfected lake waters performed at 20˚C induced a clear aneugenic/clastogenic effect, detectable at both exposure times. This effect seemed to be more pronounced at the long-term exposure (72 hours), at which PAA-disinfected lake water also appears to exert a mutagenic effect. Short-term exposure in the *Allium cepa* test gave generally negative results for all disinfectants, although NaClO- and ClO_2-disinfected lake water produced mutagenic effects compared to the negative control. On the other hand, when the exposure time was increased (72 hours, about two mitotic cycles), mutagenic effects were detected for the NaClO- and ClO_2-disinfected lake water compared to raw water.

Low temperature (10˚C) is likely to cause a delay in cell proliferation in *Allium cepa* roots, which may explain the long-term exposure inhibition of root germination and growth. At the same temperature, short-term exposure produced no genotoxic effects since mitosis rate was reduced, and possibly aberration frequency as well.

The highest temperature (30˚C) did not cause any remarkable mutagenic effects, despite the increase in frequency in some samples. The negative control also gave a high frequency of aberrations at the highest temperature. High temperature is likely to impair root proliferateing cells, inducing a sort of stress that may be the cause of most mutagenic damage. In this study the *Allium cepa* test was less sensitive to mutagenic effects at high temperature conditions, which is in agreement with the data reported in a previous study on *in situ* exposure in June, which corresponded to a very warm period [11]. In *Allium cepa* the highest temperature did not seem to influence the mitotic index, and hence the cell cycle was not influenced. This means that cell division rate was not changed, but root length was on average shorter than the negative control. Moreover, other signs of toxicity were observed in Allium roots (form and consistency of roots) and the toxicity may have concealed the genotoxic effects.

In *Vicia faba* temperature also plays an important role in modulating the expression of clastogenic/aneugenic damage. Low temperature may slow down the progresssive reduction in micronucleus frequency after short-term treatments, which is expected from their dilution/destruction [32,46], as a consequence of the lengthening of the cell cycle. This could explain why 72-hour exposure produced higher clastogenic/aneugenic effects than 6-hour exposure at a higher temperature (**Figure 2**). Increasing clastogenic/aneugenic activity of NaClO- and ClO_2-disinfected lake waters at increasing temperatures is suggested by the increase in micronucleus frequencies after 72 hours of exposure. Equilibrium value of micronucleus frequency is reached after long-term treatments (72 hours) for the rise of new micronuclei and the dilution/destruction of the old ones [32,46]. An increase in the equilibrium frequency of micronuclei therefore suggests a higher rate of new micronucleus formation. A comparison of these findings with those from *in situ* exposure [11] confirmed that NaClO- and ClO_2-disinfected lake waters have stronger clastogenic/aneugenic effects than PAA-disinfected waters; the lack of a difference in micronucleus frequency after 6-hour and 72-hour exposure in the cold season is also confirmed. The hypothesis that low temperatures slow down micronucleus dilution/destruction with time relenting cell cycle progresssion is therefore supported.

The importance of cell cycle duration in evaluating genotoxic damage has been seen for other genotoxicity plant tests as well. In the *Tradescantia*/micronuclei test, exposure to some toxic compounds or to overdoses causes a delay in the meiotic cycle, and the induced damage may be not recognized using the standard time of test protocol [47].

The temperature affecting cell cycle duration may influence sensitivity and the possibility of revealing low levels of environmental genotoxins. The results of this study show that temperature plays an important role in mitotic cell cycle progression in *Allium cepa* and *Vicia faba* root tips. *Vicia faba* micronucleus test proved to be more sensitive than *Allium cepa* chromosomal aberration test with regard to temperature influence on genotoxic damage expression. The different temperatures at which the plant tests were carried out could have impaired the plausibility of the experimental data obtained.

In conclusion, temperature is an important variable to be taken into account when planning *in situ* exposure of plants for mutagenicity tests. This poses the question of selecting appropriate test organisms, taking into account their tolerance and viability under relatively wide thermal ranges among environmental variables.

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Journal of Water Resource and Protection, 2012, 4, 648-656
doi:10.4236/jwarp.2012.48075 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Hydrochemistry as Indicator to Select the Suitable Locations for Water Storage in Tharthar Valley, Al-Jazira Area, Iraq

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Received May 12, 2012; revised June 16, 2012; accepted June 29, 2012

ABSTRACT

Four locations were chosen according to geomorphologic and engineering criterion to store the water on the midstream of Tharthar valley, water samples were collected from the four locations to evaluate the hydrochemical properties as indicator to select the more suitable location, these locations are Hatra, Abu-Hamam, Tlol Al-Baj and Al-Sukkariah from the north to the south respectively. Also, the groundwater samples were collected from two shallow wells on the banks. The samples were analyzed to determine the concentrations of most common anions and cations in the water Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, CO_3^{2-}, HCO_3^{-}, Cl^{-}, SO_4^{2+}. Also, pH, EC and TDS were measured. The results reflect high variations in concentrations of the soluble materials, the concentrations of these components are highly increased in locations of Tlol Al-Baj and Al-Sukkariah in comparison with the locations of Hatra and Abu-Hamam. The variation in geology of the area along the valley was represented a main role on the quality of water. These results can help to select the suitable locations of small dam (dams) to store the water in the valley and prevent the problem of salinity. According to the results, the northern part of midstream (north of Abu-Hamam) is suitable for water storage and the dam construction. While the locations of the downstream enriched by local sources of salts.

Keywords: Tharthar; Dam; Hydrochemistry; Salinity

1. Introduction

Tharthar Valley originated from Sinjar Mountain (northwest boundary of Iraq) and flow with gentle slope to the south in the area between Tigris and Euphrates rivers, and directed slowly to the west to arrive the depression of Tharthar Lake (125 km North Baghdad).

According to the GIS data, the area of Al-Tharthar basin is about 23,254 km^2, perimeter of 709 km, the maximum length is more than 268 km, and the average width of basin approximately 128 km. The maximum elevation in watershed of the valley is more than 400 m.a.s.l and the elevation of bottom of the valley in the upstream is about 225 m.a.s.l., but it decreased to 120 m.a.s.l. in the midstream (Hatra City) and to 55 m.a.s.l. in the north margin of Tharthar Lake "**Figure 1**".

The annual precipitation in the area is about 150 mm in the south of the basin near Tikrit city and increased to (500 mm) to the North near Sinjar. The main percentage of rainfall occurred from December to the end of March. About 40% of the catchment area with annual average rainfall more than 250 mm, and 60% of the catchment area with annual rainfall less than 250 mm. The area located in the arid to semi-arid zone, it is characterized by high evaporation, especially during the long hot summer [1].

Geologically, the site of study located in the south part of the unfolded zone of Iraq, while the upland of the valley is located in the low folded zone, and mostly this depression and the main channel of the valley located along subsurface regional fault [2].

The area covered by Miocene sediments which is represented by the succession of carbonate, evaporate, and claystone alterations, which represent a Miocene deposit (Fatha'a) which is exposed in many areas along the banks of the valley. Also, Pliocene (Injana Fm.) rocks are represented by fluvial clastic sandstone and claystone. Most of the area covered by Quaternary sediments is represented by moving sand dunes, gypsiferous and gypycreate soils originated from the older formations of

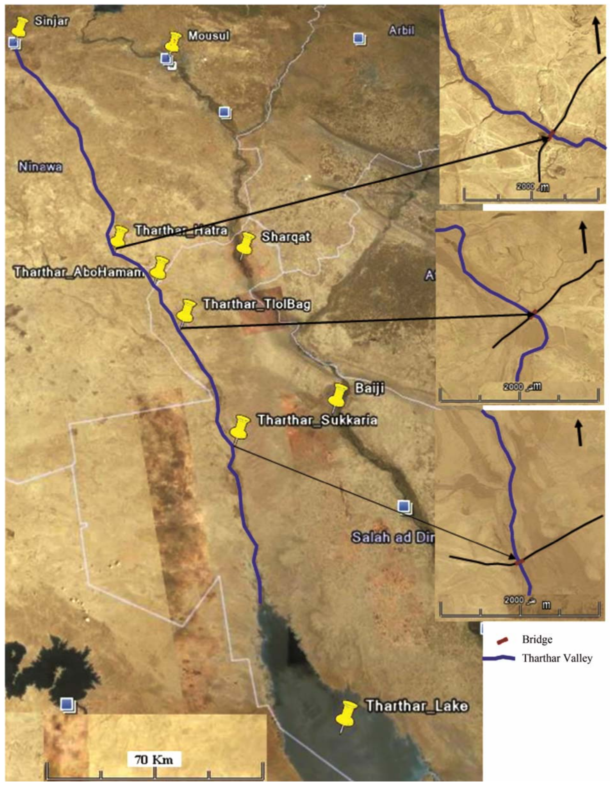

Figure 1. Location map and sampling stations on Tharthar Valley.

Miocene and Pliocene [2].

[3], studied the hydoengineering properties of the basin to the north of Hatra City using the satellite images and aerial photos, the study determined five geomorphological zones in the area, and drew the map of land use. And he suggested three locations to construct dams on the valley.

[4], investigated the Climatological, hydrogeological and geomorphological properties of six sub-basins in the northern part of the Tharthar Valley, and predicted the discharge of the valleys in the area.

This project aims to evaluate the surface and groundwater quality and the specific characteristics of water within Tharthar valley, and the hydromorphometric aspects.

Also to interpret the groundwater relationship with surface water, and discuss the variation of salinity, ionic concentrations and determination of hypothetical salts and saturation indices of these salts within the studied area along the valley.

The results are the main factors in determination of local dams and reservoirs on the studied sites.

2. Materials and Methods

The studied area locates between UTM coordinates 190,000, 340,000 eas, 3,800,000, 4,050,000 north "**Figure 1**".

Four locations have been selected for sampling of surface water and two shallow wells for groundwater, "**Figure 2**". The laboratory procedures were carried out according to the procedures of (ASTM, 1984) in [5].

The analyses of the most common cations Ca^{2+}, Mg^{2+}, Na^+, K^+, anions CO_3^{2-}, HCO_3^-, Cl^-, SO_4^{2+}, and another parameters pH, EC and TDS were carried out in the laboratories of ICARDA.

pH and EC values are determined directly by pH-EC meter for water samples. The concentrations of soluble sodium and potassium in water were determined by using the flame photometer, while the soluble calcium and magnesium were determined by titration with EDTA.

The soluble chloride was determined by titration with silver nitrate $AgNO_3$, soluble carbonate and bicarbonate were determined by titration with H_2SO_4 (for pH 8.3 - 4.5), soluble sulfate determined by precipitation method with barium chloride (ASTM, 1984) in [5].

3. Results and Discussions

Water system contains salts, kinds and concentrations of the salts depend on the interaction between the water chemistry and the rocks or soil of the stream channels or groundwater aquifers. The degree of dissolution and precipitation of soil or rocks materials is controlled by ground or surface water movement and system conditions [6]. The human activities may affect the concentrations of materials in the water; such as irrigation, fertilization, contribution of wastewater and urban uses [7].

3.1. The Results

The results of water chemistry are indicated on "**Table 1**". The relative errors of analyses were calculated by the formula of charge balance [8]. The samples analyses are acceptable according to this formula which assumes the relative error less than 5%.

3.2. The Discussion

3.2.1. Variation of Hydrochemical Parameters

From the results of the concentrations of the soluble ions "**Figure 2**", reveal the similarity in the concentrations of all ions in the first and second stations, and SO_4^{2-} is the most abundance ion in these stations, the source of this ion mainly is the gypsum and gypcrete in the rocks and soil in the area, also these concentrations of ions are similar in the groundwater of shallow aquifer near the banks of valley in Abo-Hamam and Al-Ib areas that reveal the similarity of host rocks and soil. In the direction of flow to the south, the concentrations of ions are increased suddenly in the area of Tlol-AlBaj, especially Na^+ and Cl^- concentrations, the sudden increasing is related to a local source of these ions, the concentrations of

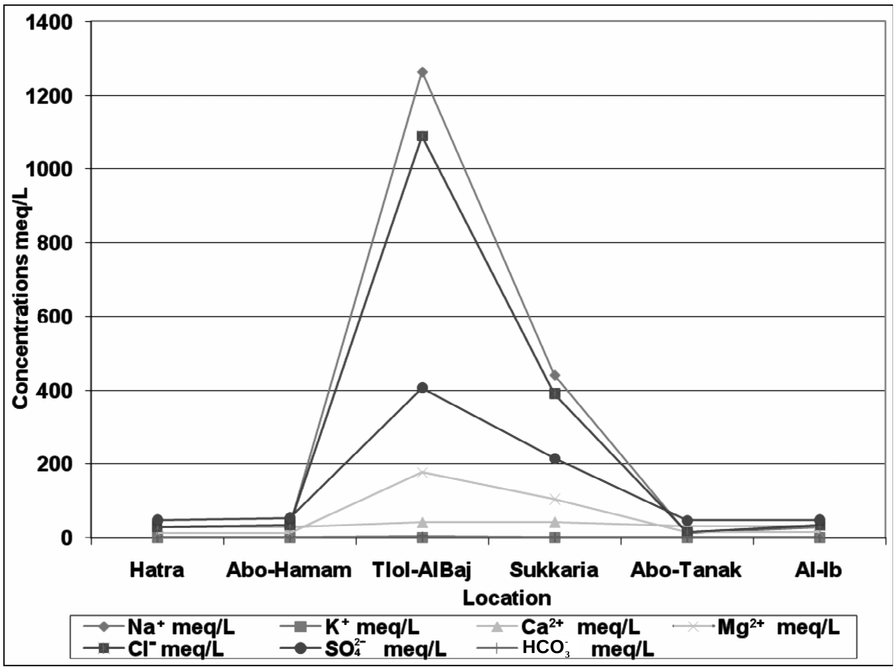

Figure 2. The variation of the concentrations of different ions along the midstream of Tharthar Valley.

Table 1. The results of water chemistry including the major cations, anions, pH, EC, secondary elements (NO_3^{2-}, NH_4), calculated chemical parameters (TDS by total ions and TDS by EC, sodium adsorption ratio SAR and total hardness TH).

Sample No.	Na^+ meq/L	K^+ meq/L	Ca^{2+} meq/L	Mg^{2+} meq/L	Cations meq/L	Cl^- meq/L	CO_4^{2-} meq/L	HCO_3^- meq/L	SO_4^{2-} meq/L	Anion meq/L
Hatra	27.8	0.11	30.56	13.2	71.67	30.63	0	2.6	48.48	81.71
Abo-Hamam	32.4	0.16	30.79	14.5	77.85	36.04	0	2.1	53.15	91.29
Tlol-AlBag	1264.6	0.59	41.57	177.4	1484.16	1090.09	0	4.2	405.73	1500.02
Sukkaria	442.1	0.88	42.47	105.9	591.35	390.99	0	2.5	214.87	608.36
Abo-Tanak	12.7	0.26	33.48	15.6	62.04	15.77	0	1.65	47.28	64.70
Al-Ib	30.1	0.26	33.71	14.9	78.97	34.23	0	1.85	48.14	84.22

Sample No.	pH	EC mS/cm	TDS = EC ∗ 640 ppm	Calculated TDS mg/L	SAR	TH
Hatra	6.9	5.25	3360	4998.69	6	2189.73
Abo-Hamam	7.2	5.66	3622.4	5515.08	7	2266.28
Tlol-AlBag	7.2	83.6	53,504	90,487.54	121	10,956.97
Sukkaria	7.1	39.8	25,472	36,679.36	51	7424.26
Abo-Tanak	7.6	4.39	2809.6	4149.08	3	2455.93
Al-Ib	7.8	5.56	3558.4	5255.01	6	2432.42

EC = Electrical Conductivity; TDS = Total Dissolved Solids; SAR = Sodium Adsorption ratio; TH = Total Hardness.

these ions in Sukkaria area less than Tlol-AlBag, which means that Tlol-AlBag area represents the main source of Na^+ and Cl^-, and water of Sukkaria effected by the salts during the pass of water in the peak of flooding, while in Sukkaria the water still in contact with the salts source for along period, NaCl rich water may be comes as leakage from the older evaporates rocks such as Fatha'a Fm. (Miocene) and exposed in the area. The water within this formation saturated with these ions because of the dissolution of rock salt layers. The change in concentrations of ions is also reflected in other parameters such EC and CTDS (calculated TDS from the summation of concentrations of major ions) “**Figure 3**”.

3.2.2. Saturation Indices

The change in the ionic content in flow water along the valley is considered as an important factor for determining the optimum location of small dams and reservoirs to avoid the source of salinity, which needs additional studies for these purposes.

The concentrations of the soluble ions are required to calculate the ionic strength (I) of the solution for a mixture of electrolytes, the calculation of the ionic strength depending on equation of [6].

$$I = 1/2 \sum m_i Z_i^2 \tag{1}$$

where (I) is the ionic strength, (m_i) is the molality of i^{th} ion, (Z_i) is the charge of i^{th} ion. The values of ionic strength for all samples listed in “**Table 2**”. The activity coefficient of an individual ion is determined by Debye-Hückel Equation [9].

$$-\log \gamma_i = \frac{A Z_i^2 \sqrt{I}}{I + a_i B \sqrt{I}} = V \tag{2}$$

γ_i is the activity coefficient of ionic species i.

Z_i is the charge of ionic species i.

I is the ionic strength of the solution.

A temperature coefficient equals to 0.5085 at 25˚C.

B depends on temperature and equals to 0.3281 at 25˚C.

a_i is the effective diameter of the ion.

The program WATEQ4F-2.62 under WINDOWS is used to calculate the activity coefficient and the chemical activity of the major ions K^+, Na^+, Ca^{2+}, Mg^{2+}, HCO_3^-, Cl^-, and SO_4^{2-}, the ionic activity product (Kiap), which is the product of the measured activities, also calculated to test the saturation (Fetter, 1980), K_{iap} calculated for anhydrite $CaSO_4$, aragonite $CaCO_3$, brucite $Mg(OH)_2$, calcite $CaCO_3$, dolomite (Ca, Mg)CO_3, epsomite $MgSO_4 \cdot 7H_2O$, gypsum $CaSO_4 \cdot 2H_2O$, magnesite $MgCO_3$, and Hallite NaCl by the computer program WATEQ4F to determine the degree of saturation by the comparison of the values of K_{iap} for a mineral in natural water with the theoretical value of the solubility product of mineral K_{sp} [5]. The Saturation Index (SI) of the mineral is defined as the state of saturation, which is represented by the equation of [10].

$$SI = \log \text{Ion Activity Product} / \text{Solubility Product}$$

The negative value of SI reflects unsaturated conditions regarding to the mineral phase and the mineral may be actively dissolved, while the positive values reflect saturation or supersaturating conditions and the mineral is precipitated, the zero value indicates equilibrium condition [11]. The saturation indices were calculated for the above minerals, the outputs were listed in “**Table 2**”. The table and “**Figure 4**” refer to unsaturated conditions in the location of Hatra for all mineral phases, while the calcite and gypsum access the saturation limit in Abo-Hamam, these minerals have low solubility and reach the saturation firstly. The halite is very far to saturation in first two locations, but SI is suddenly increased to be near the saturation in Tlol-AlBag because of the effect of salts source in this area, the saturation in Sukkaria area less than that of Tlol-AlBag for all mineral phases, that indicates the local source of salty water in Tlol-AlBag.

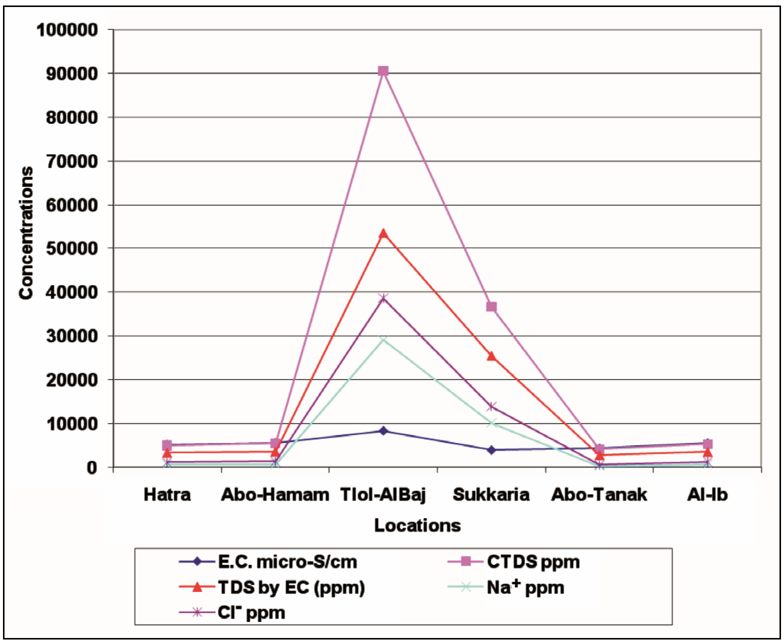

Figure 3. Showing the anomalous parameters in the locations of Tlol-AlBaj and Sukkaria.

Table 2. Ionic Index and Saturation Index of mineral phases, reflect the saturation of surface and groundwater concerning to different mineral phases.

Sample No.	Ionic Strength		Saturation Index of Mineral Phases								
	Total	Effective	Anhydrite	Aragonite	Brucite	Calcite	Dolomite	Epsomite	Gypsum	Halite	Magnesite
Hatra	0.12351	0.09167	–0.234	–0.199	–5.805	–0.055	–0.908	–2.838	–0.015	–4.877	–0.884
Abo-Hamam	0.13465	0.10044	–0.217	–0.010	–5.181	0.134	–0.493	–2.784	0.001	–4.748	–0.658
Tlol-AlBag	**1.98417**	**1.64817**	**0.091**	**0.121**	**–4.190**	**0.264**	**0.783**	**–1.605**	**0.268**	**–1.742**	**0.488**
Sukkaria	0.81136	0.64943	–0.023	–0.178	–4.672	–0.035	–0.078	–1.885	0.182	–2.688	–0.074
Abo-Tanak	0.11248	0.07728	–0.187	0.351	–4.316	0.495	0.224	–2.758	0.032	–5.494	–0.302
Al-Ib	0.13113	0.09752	–0.213	0.574	–3.947	0.718	0.650	–2.805	0.005	–4.798	–0.099

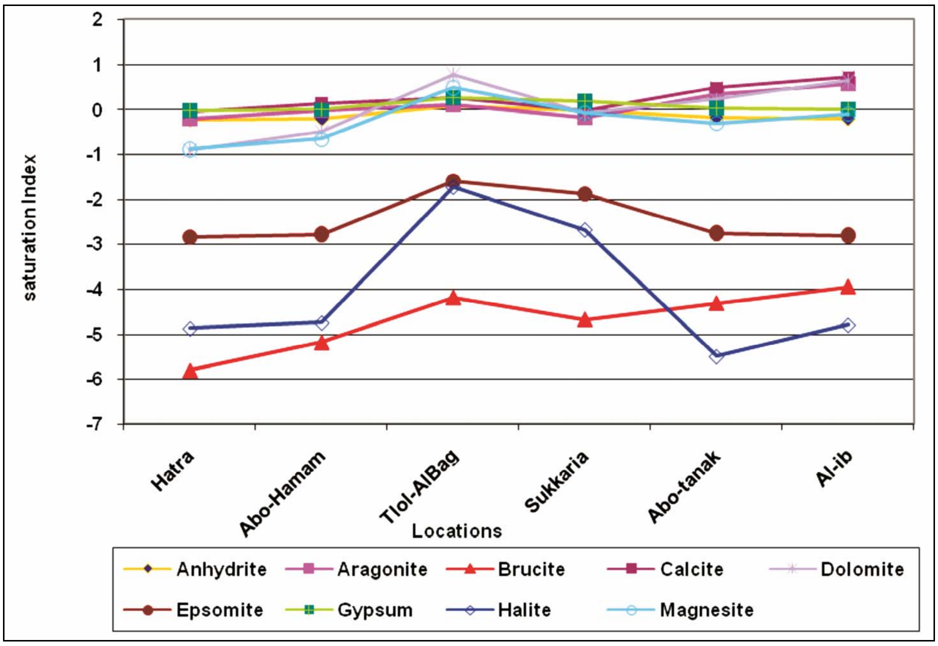

Figure 4. Saturation Index of mineral phases, reflect the saturation conditions of surface and groundwater regarding to mineral phases.

The SI for the groundwater samples in Abo-Tanak and Al-Ib near the banks of the valley similar to the behavior of the first two locations of surface water in the valley, which indicates the similar conditions of host rocks and soil.

3.2.3. Hypothetical Salts

The average of common hypothetical salts combination was calculated, NaCl and $CaSO_4$ are the main two common soluble salts "**Table 3**". "**Figure 5**" shows the behavior of hypothetical salts combination along the stream of valley and the groundwater, the salt combinations seem similar in the locations of Hatra, Abo-Hamam and in the ground water well of Al-Ib, and show different percentages in Tlol-AlBag and Sukkaria. The percentage of NaCl is increased suddenly in Tlol-AlBag to reach more than 70%, while $CaSO_4$ is decreased because it became out of the solution, and the gypsum reach to saturation limits. The decrease of NaCl in Sukkaria in comparison with Hatra indicates that the local source of NaCl, the $CaSO_4$ is increased suddenly in the groundwater of Abo-Tanak because it is from the Karistified aquifer.

4. Usability of Tharthar Water for Irrigation

The pH of the samples is ranged between 6.9 and 7.8 "**Table 1**", that means all samples in the tolerable range of 6.5 - 8.4 which is suggested by [12], "**Table 4**", for irrigation.

The electrical conductivity (EC) (mS/cm) of the water is a useful tool to evaluate the usability of water for irrigation [13]. The limits of EC shown in "**Table 4**", the values of EC in the present work more than 3 mS/cm in all

Table 3. Average of hypothetical salt combination epm%, of surface water in Tharthar Valley and groundwater of shallow aquifer on the banks.

Sample No.	$Ca(HCO_3)_2$	$CaSO_4$	$MgSO_4$	Na_2SO_4	$MgCl_2$	NaCl	KCl
Hatra	3.18	39.46	18.42	1.45	0	37.34	0.15
Abo-Hamam	2.30	37.25	18.63	2.34	0	39.28	0.21
Tlol-AlBag	0.28	2.52	11.95	12.58	0	72.63	0.04
Sukkaria	0.41	6.77	17.91	10.64	0	64.12	0.15
Abo-Tanak	2.55	51.42	21.66	0	3.49	20.47	0.42
Al-Ib	2.20	40.49	16.67	0	2.2	38.12	0.33

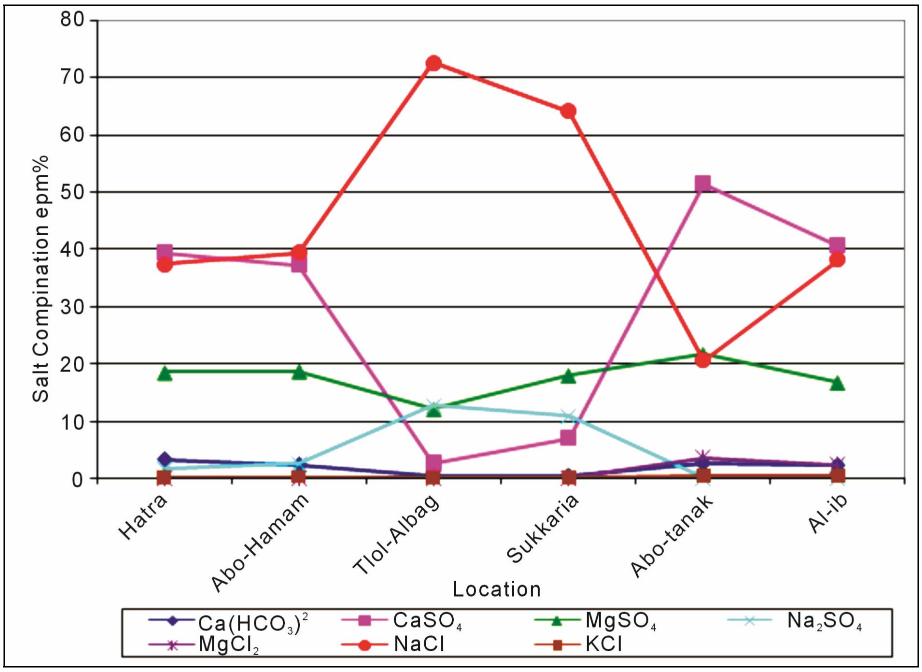

Figure 5. Average of hypothetical salt combination epm%, of surface water in Tharthar Valley and groundwater of shallow aquifer on the banks.

the studied locations "**Table 1**", and the water of the locations of Hatra and Abo-Hamam and groundwater are permissible for irrigation with modern irrigation systems such as drip and sprinkler, while the water in the locations of Tlol-AlBag and Sukkaria not permissible for irrigation.

The total dissolved solids (TDS) in water are represented by the weight of residue left when a water sample has been evaporated to dryness. It is measured by determining the actual salt content in parts per million (ppm) or (mg/L). A physiological drought condition can result from excess salts accumulating in the soil by increasing the osmotic pressure of the soil solution. Plants can wilt due to insufficient water absorption by the roots compared to the amount lost from transpiration, even though the soil may have plenty of moisture. (TDS = EC × 640). The ISI standard for dissolved solid is up to 500 mg/L and the maximum permissible quantity is 2000 mg/L "**Table 4**", [12,14,15]. The values of TDS which calculated by the summation of major soluble ions, or calculated by EC in the present work more than 2000, "**Table 1**", and that may due to the collection of sample during the draught seasons.

The calcium (Ca^{2+}) is generally found in all natural waters. When adequately supplied with exchangeable calcium, soils are friable and usually allow water to drain easily. This is why calcium in the form of gypsum is commonly applied to improve the physical properties of tight soils. Sodium will be leached from the root zone when the Ca^{2+} replaces the Na^{+} on the soil colloid. Irrigation water that contains ample calcium is most desirable, the concentrations of Ca^{2+} "**Table 1**" are more than the desired range 40 - 120 mg/l in all the studied samples.

The magnesium (Mg^{2+}) is also found in most natural

Table 4. Guidelines for irrigation water quality established by (FAO).

	Intensity of Problem[1]		
Water Constituent	No Problem	Moderate	**Severe**
Salinity EC (decisiemens meter^{-1})	<0.7	0.7 - 3.0	>3.0
Salinity TDS mg/L or ppm	<450	450-2000	>2000
Permeability (rate of infiltration affected)			
Salinity (decisiemens meter^{-1})	>0.5	0.5 - 0.2	<0.2
Adjusted SAR; soils are:			
Dominantly montmorillonite, smectites	<6	6 - 9	>9
Dominantly illite-vermiculite	<8	8 - 16	>16
Dominantly kaolinite-sesquioxides	<16	16 - 24	>24
Specific Ion Toxicity			
Sodium (as adjusted SAR) (sprinkler)	<3	3 - 9	>9
Chloride (mmol/L) (sprinkler)	<3	>3	>10
Boron (mmol/L) as B	<0.70	0.70 - 30	>3.0
HCO_3^- (mmol/L) as damage by overhead sprinkler	<1.5		>8.5
pH	6.5 - 8.4	1.5 - 8.5	0 - 5, 9.5+

Source: modified from R. S. Ayers and D. W. Westcott, "water quality for agriculture", irrigation and drainage paper, 29, FAO, Rome, 1976; rev. 1986. [1]Based on the assumptions that the soils are sandy loam to clay loams, have good drainage, are in arid to semiarid climates, that irrigation is sprinkler or surface, that root depths are normal for soil, and that the guidelines are only approximate; [2]Assumes molecular weight = mole weight (one charge) because it is slightly ionized or nonionzed.

waters. Together with calcium, Mg may be used to establish the relationship to total salinity and to estimate the sodium hazard. The concentrations of Mg^{2+} "**Table 1**" are more than the desired range 6 - 24 mg/l in all the studied samples.

Ca^{2+} and Mg^{2+} are caused by far the greatest portion of the hardness occurring in natural waters. All the metallic cations beside the alkali metals caused hardness. Hardness of the water is objectionable from viewpoint of water use.

The sum of calcium and magnesium compounds results in the total hardness are measured in milligram calcium carbonate per liter. In order to determine the total hardness, the weight percentage of the magnesium compound is converted into the equivalent $CaCO_3$.

Total Hardness = 2.497 $*$ Ca^{2+} mg/l + 4.115 $*$ Mg^{2+} mg/l [16].

The TH of all the studied samples "**Table 1**" are highest than the permissible limit which is 6.0 meq/L (300 mg/L) that prescribe by (ICMR 1975) in [15].

The sodium (Na^+) is often found in natural waters due to its high solubility. When it linked to chloride (Cl^-) and sulfide $\left(SO_4^{2-}\right)$, sodium is often associated with salinity problems. High concentrations in the soil can adversely affect turf grasses. Poor soil physical properties for plant growth will result as a consequence of continued use of water with high sodium levels. The concentrations of Na^+ more than the permissible limit of 50 ppm (9 meq/L) in irrigation water prescribed by BIS (1983) in [13]. And that may due to collection of samples during the dry condition.

SAR sodium adsorption ratio is an important parameter for determination of suitability of irrigation water. This index quantifies the proportion of sodium (Na^+) to calcium (Ca^{2+}) and magnesium (Mg^{2+}) ions [15].

The SAR is also an index of sodium permeability hazard as water moves through the soil. The main problem with a high sodium concentration is its effects on the physical properties of soil. This breakdown disperses the soil clay and causes the soil to become hard and compact when dry and reduces the rate of water penetration when wet. A breakdown in the physical structure of the soil can occur with continued use of water with a high SAR value. The effects of high SAR on the infiltration of irrigation water are dependent on the EC of the water. The permissible limit of SAR < 6 no problem, 6 - 9 moderate and >9 severe [12], while [17] classified irrigation water with SAR values less than 10 as (excellent). The sodium adsorption ratio (SAR) values of water samples were calculated by using Richard equation [12]:

SAR=(Na^{+} meq/l)/√ [(Ca^{2+} meq/l)+(Mg^{2+} meq/l)/2] (3)

The calculated values of SAR in the study area "**Table 1**" are in the desired limit for irrigation purposes, except the locations of Tlol-AlBag and Sukkaria, and that may due to local source of NaCl in these areas.

Water alkalinity, simply stated, is a measure of the water's capability to neutralize added acids. Related to pH, alkalinity establishes the buffering capacity of water. The major chemicals that contribute to the alkalinity of water include dissolved carbonates, bicarbonates and hydroxides. High alkalinity can cause an increase in the pH of the soil (reducing micronutrient availability), the precipitation of nutrients in concentrated fertilizer solutions, and reduce the efficacy of pesticides and growth regulators. The desired range is 1 - 100 ppm.

An alkalizing effect of carbonates $\left(CO_3^{2-}\right)$ results when combined with calcium and/or magnesium. This effect is much stronger when it occurs in the presence of the sodium cation, the permissible range <50 ppm. The concentrations of CO_3^{2-} are nil in the samples "**Table 1**" and within the accepted range for Irrigation.

Bicarbonates $\left(HCO_3^{-}\right)$ are also salts of carbonic acid and are common in natural waters. As soil moisture is reduced, calcium and magnesium bicarbonates can separate calcium from the clay colloid, leaving sodium to take its place. An increase of SAR in the soil solution will result. The overuse of high bicarbonate irrigation water can contribute to a soil dominant in sodium, with a resulting reduction in water infiltration rates and soil gas exchange. The permissible range is <120 mg/L, [13].

The concentrations of HCO_3^{-} "**Table 1**" are within the desirable limit in the surface water and groundwater.

Chloride is an anion that is commonly found in irrigation water. Chlorides contribute to the total salt (salinity) content of soils, Chloride salts in excess of 100 mg/1 give salty taste to water, Unusual Concentration may indicate pollution by organic waste [15]. It is necessary for plant growth in small amounts, while high concentrations will inhibit plant growth or be toxic to some plants. Irrigation water high in chloride reduces phosphorus availability to plants, high chloride content in ground-water can be attribute to lack of under ground drainage system and bad maintenance of environment around the sources, the permissible limit is 10.0 meq/L (355 mg/L) [13].

The concentrations of Cl^{-} "**Table 1**" are more than the desirable limit in the surface water, and within the permissible limit in the groundwater in the area.

Sulfate $\left(SO_4^{2-}\right)$ is relatively common in water and has no major impact on the soil other than contributing to the total salt content. Irrigation water high in sulfate ions reduces phosphorus availability to plants. The desired range is <400 ppm, and >400 ppm will acidify the soil depending on the standard limits of BIS, 1999 in [15]. The concentrations of SO_4^{2-} are more the desired limit in all the studied water samples.

5. Conclusions

According the hydrochemical properties the studied samples can conclude that:

1) regarding to the concentration of major ions the surface and groundwater rich in Na^{+}, Mg^{2+}, Cl^{-}, and SO_4^{2-}, and the percent of Na^{+} and Cl^{-} increased in the locations of Tlol-AlBag and Sukkaria.

2) The SI increased suddenly in these two locations especially in Tlol-AlBag.

3) NaCl, $CaSO_4$, and $MgSO_4$ are the major hypotheticcal salts combination in the water of the valley and groundwater of shallow aquifer, and the percentage of NaCl increased suddenly in Tlol-AlBag to reach more than 70%, while in Sukkaria less than Tlol-AlBag.

4) The indicators above reveal to local source of salts because the upward leakage of saline water from the deep salt rich formations, especially Fatha'a Fm. The leakage may be a result of structural reasons.

5) These two locations are not reliable for water storage, while the others are reliable.

6) According to EC and SAR, the surface water in the locations of Hatra and Abo-Hamam and the groundwater of shallow aquifer are usable for irrigation with modern irrigation system for the salt resistance plants, while the water of Tlol-AlBag and Sukkaria are not reliable.

The evaluation of hydrochemistry is very important in the selection of water storage projects.

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Journal of Water Resource and Protection, **2012, 4, 657-662**
doi:10.4236/jwarp.2012.48076 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Analytical and Numerical Modeling of Flow in a Fractured Gneiss Aquifer

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Received May 16, 2012; revised June 22, 2012; accepted July 2, 2012

ABSTRACT

Investigating and modeling fluid flow in fractured aquifers is a challenge. This study presents the results of a series of packer tests conducted in a fractured aquifer in Freiberg, Germany, where gneiss is the dominant rock type. Two methods were applied to acquire hydraulic properties from the packer tests: analytical and numerical modeling. MLU (Multi-Layer Unsteady state) for Windows is the analytical model that was applied. ANSYS-FLOTRAN was used to build a two-dimensional numerical model of the geometry of the layered aquifer. A reasonable match between experimental data and simulated data was achieved with the 2D numerical model while the solution from the analytical model revealed significant deviations with respect to direction.

Keywords: Ansys/Flotran; MLU for Windows; Gneiss; Packer Test; Fractured Aquifer

1. Introduction

Fractured aquifers are very important for groundwater supply because about 75% of the earth's surface consists of fractured aquifers [1] and 25% of the global population is supplied by karst waters [2]. Flow velocity in fractured gneiss is known to be highly variable over a range of scales and uncertainties which arises from heterogeneous flow pattern in fissures and fractures. This has significant implications on water resource management from borehole to catchment scales. In addition, understanding flow heterogeneity in the aquifer is of great importance for groundwater protection and for predicting contaminant transport.

Theis [3] was the first scientist to conduct a transient analysis of the groundwater flow. After Theis, many researchers like Warren and Root [4], Kazemi [5], Odeh [6], Hantusch and Thomas [7], and Streltsova [8] studied the flow through fractured rocks in the context of petroleum and groundwater engineering. Jenkins and Prentice [9] described groundwater flow in a single fracture with a very large permeability. Sen [10] used an analytical solution to analyze fractured gneiss with a linear flow pattern. Cohen [11] used a two-dimensional numerical model to analyze an open-well test in fractured crystalline rock. Gernand and Heidtman [12] used the analytical model by Jenkins and Prentice to analyze a pumping test in a fractured gneiss aquifer. Schweisinger *et al.* [13] analyzed transient changes in a fracture aperture during hydraulic well tests in fractured gneiss. Wang and Cui [14] analyzed fluid flow and heat transfer by using the distributed resistance application in ANSYS FLOTRAN. Their analysis was done without comparing the modeled results with those from experiments (Gu *et al.* [15] and Cen and Chi [16]). Slack [17] proposed a theoretical analysis for the slug test which couples elastic deformation with fluid flow within a fracture. Molina-Aiz *et al.* [18] used ANYS FLOTRAN to simulate the velocity and temperature in a ventilated greenhouse. Crandall *et al.* [19] used ANSYS FLUENT to obtain the flow solution in a fractured aquifer.

Several analytical solutions are implemented in software packages like AQTESOLV, Aquifer Win32, AquiferTest Pro, StepMaster, and MODPUMP to determine the hydraulic parameters of aquifers. Some of these packages offer analytical solutions for fractured aquifers. However, these software packages have certain limitations due to the more or less arbitrary selection of analytical solutions that are implemented. MLU for Windows [20] is based on a completely different concept: It is a multi-layer analytical model for confined and unconfined aquifers and can thus be used for any kind of groundwater testing scenario.

Several numerical models have been developed to simulate the flow and transport in fractured aquifers. Examples are GeoSys/Rockflow and TOUGH2. Walsh *et al.* [21] modeled flow and mechanical deformation in fractured rock using Rockflow/GeoSys. McDermott *et al.* [22], Myrttinen *et al.* [23] and others used the numerical simulator GeoSys/Rockflow to simulate the flow and transport in fractured rocks. Also, Pruess *et al*. [24], Pruess and García [25], and others present results for multiphase flow in porous and fractured aquifers using TOUGH2. A detailed review on characterizing flow and transport in fractured geological media is presented by Berkowitz [26]. However, models such as Rockflow, Rockflow/Geosys, and TOUGH2 do not apply the well-known Navier-Stokes equation. They use the fact that the Navier-Stokes equation can be linearized as long as the Reynolds number is less than 10 and thus can be replaced by the much simpler Reynolds lubrication equation. However, there are some doubts that the local cubic law is valid in some cases and only a few publications used models based on Navier-Stokes Equation.

The present study was motivated by the need of improving conceptualization of fractured gneiss through a combination of fieldwork and modeling. Studies of flow properties using packer techniques and geophysical readings were used to investigate groundwater flow in fractured gneiss. The major task of the paper was to model flow in fractures by means of Navier-Stokes equation on the one hand, and to assume fractured zones as a continuum and thus to apply Darcy's Law on the other hand. MLU for Windows was chosen as analytical model because it is an integrated tool to evaluate pumping tests for multi-layer confined aquifers. Finally, the analytical solution was compared to the numerical solution with respect to evaluating and estimating permeability of fractured aquifers.

2. Site Description

Gneiss is the dominant crystalline rock at the test site in Freiberg. All six wells at the test site of the TU Bergakademie Freiberg with a total depth of 50 m and 100 m, respectively, are lacking any kind of casing or screen (except for a upper protection casing (at 3 to 5 m)) and have diameters of 4 to 6 inch. The gneiss at the site is a medium- to coarse-grained Inner Gneiss. Major fractures were identified by geophysical borehole logging in six boreholes at depths of 11 to 11.5 m and minor fractures at 14 to 16, 22 to 23.6, 31.3 to 31.9, 37.5 to 38 and 47 to 47.6 m [27]. Caliper, Single-Point Resistance (SPR), High Resolution Detector (HRD), Gamma-gamma soundings, and neutron-neutron soundings were used to identify fracture zones [27]. Consequently, higher values of hydraulic conductivity are associated with the horizontal fracture zones.

The six wells serve as direct, vertical connection between the zones of higher permeability. Thus, the presence of wells intersecting with the six zones of higher hydraulic conductivity can significantly perturb fluid flow. The flow in the fractured gneiss aquifer is assumed to be extremely heterogeneous with high flow velocities in the fracture zones and very low velocities in the block matrix. Non-fractured gneiss itself is nearly impermeable.

3. Methods

Hydraulic properties of rock materials can be estimated in both: laboratory and field. However, hydraulic properties obtained in the laboratory are not a true representation of the aquifer. Therefore, especially in fractured rocks, packer tests and tracer tests are indispensable. Packer tests are a well-known method to determine aquifer properties in open boreholes [28-30]. They can be used in uncased boreholes to determine the hydraulic conductivity of the individual horizon by isolating a zone between two packers or isolating a certain part of the borehole with a single packer. The equipment needed for packer tests includes an air compressor, a submersible pump, inflatable packers, and pressure transducer probes (Diver, type CTD, Schlumberger).

Two general methods of hydraulic testing have been used: Double packer tests, and single packer tests. Single packer test provide hydraulic data either for the borehole above or below the packer. The double packer test was performed for the most dominant fault in the range 11 to 12 m. The pumping rate was kept constant at about 10 L/min until a steady drawdown was achieved. For the test, the submersible pump is mounted between the two packers and the resulting drawdown. The duration of the pumping test was 6 to 7 hours.

For the single packer test the packer was mounted 13 m below ground surface. The pumping rate in the single packer was held constant at about 16 L/min and was maintained for 6 hours until a steady state drawdown was achieved. In both cases recovery was monitored, too.

Two approaches are addressed in this paper to calculate and evaluate the hydraulic properties for a fractured gneiss aquifer:

- An analytical solution using MLU for Windows (based on Darcy's Law)
- A numerical solution using ANSYS-FLOTRAN (based on Navier-Stokes Equation)

3.1. Analytical Solution

MLU for Windows is a tool for single- and multi-layer aquifers (both, confined and unconfined), which combines Stehfest's method, the superposition principle, and the Levenberg-Marquardt algorithm. Stefest's method is performed in the numerical solution to convert the Laplace domain to the real domain. Parameter estimation is performed by applying the Levenberg-Marquardt algorithm [20]. MLU assumes a homogenous, isotropic, and uniform aquifer.

A one layer model was used to calculate the hydraulic conductivity for the first horizontal fault zone. There, the fractured layer is embedded in impermeable gneiss. The numerical values for the hydraulic conductivity are presented in **Table 1**. The values for the permeability were obtained from forward modeling using the Theis method assuming a thickness of 0.5 m and evaluating each single observation well individually.

Conductivity increases in direction of well 2 in comparison to wells 3, and 4 (for spatial distribution see **Figure 1**).

A two layer model was used to calculate the hydraulic conductivity from the single packer test for the fracture zones below the packer. Only two fracture zones were modeled because they are assumed to be the most significant ones regarding depth and permeability. The values determined for the hydraulic conductivity of the different monitoring wells are presented in **Table 2**.

The hydraulic conductivity of well 2 and well 4 is higher than that of well 3 and well 6. Concerning technical issues, it was difficult to measure the drawdown in the pumping well (well 5). So, no parameter was calculated for it.

Table 1. Hydraulic conductivity (double packer test) for the upper fracture zone.

Well No.	Hydraulic Conductivity (m/sec)
Well 2	5.83×10^{-4}
Well 3	1.17×10^{-4}
Well 4	1.17×10^{-4}
Well 6	1.25×10^{-4}

Table 2. Hydraulic conductivity (single packer) for the fracture zones below the packer at a depth of 14 m.

Well No.	Hydraulic Conductivity (m/s)
Well 2	1.25×10^{-4}
Well 3	8.33×10^{-5}
Well 4	1.25×10^{-4}
Well 6	2.5×10^{-5}

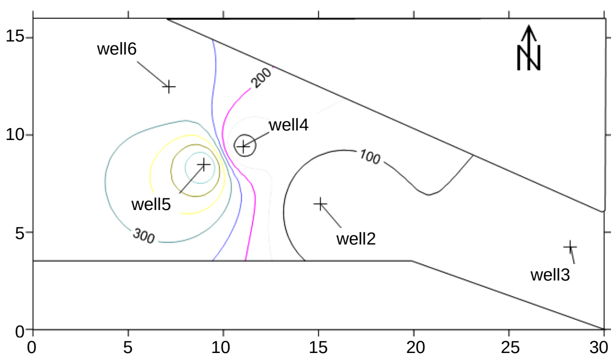

Figure 1. Drawdown at the Freiberg test site at the end of the double packer test.

3.2. Numerical Model

The ANSYS/FLOTRAN CFD (Computational Fluid Dynamics) software package offers comprehensive tools for analyzing two-dimensional and three-dimensional fluid flow fields. FLOTRAN is a finite element analysis program for solving fluid flow and conjugate heat transfer problems. The governing equations solved by FLOTRAN are the Navier-Stokes equations combined with the continuity equation, and the thermal transport equation. The general purpose of the CFD modul of ANSYS FEA systems is to solve a large variety of fluid flow problems. ANSYS/FLOTRAN simulates laminar and turbulent compressible and incompressible flows, single or multiple fluids, and thermal/fluid coupling [31]. FLUID141 and FLUID142 are two element models in ANSYS/FLOTRAN [32,33]. In this paper, FLUID141 was utilized for the 2D model.

In FLOTRAN CFD elements, the velocities are obtained from the conservation of momentum principle, and the pressure is obtained from the conservation of mass principle [34]. The matrix system derived from the finite element discretization of the governing equation is solved separately for each degree of freedom. The flow problem is nonlinear and the governing equations are coupled. The number of global iterations requires achieving a converged solution that may vary considerably, depending on the size and stability of the problem. The degrees of freedom are velocity, pressure, and temperature [35].

Figure 2 shows the geometry, node locations, and the coordinate system for a typical quadrilateral and triangular element. The element is defined by three nodes (triangle) or four nodes (quadrilateral) and by isotropic material properties.

The fluid properties density and viscosity were specified for the element and were then meshed automatically in ANSYS. The smaller the size of elements is defined, the more accurate is the result that can be achieved by the model. However, very fine element sizes result in high CPU time. In the current study, models with different meshing sizes were applied and evaluated.

ANSYS-FLOTRAN can account for fractures influenced by the concept of distributed resistanceand was applied in the element 141 through the use of real constants. The distributed resistance refers to the macroscopic representation of geometric features that are not directly concerned with the region of interest. This concept is a convenient way to approximate the effect of porous media (such as a filter) or other flow domain features without actually modeling the geometry of those

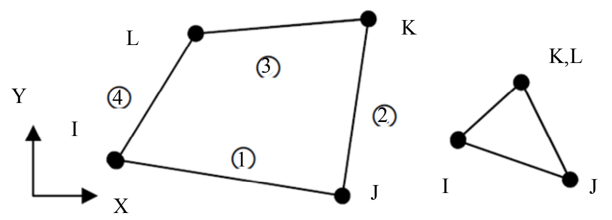

Figure 2. FLUID141 2D fluid-FLOTRAN element (modified after ANSYS, 2009).

features. Also, it can be used to simulate the flow in a fractured aquifer. A distributed resistance is an artificially imposed, unrecoverable loss associated with the geometry that is not explicitly modeled. Any fluid element with a distributed resistance will have a real constant set number greater than 1 [36].

The flow resistance, modeled as a distributed resistance, is caused by one or a combination of these factors: localized head loss (*k*), friction factor (*f*), and permeability (*C*). The equation for the total pressure gradient in x direction is shown below (1) [31]. It is a sum of the factors mentioned before.

$$\frac{\partial P}{\partial x} = -k \cdot \rho \cdot u_x \left|u\right| + \frac{f}{D_h} \rho u \left|u\right| + C\mu \cdot u_x \qquad (1)$$

where:

ρ = is the density (mass/length³)

μ = is the viscosity (mass/(length*time))

f = is a friction coefficient (dimension-less; calculated by the program): $f = aRE\text{-}b$

RE = is the local value of the Reynolds Number (calculated by the program): $RE = (\rho uDh)/\mu$

a, b = are the coefficient and exponent of Reynolds number, respectively, used in fraction factor calculation

C = is the FLOTRAN permeability ($1/\text{length}^2$).

FLOTRAN permeability is the inverse of the intrinsic or physical permeability.

The unit of the distributed resistance is $1/\text{length}^2$. The permeability of the cells in fracture zone 1 was found to be in the range of 0.00003 to 3e−8 m/s. This value has to be converted into a value that can be put into ANSYS. The flow rate of water through fractured gneiss is proportional to the hydrostatic pressure difference (δP). The hydrostatic pressure is normally expressed as a pressure potential $h = p/(\rho \cdot g)$, where ρ is the liquid density M/L^3, g is geravical accelarion L/T^2, h has the dimension L and is equivalent to the hydrostatic head. The calculation was carried out using ANSYS V.12.1.

The permeability value depends both on the material and the fluid. The permeability for Newtonian liquids during laminar flow through inert non-swelling media is inversely proportional to the fluid viscosity η. Therefore, the intrinsic permeability for the material is defined as $k' = k \cdot \eta$. In this equation k' is a material property independent from the fluid and with a dimension of L^2 [36].

The results from numerical modeling using ANSYS-FLOTRAN were calibrated with data from the packer test. The horizontal hydraulic conductivity was assumed to be anisotropic within a model cell. Heterogeneity was simulated by varying the horizontal hydraulic conductiveity between individual model cells or layers. The vertical hydraulic conductivity is based on specified values of the horizontal hydraulic conductivity.

Figure 1 illustrates high drawdown in observation well 6. Moreover, the shape of the drawdown and the high hydraulic gradients between wells 5 and 6, imply that there is a fault between them. Pumping generates a cone of depression in the hydraulic potential field that both expands outward and deepens with time. Drawdown values were obtained from packer tests using double packers (**Figure 1**). A higher drawdown was observed at monitoring well 6 with about 2.7 m and a lower one at monitoring well 2 and well 3 with about 30 cm. Inverse distance weight interpolation was used to generate the contour lines. Drawdown at the pumping well (No. 5) could not be monitored, because a double packer system with the submersible pump between both packers was used. Thus the drawdown shown in **Figure 1** was estimated to be 5 m.

For the double packer test the first fracture zones were modeled as a confined aquifer with constant head boundary zones on both sides and no recharge. The mesh size used was 0.04 m around the pumping well and 0.1 m close to the margin of the model. Due to the geometry the mesh used comprised over 40,000 elements. The pressure equation was solved using a pre-conditioned conjugate gradient method for the incompressible flow. Preconditioned Conjugate Gradient (PCG) is the most robust iterative solver in ANSYS. The exact method is the semi-direct conjugate direction method that iterates until a specified convergence criterion is reached.

4. Results and Discussion

The uppermost fractured zone located at between 11 and 12 m depth has an average thickness of 0.5 m. The simulation period of 402 minutes was chosen according to the time of the packer test. The differences between the measured and computed values are mainly due to the strong dependence of the coefficients on hydraulic conductivity which is not constant in the aquifer but highly heterogeneous. The horizontal hydraulic conductivity is assumed to be isotropic within a single model cell (**Figures 3** and **4**).

The simulation period was 6 hours. Concerning boundary conditions it was assumed that there is no drawdown at the left and right margin. The pressure is continuous across the fracture from block to block. Five fracture layers were included in this model. Moreover, hydraulic

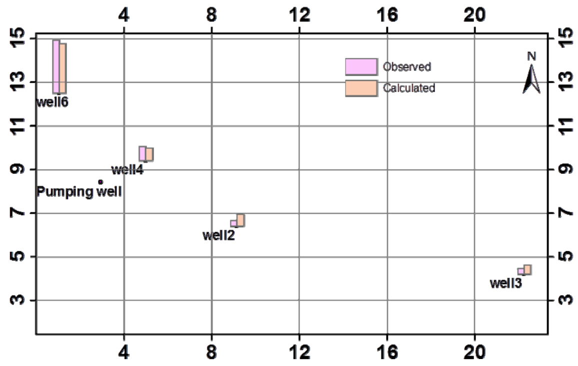

Figure 3. Map with arbitrary numbers in meter of observed drawdown during double packer test versus drawdown modeled with ANSYS-FLOTRAN.

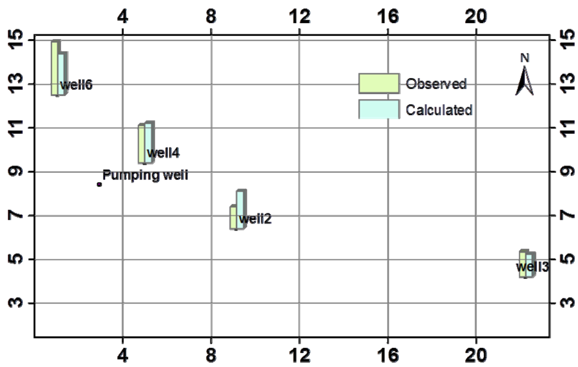

Figure 4. Map with arbitrary numbers in meter of observed drawdown during single packer test versus drawdown modeled with ANSYS-FLOTRAN.

conductivity was determined in the fracture zones between 3×10^{-6} to 9×10^{-6} m/s. In fact, the magnitude of the response of groundwater levels in the gneiss aquifer spatially varied indicating heterogeneity in the fractured gneiss. Similar to the first model, a simplified 2D-groundwater flow model was built to simulate the single packer test in the fractured gneiss.

Figure 4 depicts the difference between the observed and the calculated head for the single packer test. Good matching was observed at well 3 and well 4. In contrast, only sufficient matching can be seen at well 2 and well 6.

Calibration was performed by trial and error. The hydraulic conductivity of the fracture layer was between 3×10^{-5} to 3×10^{-8} m/s and 10^{-15} m/s or less for the non-fractured gneiss. The hydraulic conductivity of the two-dimensional model decreases with increasing depth. As stated before, having more and better information about the fracture's geometry, roughness and the network would give better matching between the observed and simulated heads.

5. Conclusion

Navier-Stokes equation is essential to predict the groundwater flow in the fractured gneiss aquifer at this scale and when turbulent flow is likely to occur. The two methods used to identify the permeability of the aquifer were analytical and numerical modeling. They are reasonable but care should be taken with respect to the interpretation of the results because of the uncertainties in the characterization of aquifer properties. In general it can be concluded that it is recommendable to carry out tracer tests in order to increase the accuracy of the model. At the investigated test site, hydraulic conductivity decreases from the right side to the left side due to the decrease in fracture thickness. The hydraulic conductivity decreases with increased distance from the ground surface (depth) because the number and width of fracture openings decrease with depth. Generally, the hydraulic conductivity values of the analytical solution were higher than the values obtained using the numerical approach. Finally, both the analytical and the numerical model proved to be useful tools for improving the knowledge of the fractured gneiss aquifer and for identifying the various flow components. ANSYS-FLOTRAN will give a better prediction of aquifer response than MLU for Windows because it is based on a non-linear equation.

6. Acknowledgements

The authors wish to thank Wondem Gezahegne and Rudy Abo for their supporting during fieldwork. Also, we want to thank anonymous reviewers for their valuable comments and suggestions. This work has been supported by the Department of Hydrogeology at TU Freiberg.

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Journal of Water Resource and Protection, **2012, 4, 663-673**
doi:10.4236/jwarp.2012.48077 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Attempts to Answer on the Origin of the High Nitrates Concentrations in Groundwaters of the Sourou Valley in Burkina Faso

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Received June 3, 2012; revised July 12, 2012; accepted July 21, 2012

ABSTRACT

Within the framework of the contract of Sourou River, a survey of the groundwater quality was performed through 7 campaigns of water sampling and analysis from 2006 till 2012. The water samples resulted from 23 drillings and 9 wells located in the Sourou Valley. Among the analyzed physico-chemical parameters, the nitrates concentrations observed were worrisome. Out of 32 water sources, 14 (44%) supplied a nitrates content superior to the WHO threshold value for drinking water (50 mg NO_3/L). Very high concentrations, superior to 500 mg NO_3/L with a peak in 860 mg/L, were observed. Given the important variations observed from a sampling point to another, a generalized contamination of the total aquifer was not possible. An individual diagnosis allowed to identify the possible causes of this degradation. Several sources of contamination, in connection with the anthropological activities, were observed near the water facilities (drillings/wells): animal and human wild defecation, presence of nontight latrines, solid waste, wastewater discharges. It is also advisable to wonder about the impact of the dynamite use for digging wells, this one being able to leave nitrates in the water. With regard to the intensive use of water from the strongly contaminated wells and drillings by the rural populations of Sourou, implementing protection areas within which would be eliminated the sources of contamination in addition to health education among populations could improve the situation. Care should also be taken in the use of nitrates explosives for digging new wells or drillings.

Keywords: Burkina Faso; Sourou; Groundwaters; Nitrates; Pollution

1. Introduction

According to the United Nations Development Program [1], access to safe drinking water in Burkina Faso clearly improved these years with a national rate of water access passed from 18.3% in 1993 to 66.3% in 2007. These good performances are the consequence of the efforts undertaken by the country to achieve the Millenium Development Goals (MDG) knowing that water access constitutes a lever of development and a mean to fight against poverty. According to the United Nations, the measures taken led to the reinforcement of the infrastructures of water supply. The network of drinking water adduction which was of 881 kilometers in 1986 reached 3129 kilometers in 2004 while between 2006 and 2007, the projects and programs allowed the realization of approximately 1882 drillings. The situation is undoubtedly variable from one place to another of the country, urban environment being privileged compared to rural environment.

Although Burkina Faso already reached the MDG for the access to safe drinking water [1], the situation is not therefore satisfactory, in particular in rural environment where the populations are confronted with the optimal management of the water supply points. Who has not met these drillings installed within the framework of cooperative projects and which, after a few years of operation, break down and aren't repaired?

But beyond the quantitative aspect, it is also advisable to remain vigilant on the level of the water quality consumed by the populations. If in urban environment, distributed water is the object of regular control, it is not the same in rural environment where the indicators of drinking water quality are missing due to the lack of analytical data. An improvement of knowledge is however essential to make the water services more powerful and to rein-

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force the policy for the access to safe water in the country. It is in this direction that in the National Action Plan for Integrated Water Management Resources (APIWRM, 2003), one of measurements relates to the installation of networks for water analysis [2,3].

The present contribution is devoted to the investigation on nitrates, an important physicochemical parameter which characterizes the quality of drinking water for which World Health Organization (WHO) fixed the threshold value of 50 mg/L. This study was conducted from 2006 to 2012 in the Sourou Valley in Burkina Faso. It consisted in evaluating the nitrates contents of water samples from wells and drillings located primarily in the Sourou Province. This survey of groundwaters nitrates contents is part of the data produced by the network of surface and groundwaters quality survey which was initiated within the framework of the contract of the Sourou River [4]. After a first restitution related to the general quality of surface water [5] and of groundwaters [6], it appeared important to examine the problems related to the high nitrates concentrations observed for wells and drillings in this zone.

After the presentation of the context of the study and the methodology used, the results obtained will be presented. We will particularly be interested in the results exceeding the standard of the WHO and more specifically through three case studies in the villages of Yaba, Diouroum and Kiembara. We will try to understand the causes of these raised concentrations of nitrates before proposing some reflections as regards to good practices of management to be implemented in order to solve the problem.

2. Context of the Study

2.1. One Action of the Sourou River's Contract

This study was realized within the framework of the implementation of a project of river contract in the Sourou's watershed. It constitutes one of the activities registered in the action program of this river contract.

In 2003, through cooperation with the Walloon Region of Belgium, a river contract based on the Walloon model was initiated [4]. This model is an approach of integrated and participatory management which aims at gathering within a river committee of the representatives of all the users of water. This committee has essential mission to define and implement a restoration actions program of the water resources, waterways and their accesses. This program is elaborate according to a consensual approach which takes care to integrate the concerns of each user while improving the environment protection. In Burkina Faso, it was proved that the river contract could also be an issue to fight against desertification and poverty.

This project which has been developed on nearly ten years was framed locally by a Burkinabè NGO, the COPROD (Convention for the promotion of a sustainable development) which dealt with the animation and the coordination of the activities. The Department of Environment of the University of Liege in Belgium ensured the general coordination and the scientific expertise for the account of the Walloon Region in collaboration with the Institute for Health Sciences Research (IRSS) of the Scientific and Technological National Research Center (CNRST) of Ouagadougou.

Considering the operational characteristic of the river contract, many activities were performed at the field [7]. The activities were divided into five sets of themes:

1) Coordination, animation and dialog between actors of water;

2) Improvement of knowledge through the data acquisition and the organization of many formations for the users of water and the local collectivities;

3) Communication, information and sensitization of schools and water users;

4) Restoration of the water resources and the environment which is concretized by an improvement of water access and environment protection (waterways protection and fight against desertification;

5) Income-generating activities in a context of fighting against poverty.

The present study is in line with an improvement of knowledge related to the quality of water resources in the Sourou basin. It also meets one of measures recommended by the APIWRM in the actions field No. 2 "water information system": action 2.2, the implementation of national networks to monitor the water quality, water uses, water requests and the risks. However, it is surprising that the National System on Water Information does not mention nitrates on the list of the basic parameters to consider for the groundwaters quality [8].

2.2. The Zone of Study

The Sourou valley is located in the North-West of Burkina Faso, in the area of the Mouhoun loop. The Sourou River takes its source in Mali at the level of Baye. It makes border between Burkina Faso and Mali, by then crossing the Burkina Faso from north to south before joining Mouhoun River at Léri. The Sourou's watershed occupies a surface of 16,200 km^2 but it is primarily the central part of the watershed located on the left bank of the river (approximately 5000 km^2) which is the object of this study (see **Figure 1**).

The Sourou valley is especially known for its hydro-agricultural installations following the erection of dam valves at the junction of Sourou and Mouhoun rivers in 1984. The realization allowed to increase significantly the level of water of the Sourou River, the river draining

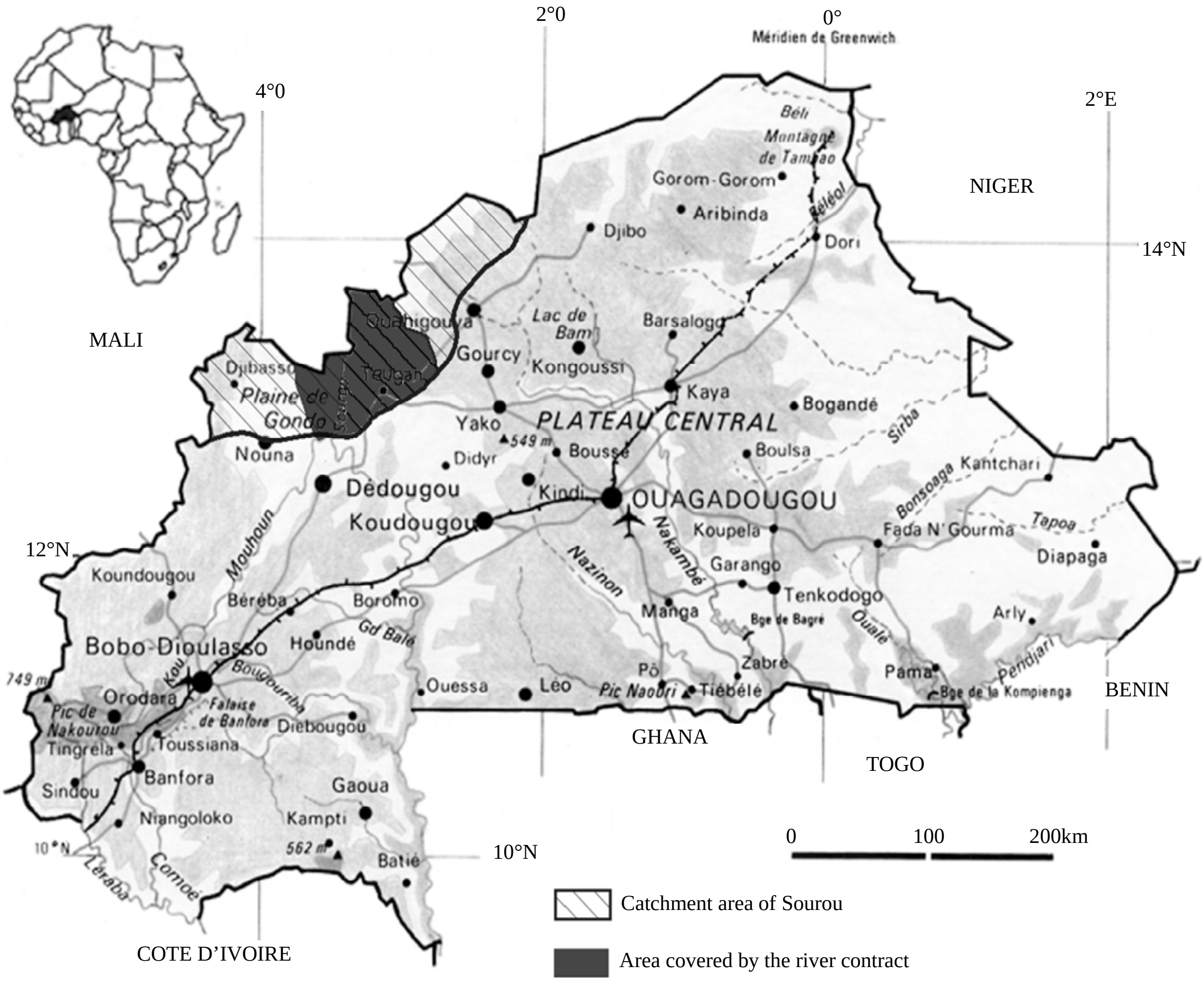

Figure 1. Zone covered by the contract of Sourou River, Burkina Faso.

important quantities of water (600,000,000 m^3) through the valley [5]. This availability of water thus allowed the creation of irrigated perimeters, making the Sourou Valley an important agricultural production zone.

Currently, the irrigated perimeters extend on a surface of 3200 ha and are managed under the direction of the Sourou Valley Development Authority (SVDA). The SVDA also envisages to extend the irrigated perimeters area with a new additional zone of 2033 ha. The Sourou valley thus constitutes an important agrarian production zone benefitting the whole country. Onion, tomato, rice, corn, cabbages, lettuces, … are produced in Sourou. Irrigated wheat cultures were also developed within the framework of cooperation with Morocco. Unfortunately the collected quantities not reaching the envisaged outputs, the culture of wheat was abandoned. The slopes of sprinkling to after the abandonment can still be observed at the field. The activities in the valley are thus directed towards the exploitation of the irrigated perimeters. Elsewhere, they are traditional cultures (millet, sorghum, niebe, …) not irrigated which are installed. The farmers also supplement their incomes through activities of cattle, sheep and goats breeding, from some heads of cattle per family to herds which can be very important.

Water access for Sourou populations is possible, from various sources of supply, primarily from groundwaters but certain insular villages or close to the river use surface water. Except for some more important localities like Tougan, chief town of the Sourou Province, which profits from a partial system of water adduction, the rural populations generally feed on from water tanks or traditional wells to a relatively dense network of drillings installed by NGO within the framework of cooperative projects or programs supported by the State, in particular by the second Soils Management National Plan (SMNP 2).

The availability of groundwater is rather stable. The Sourou aquifer system is located in a sedimentary zone and consists of hard stones represented by sandstones and limestone-dolomites which can be crossed by faults. The thickness of the aquiferous sandstones is estimated to a hundred meters [9]. The depth of drillings is about a sixty meters in the zone of this study. The water level in the traditional wells is variable from one site to another, the depth being of approximately 10 to 20 m. The refill of the aquifer can be established through a slow infiltration in the subsoil. This diffuse refill, could be supplemented by a preferential water flow through fractured zones.

Within the framework of the Sourou river contract, an inventory of wells and drillings were carried out. This inventory identified many nonfunctional works which were the object of repair within the framework of this river contract. More than 100 works were thus given back in activity. Once repairs carried out, it appeared convenient to check the distributed water quality. If initially, the repaired drillings were the object of analysis, thereafter, directed by the nitrates problem, other samplings were carried out.

3. Material and Methods

3.1. Sampling and Field Information Gathering

The network of groundwater resources analysis consisted of 32 control points along with 23 drillings and 9 wells (traditional or modern, large diameter wells). Seven samplings campaigns were carried out from 2006 to 2012 (see **Table 1**). Six of the campaigns were conducted in dry season, and one in rain season. A total of 95 samples were taken and analyzed during the campaigns. Samples collected at the field were preserved at 4˚C in cool boxes, carried to the National Laboratory for Water Analysis in Ouagadougou and stored in a refrigerator before analysis. Some samples were blocked by the addition of mercuric chloride and analysed in Belgium at the laboratory of water resources of the Department of Environment, University of Liege. In addition to nitrates, the samples were also analyzed for other physicochemical parameters (pH, conductivity, turbidity, hardness, ions, NH_4, PO_4) and for microbiological indicators (fecal contamination indicators: *Escherichia coli*, Fecal coliforms and Fecal enterocoques) [6].

During sampling, ground observations, supplemented by information collected from water users and local authorities, fed the reflection in order to try to identify the potential sources of contamination.

3.2. Analysis Method

The analyses were carried out by the National Laboratory for Water Analysis of the Ministry of Environment in Ouagadougou, in the 2 - 3 days following sampling. Meanwhile, the samples were preserved at 4˚. Proportioning is carried out by molecular absorption spectrophotometry through nitrates reduction in nitrites by cadmium (spectro Hach DR2400 method 8171). The nitrites formed are proportioned by diazotization of the sulphanilamide which in the presence of N-ethylenediamine forms a coloured complex [10].

At the laboratory of the Environment Department of the University of Liege in Belgium, the standard NF IN ISO 13395 by analysis in flow and spectrophotometric detection after diazotization was the method applied.

In addition, a rapid semi-quantitative test nitrates was carried out at the field by the use of Merckoquant strips. This test provided a rather good estimate of the nitrates concentrations in water [11].

4. Results and Discussion

The average nitrates contents observed during the whole campaigns are presented in **Tables 2** and **3** for drillings and wells. A great variation in the values obtained can be observed which ranged from 1 to 860 mg NO_3/L. Important variations can also be observed between two works very closely located. Sometimes for less than one kilometer between two works, important differences can be noted. It does not appear to be a generalized contamination of the Sourou aquifer but localized situations. For a same work, the nitrates concentration is generally stable from one campaign to another, except where the values are high (see the studies of typical cases hereafter).

Whereas many wells and drillings respect the WHO standard of 50 mg NO_3/L, very high nitrates concentrations were recorded at certain places. Thus, the quality of

Table 1. Campaigns of water sampling and samples collected.

No. campaign	Period	Season	Number of samples	Number of well sample	Number of drilling sample
1	18/10/2006	Rain (end)	3		3
2	25-30/11/2007	Dry	15	5	10
3	24-28/02/2008	Dry	14	4	10
4	6-9/06/2008	Rain	14	4	10
5	12-13/12/2009	Dry	18	5	13
6	17-18/01/2011	Dry	15	4	11
7	19-21/01/2012	Dry	16	7	9
Total			95	29 (9 controls)	66 (23 controls)

Table 2. Nitrates content in drillings samples (average results).

Locality	Drilling name	Number of samples	Nitrates contents (mg NO_3/L)
Kouy	COPROD	3	18
Sono	Dispensaire	4	8
Kassoum	CEG	6	4.5
Wawara	COPROD	6	4
Dioroum	COPROD	5	648
Dioroum	Mosquée	1	1
Kassoum	Ecole	3	14
Niassan	Dispensaire	4	15
Niassan	AMVS	4	13.5
Di	Caisse populaire	6	119
Yaba	Dispensaire	4	115
Bassan	Bassan	1	45
Diélé	Dièlé	1	20
Bonro	Bonro	1	3
Bourgou	COPROD 1	1	150
Bourgou	COPROD 2	1	309
Dian	Dian	2	74
Guiédougou	Ecole	3	19
Boaré	Ecole	2	11.5
Kiembara	PNGT 2	3	137
Kiembara	AEP	2	69
Lankoé	Ecole A	2	84
Lankoé	Rimaélé	1	17
Total	23 drillings	66 samples	

Table 3. Nitrates content in wells samples (average results).

Locality	Well name	Number of sample	Nitrates content (mg NO_3/L)
Kouy	Mosquée	6	22.5
Kouy	Mission	1	263
Sono	Sono centre	4	44.5
Kassoum	Marché	4	15
Diouroum	Diouroum 1	1	205
Diouroum	PNGT 2	6	61
Yaba	Dispensaire	3	143
Kiembara	Mongolo	3	201
Bouaré	Ecole	1	25
Total	9 wells	29 samples	

water from the COPROD drilling in Diouroum is particularly worrying with a peak of 860 mg NO_3/L observed during campaign No. 3 (February 2008). To a lesser extent, but however with always raised concentrations, drillings of Di, Yaba, Bourgou and Kiembara presented values ranging from 100 to 200 mg NO_3/L. Out of the 23 analyzed drillings, the WHO threshold value was exceeded for 9, which corresponds to 39% of nonconformity (see **Figure 2**).

Concerning the wells, the maximum concentrations seem less lower than in drillings. However, raised concentrations were observed at Kouy, Yaba, Kiembara and Diouroum. 5 wells out of 9 delivered water with nitrates exceeding the WHO standard, which corresponded to 55% of nonconformity (see **Figure 3**).

Other parameters were analyzed in the framework of the contract of Sourou river activities [6] and nitrates did not appear to be the only factor of disturbance. Whereas the ammonium (NH_4) contents were relatively low for all the works, with concentrations generally lower than 0.5 mg NH_4/L, high concentrations of fecal contamination indicators (100 to more than 1000 *Escherichia coli*/100 ml) were observed for wells samples. On the other hand, drillings globally respected the WHO standard for this indicator (0 *Escherichia coli*/100 ml). We could not connect the nitrates contents with these fecal contaminations.

High percentages of nitrates in groundwaters were previously observed in the subregion and in Burkina Faso. In the subregion, Mali is also confronted to nitrates concentrations much higher than the WHO standard, according to the Ministry of Energy, Mines and Water [12] of this country, the situation indicates a local pollution of the water supply points. In Senegal, concentrations exceeding 500 mg NO_3/L were observed for the aquifer of quaternary sands in the area of Dakar at the time of campaigns conducted in 1987, 1988, 1995 and 1996 [13]. According to the authors of the study, the contamination of the aquifer by nitrates results primarily from horizontal and vertical flows related to nontight family latrines and organic waste deposits. Moreover, Fall [14] reported values from 165 to 873 mg NO_3/L in 7

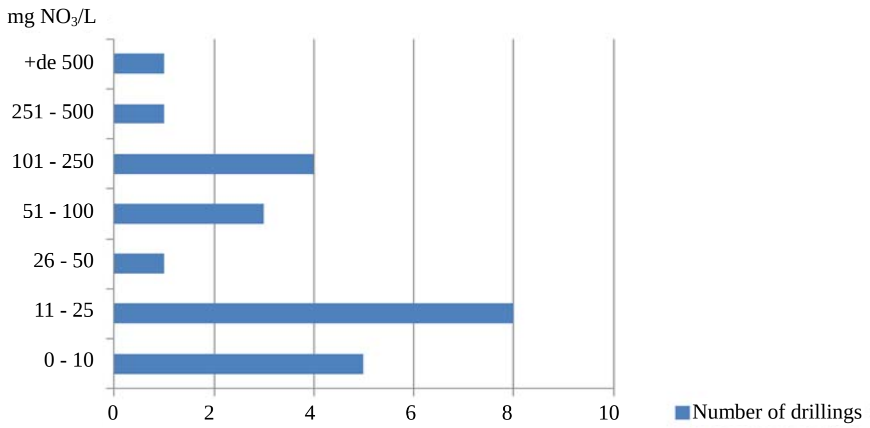

Figure 2. Nitrates contents (per classes) in drillings water samples.

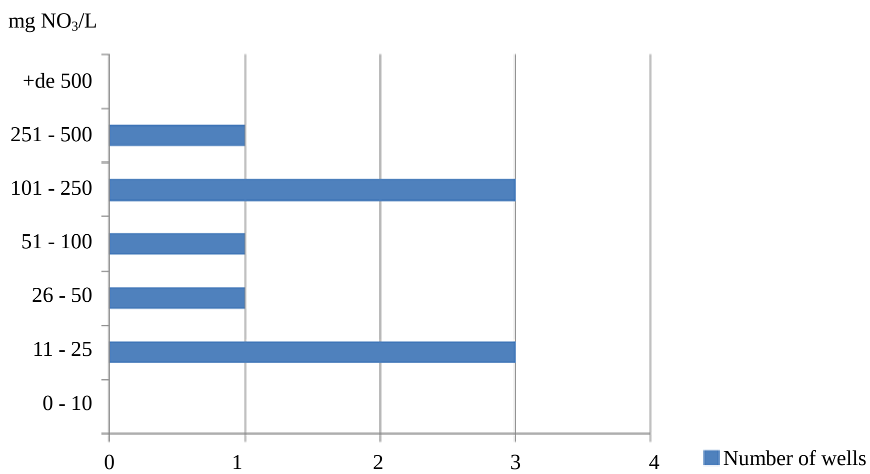

Figure 3. Nitrates contents (per classes) in wells water samples.

private wells of the town of Pikine, these analyses having been realized in 1992 by the Institute of Sciences of the Environment of Dakar.

At the country level, the National Water Information System (NWIS) of the DGIRH [15] through the follow-up network of water quality evoked raised nitrates concentrations, particularly in the urban centres. According to the design and implementation plan of the NWIS, these results could due to the insufficiency of sanitation and sewerage systems, but we are here in rural environment.

Concerning the Sourou zone, concentrations higher than 100 mg of nitrates per liter were announced by the information system AQUASTAT of FAO [16] in the areas of Mouhoun and Sourou. In addition, two campaigns of water analysis were carried out in May and August 2006 by Nabayaogo [17] within the framework of a study on the impact of the Sourou valley agricultural management on the water resources and the ecosystems. Only some drillings and wells located in the village of Niassan and on the riverside were subjected to analysis. Nitrates contents from 2.7 to 37.2 mg NO_3/L were observed in Niassan village by the author and correspond approximately to the range of values recorded in this same village during the present study (10 to 23.4 mg NO_3/L).

A study of the CIRAD [18] concludes that the pollution risk of the aquifer by nitrates of agricultural origin is nearly zero in the Sourou valley (zone of Di). On the other hand, concentrations reaching 41 mg NO_3/L were evidenced at the beginning of rainy season in the Sourou river, particularly downstream Di village, within the framework of this project. These nitrates could be due to the cultivation practices related to the production of tomato and onions at this place [5].

For the nitrates concentrations in groundwater exceeding sometimes at a large extent the WHO threshold value of 50 mg NO_3/L, what could be the sources of contamination in the Sourou valley? After rejecting the possibility of a natural origin, we tried to identify the anthropic causes which could explain these high rates of nitrates through field investigations at the respective sites.

Globally, at the selected sites, various potential sources of nitrates can be observed. Concerning diffuse pollution, the important cattle near the water supply points leads to deposits of defecation products in abundance, in addition to that of human origin. It is particularly the case at Diouroum village where the ground is strewn with excrements. In addition, specific organic matter deposits such as dunghills and composting areas can also generate rejections of nutrients. Latrines, although not very widespread in the Sourou villages can also punctually influence the quality of water (case of Yaba village), particularly in fractured zones where a fast contamination of the aquifer is possible.

Concerning nitrates coming from agricultural inputs cultures, since most of the water supply points are far away from the hydro-agricultural perimeters, we could not establish a connection between the fertilizers used and the contamination of groundwaters. Moreover, samples collected from Niassan village, next to the irrigated perimeters presented nitrates concentrations lower than the WHO standard.

It has also been announced that the digging of certain drillings and wells in hard sandstone made up required at certain places the use of dynamite (case of the village of Diouroum hereafter) which could be a source of nitrates release in water.

From the tracks evoked above, it appears difficult to generalize the situation to the whole area in which the variations are very important from one point to another. Within a few kilometers distance, we observed very different nitrates contents, sometimes even within the same village with concentrations varying from one point to another in significant proportions. In order to better understand the origin of these nitrates, we carried out three case studies in three problematic villages where an in-depth survey was carried out, namely Diouroum and Kiembara in the Sourou Province and Yaba in the Nayala Province. The investigations were performed on several periods from 2007 to 2012.

4.1. Three Case Studies

4.1.1. YABA Village

The village of Yaba (Province of Nayala, Commune of Yaba) counts 6618 inhabitants according to the 2006 INSD census [19]. This village has a dispensary and a maternity equipped with a drilling and a well with large diameter selected for this study.

The water able in the well was at eight meters depth (measured in January 2012). The well located at a few meters of the dispensary presented nitrates concentrations from 93 to 240 mg NO_3/L between 2007 and 2012. An average nitrates content of 115 mg/L was observed in the water of the drilling of about sixty meters of depth (see **Table 4**).

Within a radius of 10 to 20 meters around these water supply points, beside the dispensary and the maternity, two buildings occupied by nontight latrines were identified (see **Figure 4**). In addition to the function of latrines, these pits collect also the biological liquids coming from the maternity.

The first building located at 15 m of the well was abandoned since a few years. A new block including two latrines is today in activity. To approximately 100 m of the well and drilling, is an abandoned cemetery. No wild defecation was observed near the works.

Table 4. Nitrates concentrations in water samples of Yaba dispensary's drilling and well during the 2007, 2008, 2009 and 2012 campaigns.

Sampling source	Period	Nitrates concentration (mg NO_3/L)	Average (mg NO_3/L)
Yaba—Drilling	November 2007	122	
	February 2008	116	
	June 2008	143	
	January 2012	80	
			115
Yaba—Well	November 2007	240	
	December 2009	93	
	January 2012	97	
			143

In this case, the latrines can be responsible for these the raised concentrations of nitrates. A hydrogeological study should be able to determine the protection zone to set up and in which the activities should be regulated. It is of great importance to have near the health facility a source of water supply and latrines. However, the pits should be perfectly tight and subjected to a frequent draining under control far away the perimeter of protection of the well and the drilling.

4.1.2. DIOUROUM Village

The village of Diouroum (Province of Sourou, Commune of Tougan) counts in 2006 2048 inhabitants, essentially farmers and stockbreeders. The three analyzed water supply points are localized in a maximum radius of 1 km (see **Figure 5**). The drilling broken down for several years has been restored by the COPROD within the framework of the application of the contract of Sourou river. In 2010, again broken down, it was repaired by the inhabitants of the village. The nitrates concentrations are particularly high, a peak of 860 mg/L was observed in June 2008. In January 2012, the concentrations remained high but in clear reduction compared to the previous years. The two other points presented low but still worrying nitrates concentrations (see **Table 5**).

In the neighborhoods of these water supply points, the first observation relates to the important animal defecation, the animals coming to water itself into the basin next to the drilling. At a few hundred meters, one can observed butts, relics of old districts given up with their waste for a hundred years. A nontight dunghill is localised near the well No. 1.

Drilling would have been dug at the beginning of the Nineties. Work was particularly painful and idles because of the hardness of the rock to be bored, dynamite has been employed to come to end from this resistance. The depth of water is about 60 m. Well PNGT 2 was dug in 2005, its depth is of 10 m (measured in January 2012).

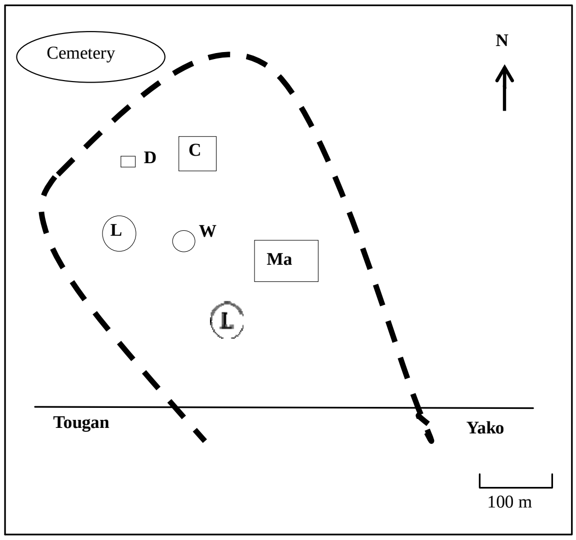

Figure 4. Localization of the water supply points within the health facility/medical center of Yaba and brief description of the close environment. W: Well with large diameter; D: Drilling; L: Latrines; C: Dispensary; Ma: Maternity; - - -: Village; —: Road.

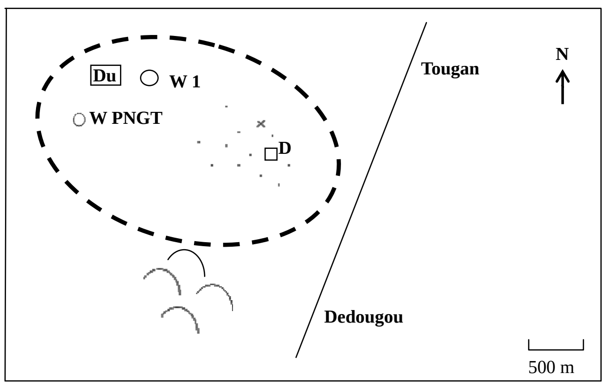

Figure 5. Localization of the three water supply points at Diouroum and description of the close environment; W1: Well with large diameter; W PNGT: Well PNGT 2; D: Drilling; Du: Dunghill; X: Defecation zone; ⌒: Old abandoned districts; - - -: Village; —: Road.

Table 5. Nitrates concentrations in groundwaters at Diouroum.

Sampling source	Period	Nitrates concentration (mg NO_3/L)	Average (mg NO_3/L)
Diouroum—COPROD Drilling	November 2007	750	
	February 2008	610	
	June 2008	860	
	December 2009	703	
	January 2012	315	
			648
Diouroum—Well 1	January 2012	205	205
Diouroum—Well PNGT 2	November 2007	45	
	February 2008	100	
	June 2008	37	
	September 2009	58	
	January 2011	72	
	January 2012	54	
			61

Except for the problems of breeding (defecation, manure), we did not observe near these water supply points other elements being able to disturb the quality of water. But what can be the contribution of the use of dynamite (nitroglycerin: $C_3H_5(NO_3)_3$ obtained by nitration with the nitric acid of glycerin)? One can suppose that the nitrates left by dynamite should grow blurred in the course of time. One should keep in mind that the drilling has more than 20 years and that the wells are also contaminated. However, this probable source of contamination should be taken seriously in consideration by avoiding the use nitrated explosives during the operations of drilling.

In addition, if the breeding also contributes to the deterioration of water quality, how to solve this problem? Knowing that the animals need water, is it not appropriate in this precise case to differentiate the water supply points? Some could be dedicated to human consumption, and the others to cattle watering with a sufficient distance in between to avoid disturbances. In this case, a better knowledge of the perimeters to be protected is essential.

4.1.3. KIEMBARA Village

The third case of study relates to the commune of Kiembara (Province of Sourou) located on the axis Tougan-Ouahigouya, halfway between these two cities. In 2006, the village counted 4605 inhabitants whose activities were varied: farmers, stockbreeders, truck farming, small shops. Three water supply points were analyzed: a drilling carried out within the framework of the PNGT 2, a traditional well, a fountain supplied with a drilling outside the village. In all the three cases, the nitrates concentrations were higher than the WHO standard (see **Table 6**).

PNGT 2 drilling is much attended. Near this drilling, in a radius of 10 to 20 m, dunghills, nontight latrines and wild defecation are observed. Two traditional wells close to the city hall are particularly vulnerable. The immediate surroundings of the wells are soiled by wastewaters and solid wastes. Located on the ground, they directly receive surface waters coming from the surface of the ground. Following the first results communicated to the city hall, a concrete curbstone was installed in 2011. The fountain is fed by a drilling with water tower (see **Figure 6**) and presents a water of better quality but not respecting however the WHO standard.

Notwithstanding the problem of distances to cover and the crossing of the road Tougan-Ouahigouya, is it not more appropriate to keep the water of the fountain for human consumption and the drilling or traditional wells one for other uses? Is it not also possible to extend the network of water adduction while placing equitably additional fountains throughout the village?

These three cases illustrate the complexity but also the specificity of each situation. They show also the interest of a local analysis, the variations being able to be important within the same village, within a short distance.

4.2. Impact on Health

The information obtained from the City hall of Kiembara, the dispensary of Yaba and the hospital of Tougan did not allow to establish a direct relation between the consumption of water with high nitrates concentrations and the consumers' health status. Even if cases of cancers are

Table 6. Nitrates concentrations in groundwaters at Kiembara.

Sampling source	Period	Nitrates concentration (mg NO_3/L)	Average (mg NO_3/L)
Kiembara—PNGT 2 Drilling	December 2009	106	
	Jannuary 2011	179	
	January 2012	125	
			137
Kiembara—Traditionnel well	December 2009	171	
	January 2011	259	
	January 2012	174	
			201
Kiembara—Fountain	December 2009	71	
	January 2011	66	
			69

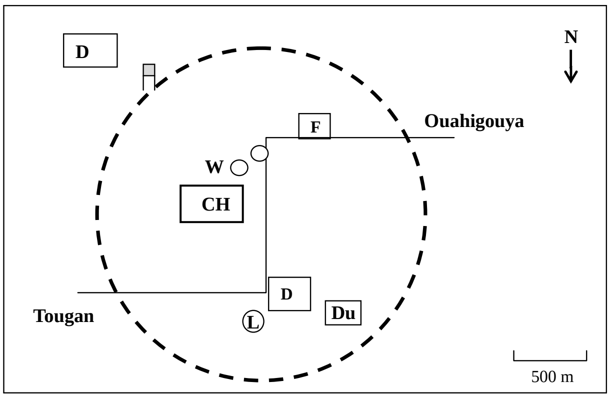

Figure 6. Localization of the three water supply points at Kiembara and brief description of the close environment; W: Traditional well; Du: Dunghill; D: Drilling; CH: City Hall; F: Fountain; ----: Village; L: Latrines; —: Road.

evoked, the patients are transferred towards more important centers for a better assumption of responsibility.

4.3. Conclusions and Prospects

From the analysis of the nitrates concentrations in groundwaters of Sourou, it was thus observed that on 32 analyzed water supply points, 14 provided water with nitrates exceeding the WHO threshold value of 50 mg NO_3/L. This proportion corresponds to 44% of noncomformity. Very high nitrates contents were revealed in some sampling sites.

We tried to understand the origin of these worrying concentrations. Several tracks of contamination were thus evoked near the works, in relation with the anthropic activities: animal and human wild defecation, presence of nontight latrines, presence of pits with dunghill, surfaces of composting, waste deposits, wastewaters rejections, …

To these possible causes of nitrates pollution, it is appropriate to wonder about the impact of the use of dynamite for digging on the groundwater quality. Many explosives contain in their structure a nitrate radical which could remain in water after drilling. However, the precise diagnosis is not obvious to establish and it requires to be examined case-by-case since the fluctuations of the nitrates contents can be important from one work to another, even if they are very close. The situation is not related to a general contamination of the aquifer of Sourou but born to punctual specific effects which deserve specific observations. Therefore, we examined three typical cases to try to better understand the origin of these nitrates.

The processes of nitrates transfer in the aquifer however deserve to be better documented from an aquifer susceptible to be fractured and sensitive to direct contributions of contaminants. It would be useful to follow nitrate flows from the sources of contamination and to define in particular the transfer times. A refined knowledge of hydrogeology near the works would be also necessary in order to specify the sensitive areas.

But beyond this expertise, it is the rural populations that are confronted to the access to non-safe water. In the case, could it not be advisable to relate water uses to the quality of available water? But that may require longer displacements for certain families towards not contaminated water supply points. Given the difficulty in setting up denitrification processes, could it not be appropriate to define perimeters of protection in which the water sources could be preserved from contamination? In the future, could it not be required to show prudence for the use of nitrated explosives during the digging of new works?

This contribution which aims at improving knowledge on the quality of water resources in Burkina Faso Registers to the policy of integrated water management to which Burkina Faso subscribed through the APIWRM. These advances in knowledge must be provided to the institutions in charge of the water policy and to the local collectivities, which within the framework of the decentralization, were seen entrusted the responsibility for the natural resources management. It is also appropriate that local participating structures as the Local Water Committees (LWC) catch these problems in order to improve the water services in response to the populations needs. Not to forget that in addition to the quantitative aspect related to water access, water to be provide to populations should be of good quality, with respect to the international standards of drinking water.

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Journal of Water Resource and Protection, **2012, 4, 674-685**
doi:10.4236/jwarp.2012.48078 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Aquifer Vulnerability Assessment and Wellhead Protection Areas to Prevent Groundwater Contamination in Agricultural Areas: An Integrated Approach

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Received June 22, 2012; revised July 25, 2012; accepted August 3, 2012

ABSTRACT

To implement successful policies for the protection of groundwater and curtail the possibility of water supply contamination, an early evaluation of aquifer vulnerability is needed. Rather than implementing broad restrictions to land use and effluent discharge, it is more cost-effective and economically favourable to approach protection in a stepwise manner by first assessing the intrinsic vulnerability of the aquifer when defining the level of land use control that is needed to protect groundwater quality. Following aquifer vulnerability evaluation, specific land uses and restrictions should be defined locally for each water supply within the wellhead protection areas (WHPAs), which are identified by means of the groundwater time of travel (TOT). The WHPA should be established for each individual situation, considering the level of vulnerability of the exploited aquifer. We applied our findings to a specific test site in the Piemonte region of NW Italy, following the current local procedure for individuating the WHPAs. Using data gathered from this site-specific exercise, we identified that the procedure allows methods that consider only aquifer parameters to evaluate vulnerability and discourages the use of techniques that already compartmentalize soil parameters in the vulnerability assessment.

Keywords: Groundwater Protection Zones; WHPA; Vulnerability; FEFLOW; Piemonte; Italy

1. Introduction

Groundwater quality in many parts of the world has experienced significant degradation due to agricultural, industrial and/or commercial activities. Historically, damage to groundwater supplies has often occurred when contaminants reach aquifers via the vertical pathway introduced by surface wells. Because of the importance of groundwater, and the difficulty and expense in remediating groundwater supplies, steps are now often taken to prevent initial pollution. Those steps can include protecting the whole aquifer, as well as the area surrounding the surface wellhead, from inadvertent contamination [1-6].

Use of the term "aquifer pollution vulnerability" began in the 1970s in France [7] and more widely in the 1980s [8-10], when it became increasingly clear through research that many aquifers were suffering from significant anthropogenic contamination resulting in degradation which compromised usability of the resource. Several studies have targeted the development of vulnerability mapping techniques; results of these investigations have led to new definitions applicable to groundwater protection issues [11-20]. "Vulnerability" is generally defined as the (intrinsic) sensitivity of an aquifer to being adversely affected by a contaminant; "groundwater pollution hazard" relates to the probability that groundwater in an aquifer will be contaminated at concentrations that pose a risk to human health or the environment that is hydraulically connected to the groundwater [4]. An absolute (numeric) index of aquifer pollution vulnerability is far more useful than relative indicators for all practical applications in land-use planning and effluent discharge control. With this goal in mind, several methods proposed in the literature are focused on vulnerability assessment. They vary by the parameters and mathematical expressions considered. GOD [10] and modified GOD [4], DRASTIC [9], EPIK for karst settings [21] and SINTACS [22] are acronimous of the most widely utilized processes. Each method can provide a numerical index which is generally correlated to a vulnerability class definition that is qualitatively described at the end of the evaluation process (usually high, moderate, low and negligible). The selection of the appropriate methodology depends on the hydrogeological setting and

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available data. It is quite possible to have high vulnerability but no pollution hazard because of the absence of significant subsurface contaminant load, or vice versa [4]. Following this approach, an inventory of potential subsurface contaminant load is necessary to design adequate safeguards for specific situations.

In order to protect the groundwater intercepted by a production well, it is essential to develop a thorough understanding of the groundwater flow system and to delineate the area surrounding the well where potential contamination could occur [23]. The proximity of the land-use activity to a groundwater supply (well or spring) is a critical factor in determining the potential for contamination [24,25]. More specifically, the pollution threat depends on 1) whether the activity is located within the (subsurface) capture area of the supply and 2) the horizontal groundwater flow time in the saturated aquifer. In order to completely eliminate the risk of unacceptable pollution of a supply source, all potential activities that might lead to contamination in the recharge zone would have to be prohibited. This will often be unsustainable or economically impractical, especially in developed areas with pre-existing land-use constraints [26]. It may be more practical to segregate the recharge zone, so that the most stringent land use restrictions will only be applied in areas closer to the source [27].

To achieve the necessary segregation, a series of generically concentric surface zones around the groundwater source can be defined through knowledge of local hydrogeological conditions and the characteristics of the groundwater supply source itself [28]. Once delineated, the protection areas may be managed to prevent contamination and for clean-up if contamination occurs. The supply protection areas must protect the source against persistent contaminants as well as those that degrade over time [29]. Both are necessary for comprehensive protection. Such an area is referred to as the wellhead protection area (WHPA). The US EPA [30] early defined a WHPA as "the surface and subsurface area surrounding a water well or well field, supplying a public water system, through which contaminants are reasonably likely to move toward and reach such water well or well field". Several WHPA delineation methods exist, differing in their degree of complexity and precision. Naturally, the integration of more geological and hydrogeological characteristics of the study area increases the accuracy of any given method. These methods include [1]:

- Arbitrary fixed radius
- Calculated fixed radius
- Simplified variable shapes
- Flow system mapping with uniform flow equation
- Analytical flow/particle-tracking tools
- Numerical flow/transport models

Since the early 1990s, many WHPA studies have been completed, some of which have stressed the necessity of integrating various hydrogeological characteristics into the delineation methods. A comparative review of WHPA delineation methods is provided in Paradis *et al*. [31]. From a practical perspective, the most appropriate method for WHPA delineation should be one that simplifies the flow system as much as possible while preserving its geologic and hydrologic characteristics [32,33].

The WHPA can be referred to as the zone of contribution, *i.e.*, the two-dimensional (2D) projection to the land surface of the aquifer volume containing all the groundwater that may flow toward a pumping well over an infinite time period. The zone of travel is defined within the zone of contribution and can be described as an isochrone indicating the transfer time—time of travel (TOT)—necessary for water or a conservative contaminant to reach the well from that location. The TOT will depend on the pumping rates and the aquifer characteristics such as transmissivity, hydraulic gradient, porosity and aquifer thickness. The level of aquifer vulnerability should address the selection of TOT for identification of WHPAs. In fact, water wells exploiting low-vulnerability aquifers can be protected by limited WHPAs (low TOT values) without compromising the level of the protection. Conversely, wells tapping vulnerable aquifers require extended WHPAs (high TOT) to ensure adequate safeguards are in place. The proper evaluation of aquifer vulnerability and the selection of a suitable TOT for WHPAs is thus very important to avoid over- or underestimating the level of land protection that is required. This selection is especially significant in agricultural areas where fertilizers, agrochemicals and pesticides are intensively utilized [34]. Therefore an effective comprehensive protection strategy for groundwater quality should integrate the assessment of the aquifer vulnerability with the WHPAs in a suitable way.

As early as 1980 the European Union developed a directive concerning the preservation of water quality for human consumption [35]. In concordance with that directive, the Italian Government, in the 1980s, developed a national regulatory framework for the protection of groundwater resources, including the need for WHPA delineation for water supplies (wells, springs and surface water acquisition points) [36]. Subsequent European directives designed to protect the subsurface environment from unacceptable contamination [37-40] were progressively incorporated into the Italian national legislative framework [41-43]. These new legislative directives introduced novel procedures and scientific aspects to groundwater protection policies. Based on these newly-defined perspectives, some Italian regional governments implemented specific groundwater resources programs to safeguard water supplies within their territory. Through specific regulations [44] the Piemonte region environmental authority (NW Italy) tried to ensure proper alignment between aquifer vulnerability and the WHPAs delineation.

Actions taken to preserve and protect groundwater resources within a WHPA, particularly those encompassing limitations on certain agriculture practices, must be approached in a collaborative fashion with local agriculture stakeholders, and must take into account available best practices and supporting scientific data.

In this paper we tested 1) the current comprehensive technical framework for individual WHPAs in the Piemonte region on a representative case study. We highlighted some critical and 2) we proposed a limited review of the adopted methodology.

2. Methods

2.1. Identifying WHPAs in the Piemonte Region (NW Italy): Techniques and Regulations

As implemented in the Piemonte region [44], a WHPA consists of three different decreasing protection levels situated at increasing distances, respectively, from the well (**Table 1**).

The WHPA has to be defined through a procedure based on existing information and specific surveys. The regulations procedure individuating the WHPA requires an initial geological and hydrogeological general invest-tigation of the area. It is followed by evaluation of aquifer vulnerability, assessment of the aquifer hydrodynamic parameters by means of appropriate pumping tests, calculation of the isochrones through analytical or numerical models and, finally, an inventory of activities that have the potential for causing contamination within the WHPAs. These data allow for delineation of the WHPA and definition of the land use management plan within the area. Once defined, land use restrictions are controlled by the water supply company managing the well in cooperation with the regional environmental authority.

The WHPA is usually divided into two sub-areas, namely the inner protection zone (IPZ) and the outer protection zone (OPZ). The IPZ is always individuated by the 60-d isochrone, while the TOT that identifies the OPZ depends on the vulnerability of the exploited aquifer. There are four generally accepted vulnerability categories: Very High, High, Medium and Low. For low aquifer vulnerability the OPZ must be calculated using the 180-d isochrone; the remaining vulnerability categories are determined by utilizing the 365-d isochrone. It should be noted that regulations do not provide any specifications about the methodology for assessing aquifer vulnerability. The suitable method must be decided on a case by case basis. For WHPAs overlaying agricultural areas, a specific fertilizer and phytosanitary management plan must be developed which integrates the general land use management plan. It should ensure the safe application of fertilizers, agrochemicals and pesticides, taking into account the attenuation capacity of the soil cover with respect to groundwater pollution. Determination of this protection capacity must consider at least the following soil parameters: texture, skeleton, soil depth and cracks. These soil data are generally available for the general region. Soil protection capacity has been evaluated by IPLA [45] and is currently available digitally via the internet. In areas for which historical data are not available, a specific site evaluation should be developed (minimum 1 soil profile/2 ha of WHPA). Four soil protection capacity categories have been established: very high, high, medium and low. By combining the aquifer vulnerability and the soil protection capacity within the WHPA in a suitable manner (**Table 2**), four levels of land use restrictions are identified and the corresponding agricultural land use limitations have been specifically defined (**Table 3**).

2.2. Test Site: The Castagnole Well

The procedure for developing a specific WHPA, as described in Section 2.1, was tested on a water well supply ing the Castagnole municipality, located 20 km south of the Turin urban area (see **Figure 1**), which is the capital of the Piemonte region (well geographical coordinates are 45˚54'01.93"N, 7˚33'23.55"E). The elevation of the site is 244 m asl. The tested well is 88 m deep. The di-

Table 1. WHPA differentiation and permitted land uses according to the Italian water regulations (modified after [42]).

WHPA zone	Individuating criteria	Land uses
Total protection zone (TPZ)	Fixed radius (10 m minimum)	None. This zone should be fully preserved, impermeabilized, enclosed, and with limited access for authorized personnel only.
Inner protection zone (IPZ)	Time of travel (60-d isochrone)	Strongly limited. No excavation and subsurface work is allowed. Hazardous activities should be re-located if they are present. New buildings construction is prohibited.
Outer protection zone (OPZ)	Time of travel (180-d and 365-d isochrones for low vulnerability aquifers or medium, high and very high vulnerability aquifers, respectively)	Limited. Only minor anthropogenic activities are allowed, and safeguard measures against groundwater pollution are necessary for existing and new buildings.

Table 2. Identification of the land use protection levels required within the WHPA in the agricultural areas by the association of aquifer vulnerability and soil protection capacity. See Table 3 for details concerning authorized land uses and agricultural practices (modified after [44]).

	Soil protection capacity (related to groundwater pollution)	
	Very high and high	Medium and low
Low aquifer vulnerability	Level 4 (minimum protection)	Level 3
Medium aquifer vulnerability	Level 3	Level 2
High and very high aquifer vulnerability	Level 2	Level 1 (maximum protection)

Table 3. Authorized land uses and agricultural practices within the WHPAs as indicated by the protection levels derived by the association of aquifer vulnerability and soil protection capacity (see Table 2) (simplified and modified after [44]).

Water supply protection level	In the inner protection zone (60 d isochrone)	In the outer protection zone (180 d or 365 d isochrone)
Level 1 (maximum protection)	Pasture, fertilizers and phytosanitary products are fully prohibited	Fertilizer balance plan is mandatory. Nitrogen effluent discharges are limited below yearly 170 Kg/ha maximum value. Phytosanitary products are authorized under European regulations for organic farming [46]
Level 2	Fertilizer balance plan is mandatory. Nitrogen effluent discharges must be less than the maximum annual value of 170 Kg/ha. Phytosanitary products are authorized under European regulations for organic farming [46]	Same as the IPZ. A wider range of phytosanitary products and weed practices can be allowed on a case by case basis under specific conditions and regulations defined by the public surveillance authority.
Level 3	Fertilizer balance plan is mandatory. Nitrogen effluent discharges must be less than the maximum annual value of 170 Kg/ha. Phytosanitary products are authorized under European regulations for organic farming [46]. A wider range of phytosanitary products and weed practices can be allowed on a case by case under specific conditions and regulations defined by the public surveillance authority.	Same as the IPZ
Level 4 (minimum protection)	Fertilizers balance plan is mandatory. Nitrogen effluent discharges must be less than the maximum annual value of 170 Kg/ha. Phytosanitary products and weed practices are allowed on a case by case basis under specific conditions and regulations defined by the public surveillance authority.	Same as the IPZ

ameter of the casing is 650 mm. The well is cemented from the surface to a depth of 28 m. Three screened sections are located in producing sand-gravel layers between depths of 46 - 50 m, 67 - 69 m and 78 - 81 m. The undisturbed water level of the confined aquifer (without any pumping) is at 242 m asl on the well vertical (**Figure 2**). The withdrawn groundwater is analyzed by regional sanitary authorities twice a month to control the chemical and bacteriological parameters according with Italian regulation for water intended for human consumption [36, 38-40]. Since 1990 no organic or inorganic pollution was detected.

2.3. Geology Site Description

The Castagnole area is mainly developed on the outwash plain comprised of several glaciofluvial coalescing fans connected to the Pleistocene-Holocene expansion phases east of the Alpine glaciers. The substrate of the outwash plain outcrop corresponds to the Torino Hill and consists of a Cenozoic terrigenous marine succession deposited in an episutural basin [47] (see Unit 3 in **Figure 1**). As a result of a complex Pliocene-Holocene evolution characterized by the deposition of continental sediments related to the dynamic evolution of the Plio-Pleistocene "Villafranchian" glaciolacustrine facies [48] and the Pleistocene-Holocene expansion phases of the main Alpine glaciers, the geological setting of the plain is characterized by a strong geographical anisotropy.

The hydrogeological setting can be described with a high degree of confidence due to the large number of wells drilled in the plains area [49]. Downhole log data in the study area indicate the presence of two lithologic zones with distinct hydraulic properties. On the well vertical it is possible to identify Units 1 and 2 (**Figure 2**), which are described in greater detail.

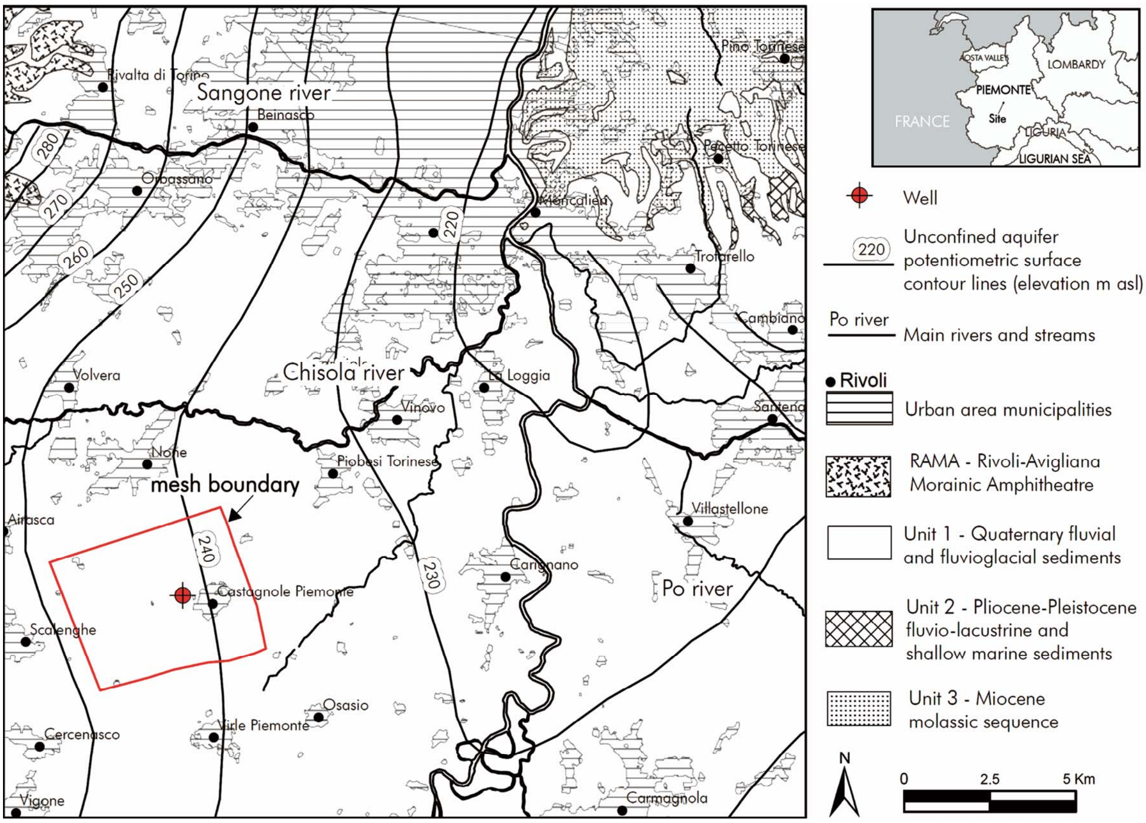

Figure 1. Hydrogeological map of the southern Turin area and location of the study site (modified after [50]).

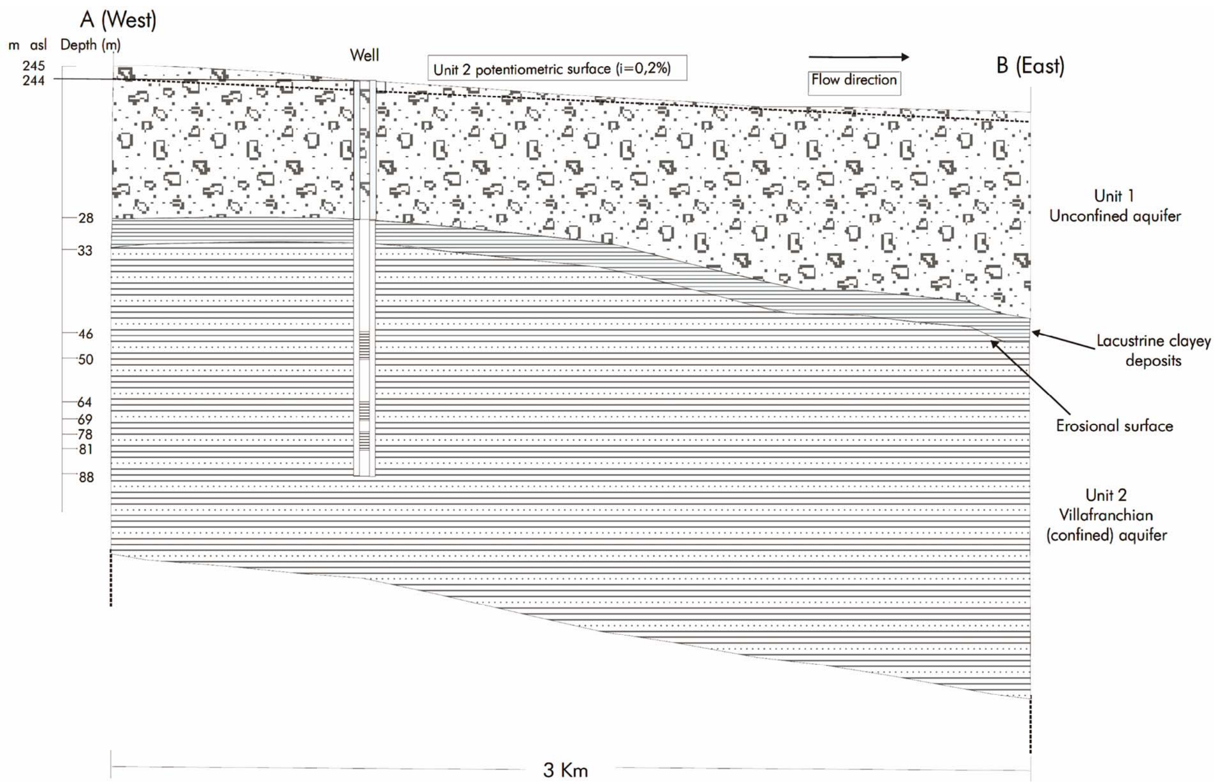

Figure 2. Schematic hydrogeological cross section of the study site (see Figure 3 for location). i: gradient of confined aquifer potentiometric surface.

Unit 1—(Middle Pleistocene-Holocene; from the surface to a depth of 28 m). Continental alluvial cover is composed mainly of coarse gravel and sandy sediments (locally cemented) derived from alluvial fans aggraded by the Alpine rivers flowing downgradient to the east. At the base of the unit there are clayey lacustrine deposits (ca. 4 - 5 m thick) that extend over the entire area and act as a confining layer between Units 1 and 2. The base of Unit 1 (erosional surface) dips gently (0.5%) to the east and overlays Unit 2.

Unit 2—(Early Pliocene-Middle Pleistocene; from a depth of 33 m). Fluvio-lacustrine facies usually referred to as the "Villafranchian", consisting of fine-grained sediments (sand, silt and clay with interbedded gravel) divided into several sedimentary bodies. Other portions of the plain highlight the heteropic relationships with sediments originally deposited in a shallow marine environment and traditionally defined as Sabbie di Asti and/or Argille di Lugagnano. They are mainly composed of fossiliferous sand-clay layers with subordinate fine gravel and coarse, sandy marine layers, or by quartz-micaceous sands with no evidence of fossils. The top of Unit 2 has been eroded away and covered by the lacustrine facies and alluvial deposits of Unit 1.

2.4. Hydrodynamic Characterization of the Aquifers

In order to numerically model groundwater flow, an accurate characterization of the site's hydrogeological properties, groundwater flow direction and hydraulic gradient (the potentiometric surface), and the hydrodynamic properties (transmissivity, hydraulic conductivity, storage coefficient) is required. The unconfined aquifer that extends over the entire plain, including the study site location, is hydraulically connected to the main surface water drainage network (*i.e.* Chisola River and Po River). The potentiometric surface, 2 m below ground level, shows a W-to-E gradient of 0.2%. The saturated thickness of the unconfined aquifer at the site is about 26 m.

In order to characterize the hydrogeological properties of the aquifer in Unit 1, an appropriate step drawdown test was initially performed on a 30 m deep well located less than 1.5 km from the site. The test data yielded a transmissivity (T_1) of 7.3×10^{-3} m^2/s. The hydraulic conductivity ($K_1 = 3.65 \times 10^{-4}$ m/s) was calculated assuming an average saturated thickness of 20 m. On the basis of a constant-rate pumping test, the storativity (S_1) was assumed to be 0.20.

A confined aquifer system occurs in Unit 2. The available subsurface data indicate that the direction of groundwater flow and the potentiometric gradient (0.2%) in the Unit 2 aquifer system are similar to those in the unconfined aquifer of Unit 1. In the productive well the potentiometric surface of the confined aquifer stabilizes 31 m above the top of Unit 2, just 2 m below the ground's surface, which is roughly equivalent to the value measured in the overlaying Unit 1. The hydraulic transmissivity (T_2) of the Unit 2 aquifer system (7.52×10^{-3} m^2/s) was determined by means of a specific step-drawdown test in the studied well. The storativity (S_2) was calculated as 10.6×10^{-4}.

2.5. Modelling Study of the Aquifers

The modelling study was performed using the finite-element FEFLOW® package developed by Diersch [51]. A conceptual model with three layers was simulated using physical properties appropriate to the hydrogeology of the formation. Layer 1 represented the unconfined aquifer in Unit 1, Layer 2 corresponded to the 5 m thick impermeable clay layer at the base of this aquifer and Layer 3 represented the confined aquifer system of Unit 2. The distribution of the different layers in the model area was determined from topographic elevation data for the different geological units as listed in the regional authority database [49].

A plan view of the area covered by the computational grid (about 27.82 million m^2; 14,133 elements and 9800 nodes) is shown in **Figure 3**. The ground surface ranges from 253 m at the NW mesh vertex boundary to 240 m at the SE vertex. The horizontal dimensions of the model grid are 5238 m (SW-NE) and 4334 m (NW-SE). The average mesh spacing in the modelling domain is 70 m, which was refined to 8 m in the central area close to the well to provide enhanced estimation of potentiometric disturbed surface and the wellhead protection area isochrones. The north and south boundaries are set as no-flow boundaries. The east and west boundaries are constant-head boundaries (Dirichlet conditions). These levels were determined by initially calibrating the model against the steady-state groundwater heads obtained from a potentiometric surface map [50] and a specific survey of the area. An assumption of the model was that the system was closed to fluid flow at bottom (Layer 3 is set 200 m thick). The system has only recharge from rainfall and the ground surface is set as a prescribed flux boundary recharged by rainfall. An infiltration rate of 5.7×10^{-4} m/day is used in the model, which is equivalent to 25% [52] of the annual rainfall of 834 mm.

The simulations were run assuming steady-state conditions for groundwater flow. The withdrawal rate on the tested well (12 L/s) corresponds to the abstraction peak conditions. In reality, such conditions never actually occur because of variable (transient) water demand and the presence of a groundwater storage tank. Therefore, the actual impacts to the aquifer in terms of potentiometric surface changes due to well pumping will be less than

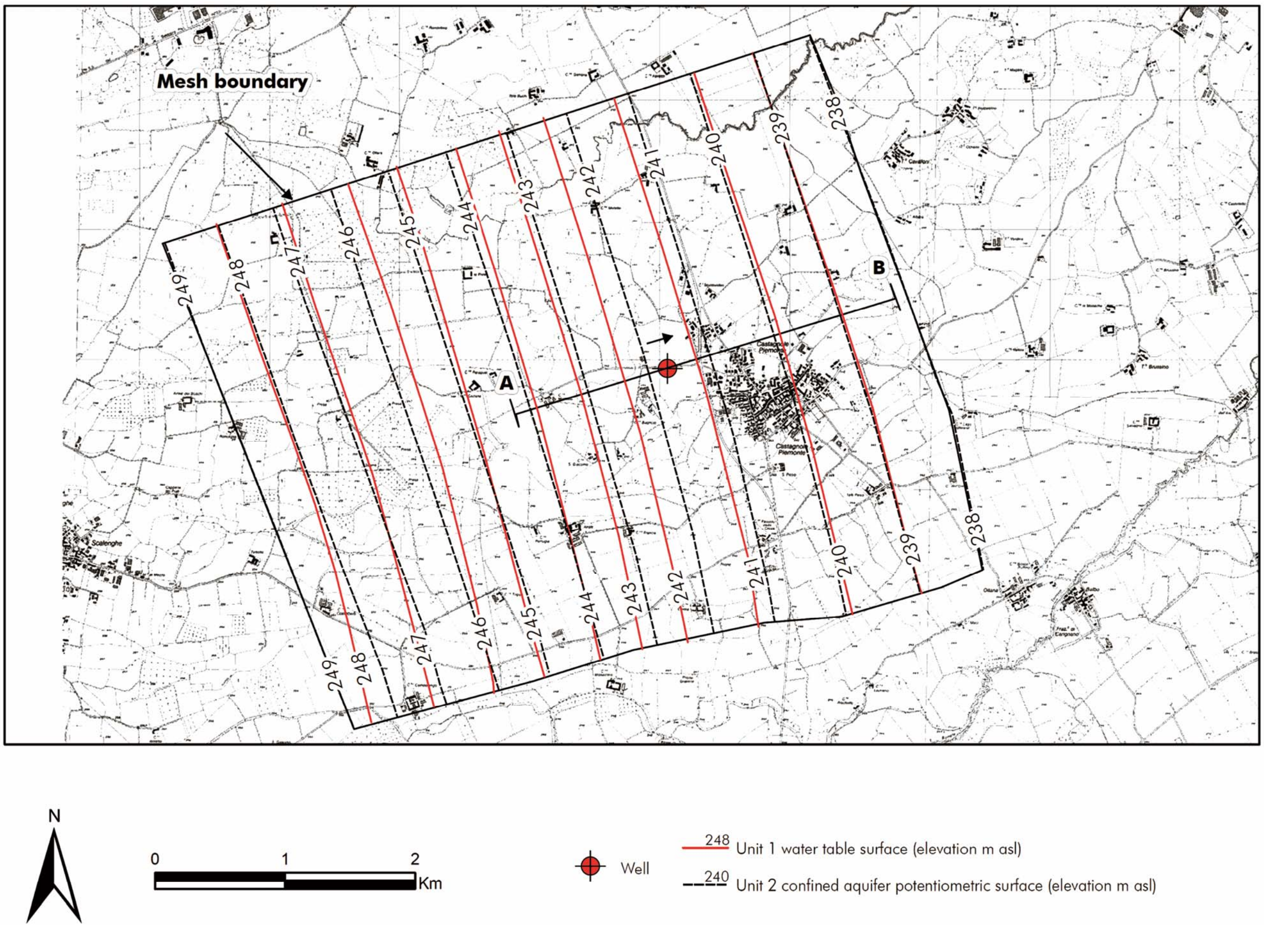

Figure 3. Plan view of the site and modelling domain overlain onto the topographical map. Potentiometric surface for the Unit 1 unconfined aquifer (continuous lines) and Unit 2 confined aquifer (dashed lines) under undisturbed conditions (in meters above mean sea level; m asl). Contour spacing: 1 m. Shown is the location of the cross section given in Figure 2.

those computed by the model. As a result, the WHPAs individuated by means of the calculated isochrones will be slightly overestimated and thus conservative relative to aquifer protection.

2.6. Aquifer Vulnerability and Soil Protection Capacity

Qualitatively, the unconfined aquifer accessed in Unit 1 is considered highly vulnerable to pollution because of its shallow depth and the direct connection with the surface water drainage network. The confined aquifer in Unit 2, on the other hand, is only moderately vulnerable to pollution, due both to depth (on average, the top of Unit 2 is situated at 30 - 35 m) and to several clay interlayers subdividing the formation. Only damaged or improperly constructed wells could introduce contaminants to this system of confined aquifers. To identify the suitable isochrone values delineating the WHPAs, aquifer vulnerability must be numerically defined. To achieve this, the modified GOD method [4] was selected as a suitable method. In fact, more sophisticated vulnerability assessment methods such as DRASTIC or SINTACS are not suitable because they already include in the aquifer vulnerability assessment the soil parameters affecting the protection capacity. Therefore the required protection level identified by means of the procedure described in the **Table 2** could be erroneously evaluated. The GOD technique assigns numerical values between 0 and 1 to the Groundwater confinement level (*i.e.* G value), the lithological characteristics and the degree of consolidation of the vadose zone or confining layers (*i.e.* Overlying strata or O value) and depth to the groundwater table for unconfined aquifers, or to the strike for confined aquifers (*i.e.* Depth or D value). No soil parameter is considered. The resulting GOD value that identifies aquifer vulnerability is calculated by the multiplication of these three parameters. Due to the relative homogeneity of the aquifer over the entire modelling domain, the GOD value has been computed on the well vertical. At the test site, the Unit 2 aquifer has a G value of 0.2 (confined aquifer), an O value of 0.8 (alluvial and fluvio-glacial sands and gravels) and a D value of 0.7 (depth of 20 - 50 m), resulting in a GOD value of 0.112, which indicates low vulnerability. Therefore, the OPZ can be identified by the 180-d isochrone.

The modelling domain overlays different soil units characterized by an appropriate level of protection against groundwater pollution. **Figure 4** and **Table 4** highlight the result of a GIS analysis of the soil units over the whole modelling domain.

3. Results and Discussion

The calculated WHPA for the test site is delineated in **Figure 5**. The 60-d isochrone (IPZ) covers about 4334

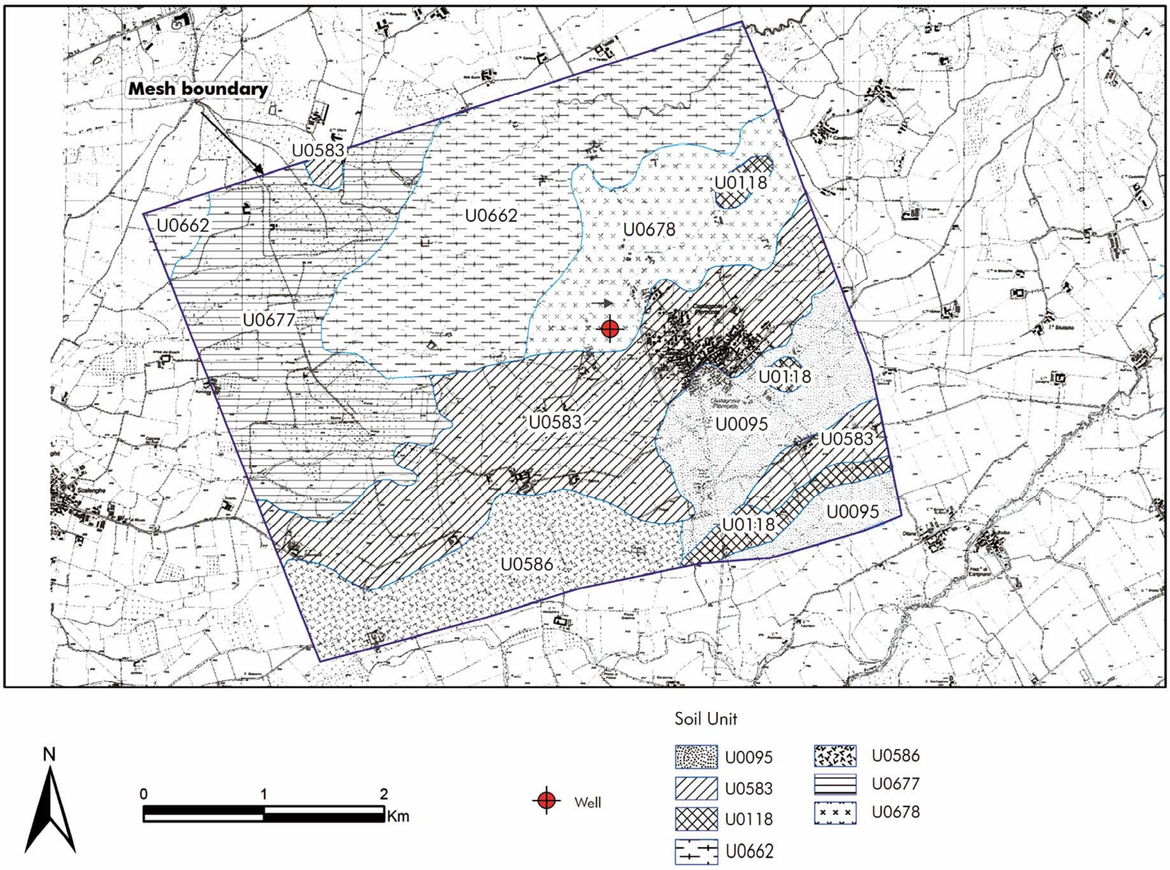

Figure 4. Soil units in the modelling domain (modified after [45]). See Table 4 for description.

Table 4. Soil units in the modelling domain and corresponding degree of soil protection capacity (Simplified and modified after [45]).

Soil unit	Soil classification	Area (m^2 and %) in the modelling domain	Soil protection capacity (related to groundwater pollution)
U0677	Typic endoaquept, coarse-loamy, mixed, nonacid, mesic	4,248,990 (15.3%)	Low
U0095	Dystric fluventic eutrudept, coarse-loamy, mixed, nonacid, mesic	8,439,617 (30.3%)	Very high
U0118	Psammentic haplustalf, coarse-loamy, mixed, nonacid, mesic	580,037 (2.1%)	Very high
U0583	Typic endoaquept, coarse-loamy, mixed, nonacid, mesic (70% UTS—Unit territorial surface) Aquic dystric eutrudept, coarse-loamy, mixed, nonacid, mesic (30% UTS)	5,149,067 (18.5%)	Medium
U0586	Dystric eutrudept, coarse-loamy, mixed, nonacid, mesic (60% UTS) Aquic dystric eutrudept, coarse-loamy, mixed, nonacid, mesic (40% UTS)	2,139,180 (7.7%)	Very high
U0662	Typic endoaquept, coarse-loamy, mixed, nonacid, mesic (70% UTS) Aeric endoaquept, coarse-loamy, mixed, nonacid, mesic (30% UTS)	4,996,325 (18.0%)	Medium
U0678	Fluventic dystrudept, coarse-loamy, mixed, acid, mesic	2,269,850 (8.2%)	Medium

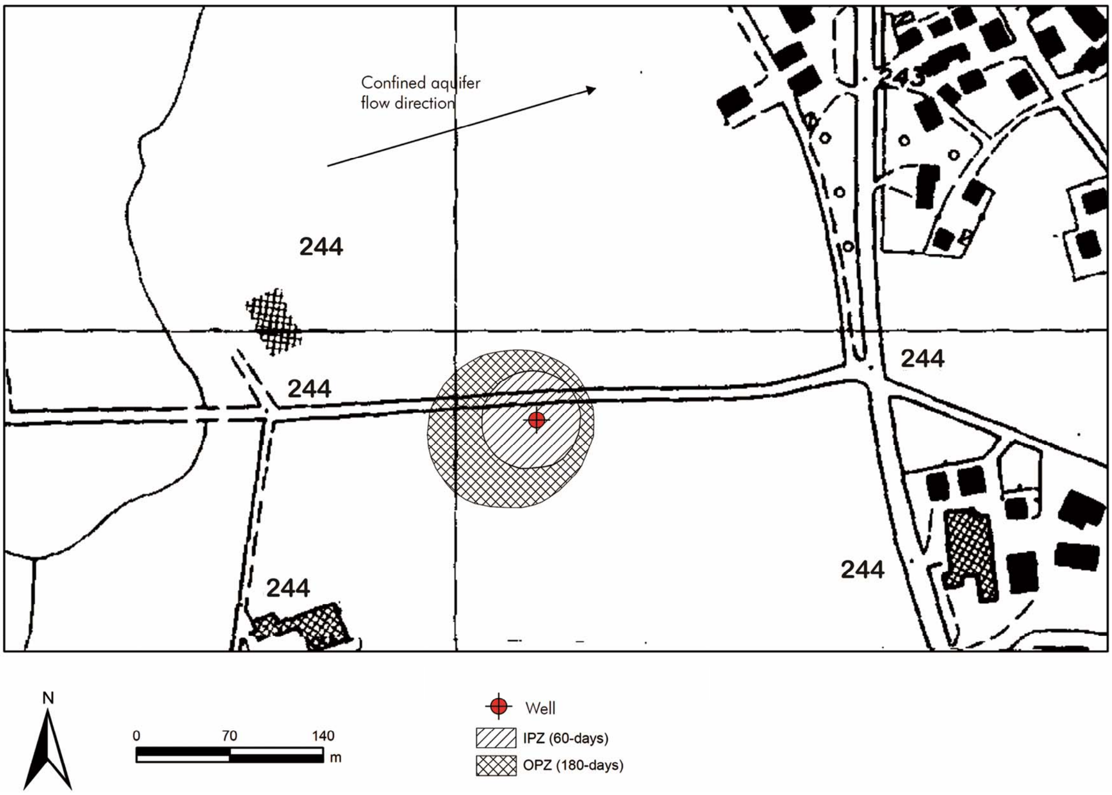

Figure 5. Wellhead protection areas identified by means of isochrones.

m^2, while the 180-d isochrone (OPZ) covers 11,734 m^2. Considering both the aquifer vulnerability (low) and the medium soil protection capacity of the soil unit overlaid by the WHPA (U0678), the corresponding level of protection was calculated as Level 3 (**Table 2**). Given this level of protection, certain restrictions on agricultural practices must be observed (**Table 3**). The calculated WHPA includes a minor road and therefore additional precautionary measures should be developed in order to prevent contaminant migration from the surface due both to accidental spills and infiltration of dust and water running off the road surface that might carry contaminants (e.g., petroleum hydrocarbons, metals). Safety measures applied along the motorway would include:

a) Using catchments and channelling to collect rain water that contacts the road surface, or any other fluid that is accidentally released;

b) Transport of collected fluids to monitoring basins and, after verifying the absence of contamination, sending those liquids for final disposal (water drains or streams). If contaminants are found to be present at concentrations exceeding established criteria, standards or benchmarks, the fluids will be sent to treatment plants;

c) The final destination of clean water should be outside, and downgradient, of the WHPA.

The specific measures that are instituted to safeguard the WHPA should be managed by the regional environmental authority in cooperation with the farmers, the water well managing company and the road maintenance company.

4. Conclusions

An effective and economically-sustainable land management strategy to protect subsurface water resources from anthropogenic pollution must combine general safeguards applied to the whole aquifer recharge area with specific local land use restrictions in the proximity of the abstraction point (*i.e.* WHPAs). The first component, *i.e.* general protection strategies, can be derived through an extensive, broad-scale investigation, taking into account aquifer vulnerability, while data for the second component can be obtained using site-specific investigations within a narrowly-defined area proximal to the abstracttion point. The importance of considering these two components in an integrated fashion cannot be understated. In particular, the selection of TOT for WHPA delineation is critically linked to the anticipated vulnerability of the aquifer in question.

This study has highlighted a technical approach developed in the Piemonte region, and designed to protect drinking water wells. An important aspect of this approach was the integration of broad-scale aquifer vulnerability assessment with localized WHPA delineation.

The method has been successfully tested on a community drinking water well and is both affordable and effective. However, for this method to be accepted for broad application, additional refinement is needed in certain areas. In particular, improved specifications should be provided to allow the user to more confidently select an appropriate aquifer vulnerability assessment method. The present version provides little guidance, leaving the selection to professional subjectivity and experience. However, current regulations combine the vulnerability level with soil protection capacity, thus discouraging the use of techniques that already compartmentalize soil parameters in the vulnerability assessment (e.g. DRASTIC and SINTACS). Given this constraint, only methods that consider aquifer parameters (*i.e.* GOD) seem suitable to evaluate vulnerability. Future iterations should simplify the procedure to individuate the necessary level of protection within the WHPA if soil protection capacity is directly included in the aquifer vulnerability assessment.

5. Acknowledgements

The authors thank the environmental authority of the Piemonte region for their assistance during the field survey, the modelling, and for allowing publication of the data. This study was partially supported by the Piemonte regional Government and by SMAT S.p.a. integrated water services company.

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Journal of Water Resource and Protection, **2012, 4, 686-693**
doi:10.4236/jwarp.2012.48079 Published Online August 2012 (http://www.SciRP.org/journal/jwarp)

Time Step Issue in Unit Hydrograph for Improving Runoff Prediction in Small Catchments

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Received May 20, 2012; revised June 20, 2012; accepted June 28, 2012

ABSTRACT

Unit hydrograph is a very practical tool in runoff prediction which has been used since decades ago and to date it remains useful. Unit hydrograph method is applied in Way Kuala Garuntang, an ungauged catchment in Lampung Province, Indonesia. To derive an observed unit hydrograph it requires rainfall and water level data with fine time scale which are obtained from automatic gauges. Observed unit hydrograph has an advantage that it is possible to derive it for various time steps including those with time step less than an hour. In order to get a more accurate unit hydrograph, it is necessary to derive a unit hydrograph with small time step for a small catchment such as those used in this study. The study area includes Way Kuala Garuntang and its tributaries, *i.e*. Way Simpur, Way Awi with areas are 60.52 km^2, 3.691 km^2, and 9.846 km^2 respectively. The results of this study highlight the importance of time step selection on unit hydrograph, which are shown to have a significant impact on the resulting unit hydrograph's variables such as peak discharge and time to peak.

Keywords: Unit Hydrograph; Time Step; Peak Discharge; Time to Peak

1. Introduction

The development of hydrology model in runoff prediction is very advance, in which there are several methods that can be used in runoff prediction in ungauged basin. Especially with the existence of PUB (Prediction in Ungauged Basin) [1,2], there are several supporting tools and methods which makes prediction possible in such catchments. The choice of methods and tools are based on available data in that region. The limitation of fine data such as data from radar, leave little choice to carry out prediction in some catchments. As many other catchments in many parts of the world, Way Kuala Garuntang is an ungauged catchment. There was no runoff measurements recorded before. This increasingly grows into significant matter as floods occur more frequently in this region recently [3]. It is believed that one of the best options to do runoff prediction is by taking runoff measurements [4]. Therefore this study deals with instrumenting this ungauged catchment to gain important information and carry out necessary analysis, as well as predicting runoff using observed unit hydrograph (UH) method.

Despite its conservative method, the unit hydrograph approach to rainfall-runoff modelling remains a very useful and practical approach to deal with operational hydrological forecasting [5]. In this case UH model structure is assumed to be appropriate to represent catchment behavior by assuming two separately acting functions, *i.e*. the production and the transfer functions [5]. When a certain amount of rainfall reaches the ground, some will loss due to infiltration or others, and there remains a reduced part called the effective rainfall which then transformed into direct runoff. This runoff is then delayed and transferred to the outlet by various routing mechanisms. Unit hydrograph is a linear transfer function that represents those mechanisms with an assumption that the mechanisms behave similarly from event to event.

The choice of using observed unit hydrograph, because this method is capable in predicting time to peak of runoff more accurately as this method can do the computation for time step less than one hour. This obviously an advantage of using observed unit hydrograph compared to synthetic unit hydrographs (SUH) such as Nakayasu, GAMA I and Snyder and other kind of SUH which have time step of hour [6-9]. Time step becomes an issue here as the selected catchments are small catchments less than 100 km^2 of area, which may need short time concentration for the flow to propagate to the outlet. Hence, this study aims to investigate the impact of time step selection in resulting unit hydrograph.

2. Methodology

2.1. Description of Study Area

The work took place in Way Kuala Garuntang catchment including its two sub-catchments, Way Simpur and Way Awi as presented in **Figure 1**). Way Simpur and Way Awi are two neighbouring sub-catchments, while those two sub-catchments are cascading to Way Kuala Garuntang catchment. The catchments located in Lampung Province, Indonesia. The area of Way Simpur, Way Awi and Way Kuala Garuntang catchments are 3.691 km^2, 9.846 km^2 and 60.52 km^2 respectively. Three runoff measurements were carried out, two in the tributaries *i.e.* Way Simpur, Way Awi and one in the downstream of Way Kuala Garuntang River. There is no runoff measurements in these catchments before. In order to construct an observed unit hydrograph, several things need to be prepared. Three automatic water level recorder (AWLR) needs to be installed in those locations, one for each point. There is one tipping bucket raingauge located in Way Kuala Garuntang catchment and the rainfall data obtained from this raingauge is used to calculate the unit hydrographs for each catchment.

The topography of upstream part of the catchment is hilly and the slope is flatter toward downstream catchment. Way Simpur and Way Awi, they are neighbouring catchments but the catchment characteristic is slightly different. Way Awi catchment is highly populated where their house is located close to each other, therefore most rainfall is transformed into runoff. During intense storm event, flood comes quickly, but then releases in short period of time. The channel width varies, where the width at the location study is 8 meters. Way Simpur is also a rural catchment and highly populated. The slight difference is during intense storm event, flood comes quickly but releases slight longer period of time compared to release time in Way Awi. The channel width at the location of study in Way Simpur is 7.5 meters and in Way Kuala Garuntang the river has 9 meters width.

2.2. Rating Curves

Measurements of discharges and water levels at those three points were carried out during wet season October 2009-April 2010. Velocities were measured using current meter and water levels were observed using peilschaal attached on the river bank. Based on those measurements, a rating curve for each point is determined and results are presented in **Figure 2**. Rating curve for Way Simpur (**Figure 2(a)**) shows the increase of water levels resulted in lower increase of discharges compared to that for Way Awi (**Figure 2(b)**), which is presented by sharper slope of Way Awi's rating curve. Please note that the scales of rating curves for both Way Simpur and Way Awi are the same, but differ from those of rating curve for Way Kuala Garuntang. Rating curve for Way Kuala Garuntang (**Figure 2(c)**) shows the extensive range of discharges, which in the measurement for 1.2 m water level causes discharge of about 25 m^3/s.

2.3. Effective Rainfall

This study used a classic ϕ_{index} approach to determine the effective rainfalls. Although there are quite a number of approaches used to determine the effective rainfalls such

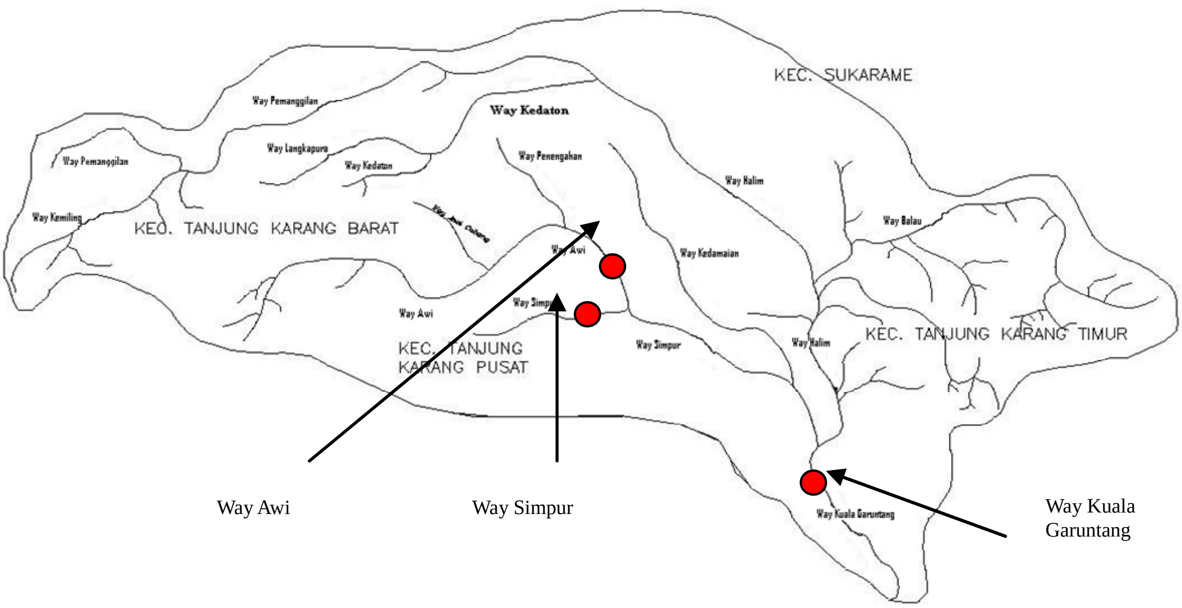


Figure 1. Way Simpur, Way Awi and Kuala Garuntang catchments and the locations of the runoff gauges.

as Green Ampt infiltration and others, ϕ_{index} approach is still widely used due to its simplicity. The approach produces a series of excess rainfall (PE) or effective rainfall values from the observed gross rainfall (PG) values. The only constraint is to fit the overall so-called "stormflow" volume which is thought to have become runoff. Subsequently, the computed series of excess precipitation and the observed discharge (Q) are used to calibrate the UH in a 'known input known output' context [5].

The equation used to calculate ϕ_{index} is shown in Equations 1 and 2, where runoff depth (Q_{DR}) is a result of volume of direct runoff (V_{DR}) divided by catchment area (A). Thus ϕ_{index} is the difference between gross rainfall (P) and runoff depth (Q_{DR}) divided by time (t). Excess precipitation or effective rainfalls are obtained as gross precipitation subtracted by ϕ_{index}.

$$Q_{DR} = \frac{V_{DR}}{A} \quad (1)$$

$$\phi_{\text{index}} = \frac{P - Q_{DR}}{t} \quad (2)$$

3. Results and Discussions

3.1. Flood Events, Time Steps and ϕ_{index}

There are several flood events recorded during wet season 2009-2010, and the events are presented in Tables 1 to 3 for flood events selected for Way Simpur catchment (**Table 1**), Way Awi catchment (**Table 2**), and Way Kuala Garuntang catchment (**Table 3**). For each event, other related parameters such as rainfall depth, rainfall duration, calculated ϕ_{index} are also presented. Please note that the calculated ϕ_{index} are for three time steps, *i.e.* 10, 30 and 60 minutes.

It can be seen that the first recorded flood event was in December, although the start of wet season is in October. This happened because the first few rains were mostly infiltrated to fulfill soil moisture capacity. Furthermore, flood events presented in **Tables 1-3** are those which can be used to develop unit hydrograph. The advantage of using observed unit hydrograph to synthetic unit hydrograph such as Nakayasu, Snyder and GAMA 1, is the possibility to develop a unit hydrograph with finer time

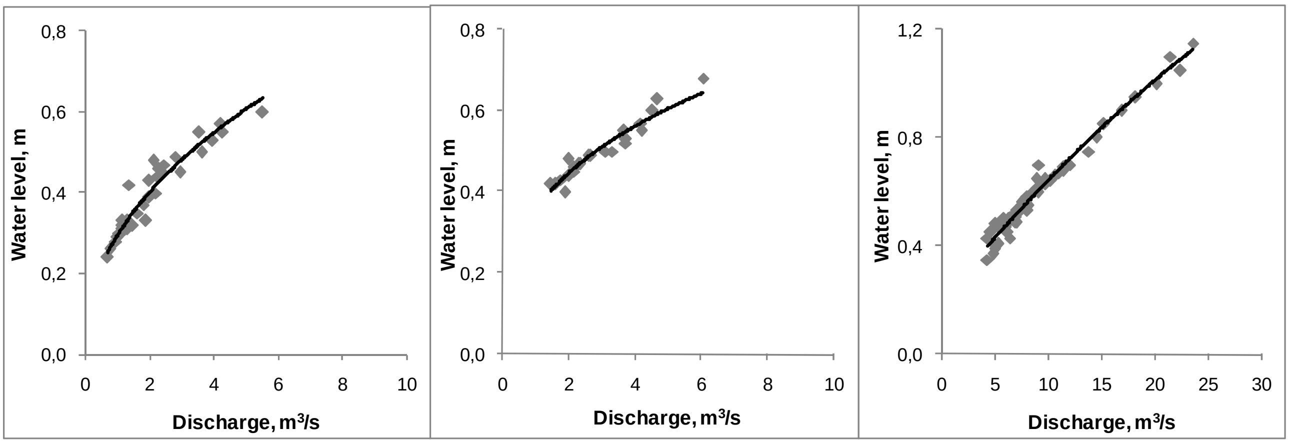


Figure 2. Rating curves for (a) Way Simpur; (b) Way Awi; and (c) Way Kuala Garuntang.

Table 1. Flood events selected for Way Simpur catchment.

No.	Date	Peak Discharge (m^3/sec)	Rainfall Depth (mm)	Rainfall Duration (hours)	ϕ_{index} Time step 1 hr	ϕ_{index} Time step 30 min	ϕ_{index} Time step 10 min
1	16-01-2010	3.453	8	1	7.617	6.012	5.544
2	17-01-2010	6.437	6	1	5.225	2.908	-
3	17-01-2010	4.050	5.6	1	4.511	3.024	2.591
4	31-01-2010	3.769	10.8	2	9.273	4.044	2.006
5	01-02-2010	20.015	22.2	2	7.622	-	-
6	04-02-2010	2.705	7.2	1	4.676	2.768	-
7	06-02-2010	1.339	2.8	1	2.554	1.854	0.757

Table 2. Flood events selected for Way Awi catchment.

No.	Date	Peak Discharge (m^3/sec)	Rainfall Depth (mm)	Rainfall Duration (hours)	ϕ_{index} Time step 1 hr	ϕ_{index} Time step 30 min	ϕ_{index} Time step 10 min
1	08-01-2010	21.693	7.4	1	6.346	4.025	2.517
2	16-01-2010	17.354	7	1	5.640	3.038	3.147
3	17-01-2010	19.043	6.2	1	2.589	5.251	4.364
4	12-02-2010	14.426	8.6	2	6.170	4.965	2.868

Table 3. Flood events selected for Way Kuala Garuntang catchment.

No.	Date	Peak Discharge (m^3/sec)	Rainfall Depth (mm)	Rainfall Duration (hours)	ϕ_{index} Time step 1 hr	ϕ_{index} Time step 30 min	ϕ_{index} Time step 10 min
1	25-12-2009	12.770	28.4	1	26.746	25.734	16.044
2	28-12-2009	31.426	10.2	1	5.315	3.717	1.259
3	31-12-2009	9.413	9.4	1	8.484	8.347	4.541
4	08-01-2010	14.111	7.4	1	5.826	1.790	4.506
5	10-01-2010	26.606	31.6	4	10.076	-	-
6	13-01-2010	9.413	8.6	2	3.372	2.381	0.982
7	14-01-2010	19.509	5.8	1	3.184	2.44	-
8	16-01-2010	21.089	8	1	4.744	3.406	3.320
9	20-01-2010	19.234	8.2	1	5.516	5.646	2.216
10	27-01-2010	48.232	44.6	2	35.135	25.128	11.431
11	28-01-2010	15.205	19.6	1	18.054	18.053	12.834
12	01-02-2010	38.442	22.2	2	18.337	10.393	-
13	04-02-2010	23.164	7.2	1	3.935	1.832	0.700
14	05-02-2010	47.388	14	3	5.594	2.351	-
15	08-03-2010	28.895	15.8	1	13.869	6.698	-
16	10-03-2010	30.537	18.8	2	7.031	-	-
17	13-03-2010	11.889	5.6	1	3.071	2.253	1.792

step, *i.e*. less than 1 hour. In this study time steps of 10, 30 and 60 minutes are used as presented in **Tables 1-3** and **Figures 4-6**.

Calculated ϕ_{index} for each event and time step are presented in the last three columns of **Tables 1-3**. The first event which is in December 25, 2009 shows large value of ϕ_{index}, which can be understood as a lot of portion of rains were infiltrated. The value of ϕ_{index} decreases for the next few events, but increases considerably for these subsequent events of 27-01-2010, 28-01-2010 and 1-02-2010 and again on 8-03-2010. Therefore, it cannot be concluded that the value of ϕ_{index} will decrease toward the peak of wet season (*i.e*. in January-March). In fact, the value of ϕ_{index} is defined in such a way that the computed series of excess precipitation suitable with the observed discharge. In contrast to the absence of trend of ϕ_{index} values in the flood events, the value of ϕ_{index} tends to decrease for smaller time step.

The results presented in **Tables 1-3** and **Figures 4-6** show that not all events which can be used to develop unit hydrographs for a certain time step can be used to develop those for smaller time steps. This may happen as the within storm rainfall pattern (distribution of rainfall depth for each time step) is more detail for smaller time step, so that for particular rainfall is not possible to get the ϕ_{index} and volume of effective rainfall which fit runoff volume. This may also due to the selected method for calculating effective rainfall which uses a linear approach rather than non-linear approach such as Green-Apmt or other methods.

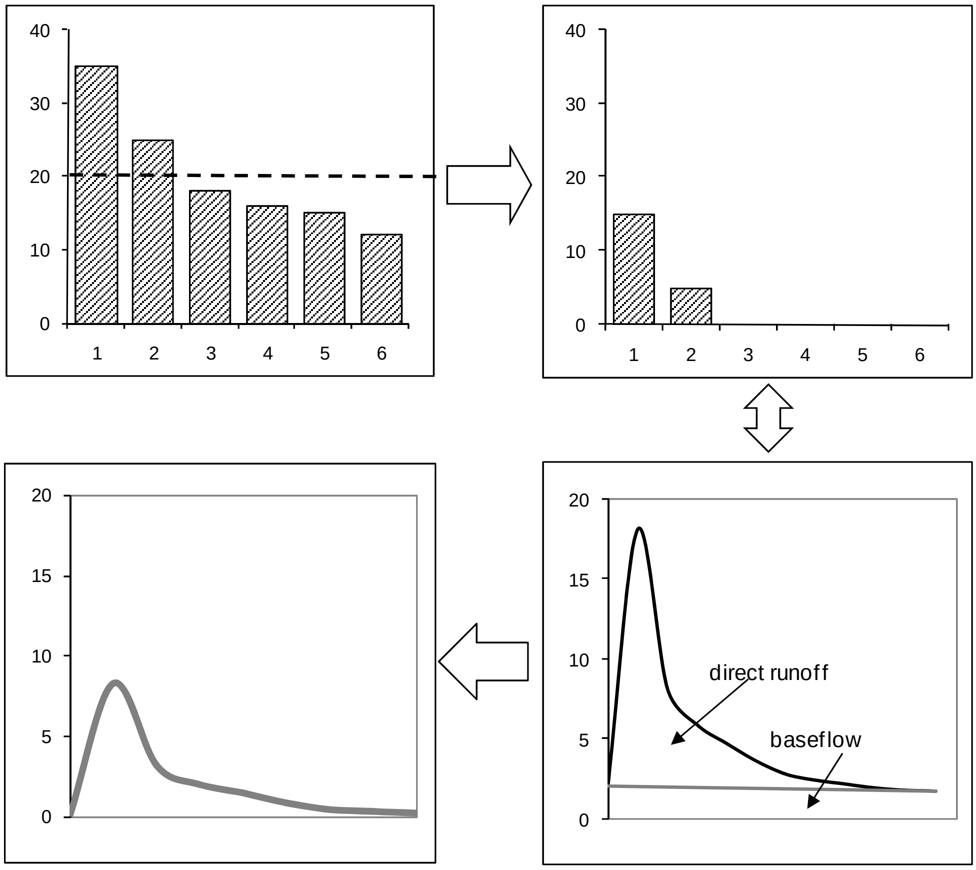
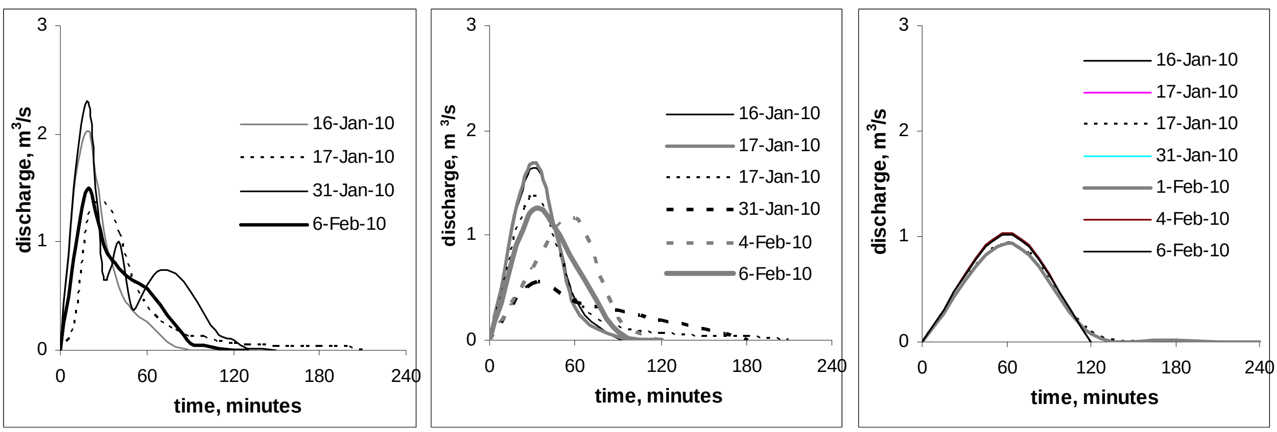


Figure 3. Effective rainfall, baseflow separation and unit hydrograph.

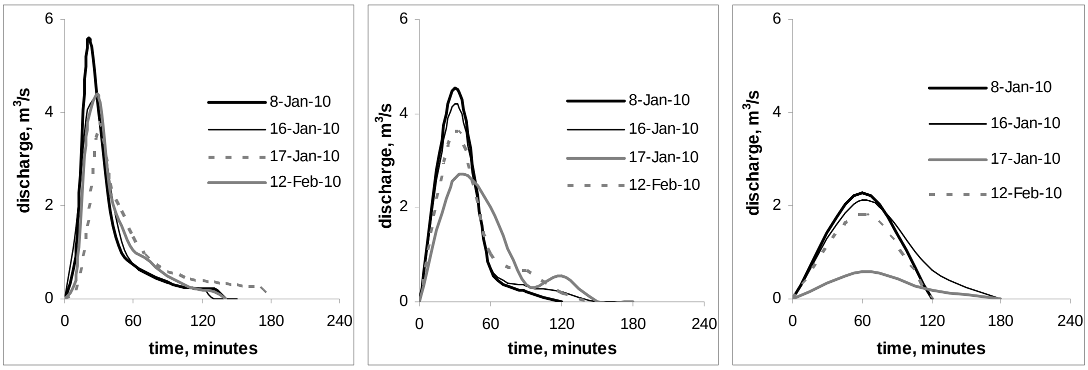


(a) (b) (c)

Figure 4. Observed unit hydrographs for Way Simpur using time steps (a) 10; (b) 30; and (c) 60 minutes.

3.2. Time Steps and Time to Peak

The unit hydrographs developed are presented in **Figures 4-6**, where **Figures 4-6** show unit hydrographs of Way Simpur, Way Awi and Way Kuala Garuntang respectively. For each catchment, the unit hydrograph is developed for time step 10, 30 and 60 minutes. The advantage of using small time step is to gain an understanding about the real time to peak for the catchment. For the case of Way Simpur (**Figure 4**), using time step of 10 minutes it can show that the average time to peak in that catchment is 20 minutes. While using time step of 30 and 60 minutes show that the averages of time to peak are 30 and 60 minutes respectively. Among those three time steps, it seems that time to peak resulted from time step of 10 minutes is the most reasonable as the catchment is



(a) (b) (c)

Figure 5. Observed unit hydrographs for Way Awi using time steps (a) 10; (b) 30; and (c) 60 minutes.

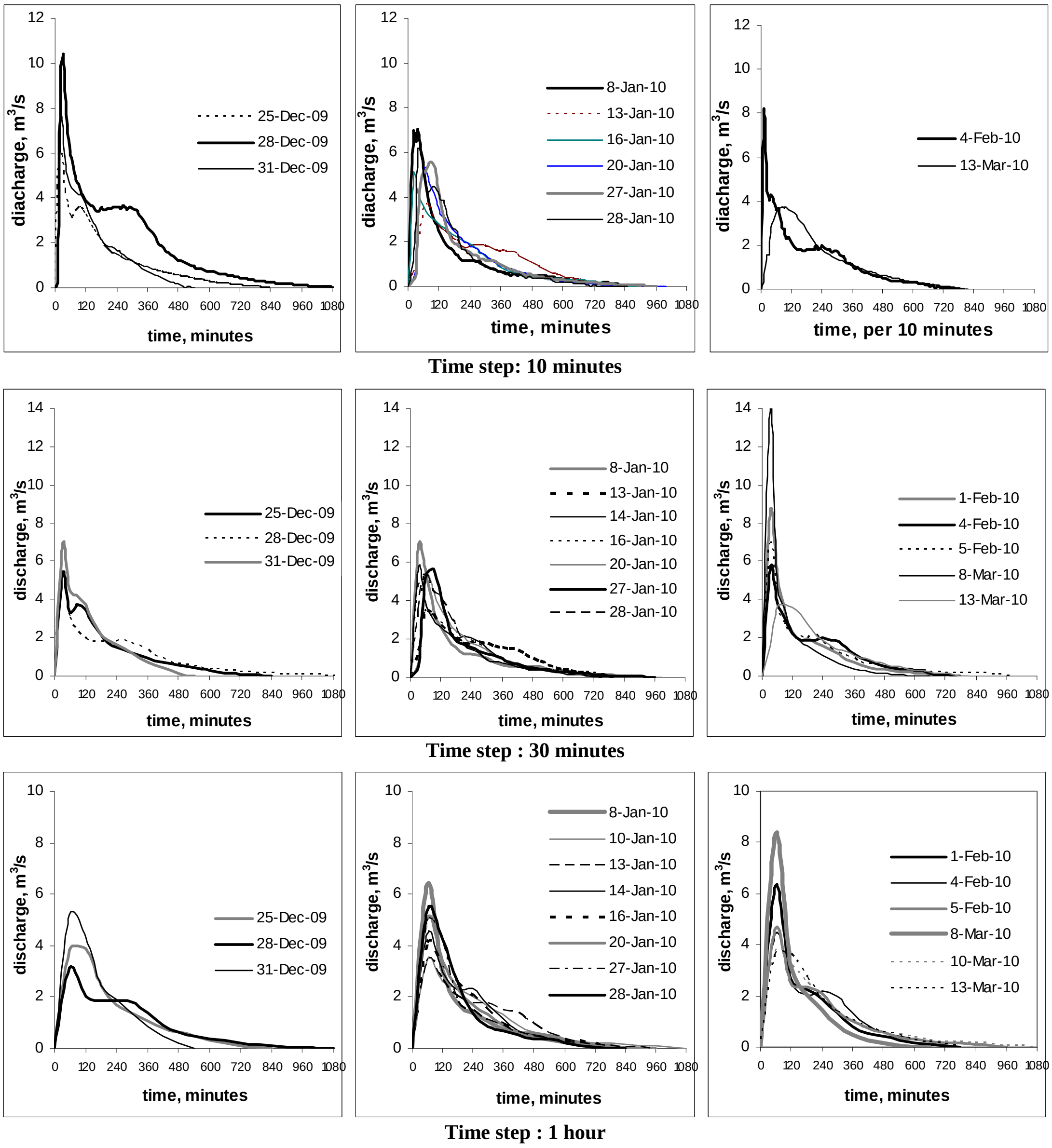


Figure 6. Observed unit hydrographs for Way Kuala Garuntang according to the months using time steps 10, 30 and 60 minutes.

considered small.

For Way Awi (**Figure 5**), using different time steps also show different results for time to peak. Using time step of 10, 30 and 60 minutes resulted in average time to peak of 30, 30 and 60 minutes respectively. Again, smaller time step gives more reasonable results in indicating time to peak.

For Way Kuala Garuntang, in addition to time step, the unit hydrographs are also made into three groups according to the months. The groups are for December, January, as well as February and March events (**Figure 6**). The average time to peak for time step of 10 minutes is 30, 60 and 60 minutes for December, January and February-March events respectively. While the average time to peak for time steps of 10 minutes for overall events is 60 minutes. For time step of 30 minutes, the average time to peak is 30, 60, 60 and 60 minutes for December, January, February-March and overall events respectively. The average time to peak for time step of 60 minutes is 60 minutes for December, January, February-March and overall events. For a larger catchment such as Way Kuala Garuntang, smaller time steps confirm time to peak as resulted from larger time step. In this case, it is predicted that the appropriate time to peak for Way Kuala Garuntang is 60 minutes.

3.3. Time Steps and Peak Discharges

In addition to time to peak, another important issue with regard to unit hydrograph is the peak discharge. For Way Simpur (**Figure 4**) peak discharges for all time steps are in the range of 0.5 - 2.4 m^3/s, where the average peak discharges for time steps 10, 30 and 60 minutes are 1.8 m^3/s, 1.2 m^3/s and 1 m^3/s respectively. Please note that in fact there are seven peak discharges in the unit hydrographs for Way Simpur using time step 60 minutes (**Figure 4(c)**), which seems to be sorted into two groups because five of them are in the range of 1.023 - 1.028 m^3/s and the other two are 0.931 and 0.937 m^3/s. Therefore it looks like there are only two curves as the outcome of seven flood events (**Figure 4(c)**).

For Way Awi (**Figure 5**) the average peak discharges for time steps 10, 30 and 60 minutes are 4.1 m^3/s, 3.7 m^3/s and 1.8 m^3/s respectively. The results show the smaller the time step the larger the peak discharge. This happen because only selected flood events which have high rainfall intensity during short time interval are able to be utilized in constructing unit hydrograph. Therefore, the rainfall intensity is larger at smaller time step which impacts on larger peak discharge.

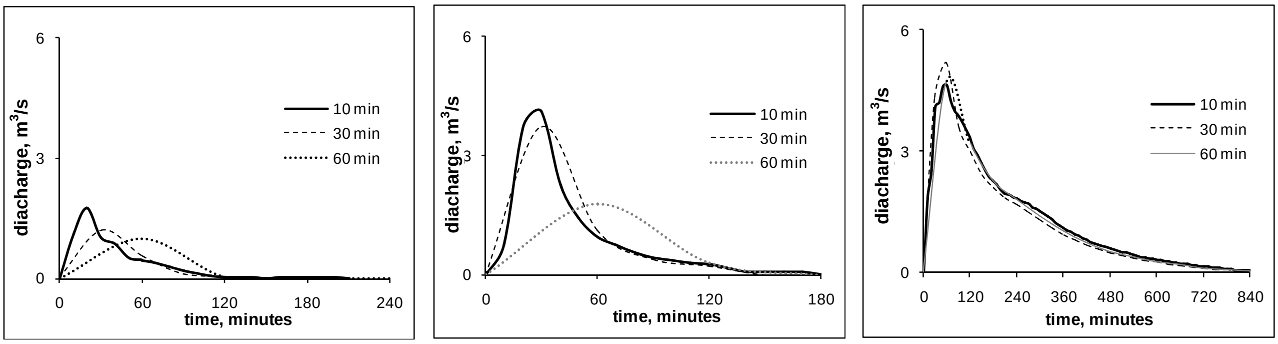
The average peak discharges for Way Kuala Garuntang for December, January and February-March events (**Figure 6**) using time step 10 minutes are 7.7 m^3/s, 4.5 m^3/s and 4.2 m^3/s respectively, using time step 30 minutes are 5.8 m^3/s, 4.7 m^3/s and 7.4 m^3/s respectively and using time step 60 minutes are 4 m^3/s, 4.6 m^3/s and 5.2 m^3/s respectively. While the trend of average peak discharges seem opposite for time step 10 minutes, the trend of those for other time steps shows there is an increase of average peak discharges toward the peak of wet season. For February-March flood events there were only two out of six flood events which were able to be utilized in unit hydrograph using time step 10 minutes, and there were only six out of eight events for January flood events could be utilized for 10 minute time step hydrograph. Meanwhile, all three flood events in December could be used for 10 minute time step hydrograph. Therefore the results from using time step 10 minutes show inconsistent trend with regard to the wetter season as the lack of data.

3.4. The Average of Unit Hydrographs for Different Time Steps

Comparing the results between those three catchments there is a general trend of average peak discharge, *i.e*. the higher the time step, the lower the average peak discharge (**Figure 7**). This trend does not fully work for Way Garuntang as the peak discharge using time step 10 minutes is lower compare to that using time step 30 and 60 minutes. Considering overall events for Way Kuala Garuntang, average peak discharges for time steps 10, 30 and 60 minutes are 4.64 m^3/s, 5.2 m^3/s and 4.7 m^3/s. Although in nearly all unit hydrographs, peak discharges resulted from using time step 10 minutes are larger compared to peak discharges resulted from using larger time step. This may happen because the method in calculating the average peak discharge is so simple, that is simply taking the average of the events for particular time step, both for the discharge and time to peak. Furthermore, peak discharge is closely related to time to peak. Using small time step, time to peak may vary significantly between 10 to 60 minutes. Considering peak discharges which occur at various time to peak, this may result in low average of peak discharge as in the case of average peak discharge of Way Kuala Garuntang using time step 10 minutes.

4. Conclusions

This study shows the impact of time steps on unit hydrographs with regard to time to peaks and peak discharges. In general, smaller time step gives more accurate resulted unit hydrographs. It was observed that the average time to peaks for Way Simpur are 20, 30 and 60 minutes using time steps 10, 30 and 60 minutes respectively. The average time to peaks for Way Awi are 30, 30 and 60 minutes using time steps 10, 30 and 60 minutes respectively. And the average time to peaks for Way Kuala Garuntang are



(a) (b) (c)

Figure 7. Average of unit hydrographs for time steps 10, 30 and 60 minutes for catchments (a) Way Simpur; (b) Way Awi; and (c) Way Garuntang.

60, 60 and 60 minutes using time steps 10, 30 and 60 minutes respectively.

Time steps used in determining unit hydrographs may produce different peak discharges. The results show that the trend of peak discharges increases by using smaller time step. However, the average peak discharge provided by using time step 10 minutes for Way Kuala Garuntang does not correspond with the trend. For several flood events used in determining unit hydrographs, the time to peaks and corresponding peak discharges using time step 10 minutes vary considerably. Therefore the averaging of those variables causes the average value of peak discharge is not maximum.

In addition to that, the resulted unit hydrographs also show that the trend of peak discharges increases toward wetter months during the wet season. However, the peak discharges resulted from using time step 10 minutes do not show this trend due to limited number of flood events which could be used to calculate the unit hydrograph using such small time step.

5. Acknowledgements

This study was supported by National Strategic Research Funding 2009 and 2012 under Department of Research and Community Service, Department of Higher Education Indonesia (DIKTI).

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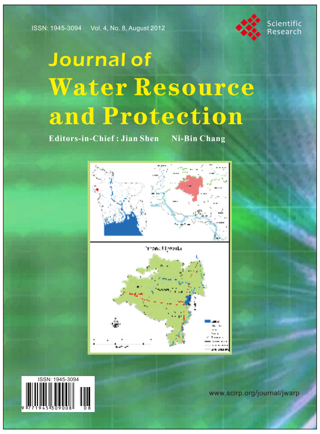

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J. Water Resource and Protection, 2012, 4, 567-693
Published Online August 2012 in SciRes (http://www.SciRP.org/journal/jwarp).

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