BIRZEIT UNIVERSITY

Effect of Depth on Nitrogen Transformation in Duckweed and Algae-Based Ponds as a Post-Treatment Stage for a UASB-Septic Tank

BY

Ashraf A. Isayed

This Thesis was submitted in partial fulfillment of the requirements for the Master Degree in Water Engineering from the Faculty of Graduate Studies at Birzeit University-Palestine

April, 2005

Effect of Depth on Nitrogen Transformation in Duckweed and Algae Based Ponds as a Post-Treatment Stage for a UASB-Septic Tank

By

Ashraf A. Isayed
Student No. 1025081

This thesis was prepared under the supervision of Dr. Omar Zimmo and has been approved by all members of the examination committee.

Dr. Omar Zimmo ……………………………
Chairman of Committee

Dr. Rashed Al-Sae'd ……………………………
Member

Dr. Nidal Mahmoud ……………………………
Member

Date of defense: April 12, 2005.

The findings, interpretations and the conclusions expressed in this study do not necessarily express the views of Birzeit University, the views of the individual members of the MSc-committee or the views of their representative employers.

Dedication

To my dear parents, sisters and brothers

God bless them all

Acknowledgements

First of all, I am grateful to my supervisor, Dr. Omar Zimmo who spent long time discussing and giving advice along this year during the implementation and finalization of the experimental work, and for the valuable comments and ideas along the research period.

I am especially indebted to Dr. Nidal Mahmoud and Dr. Rashed Al-Sae'd who spent many hours discussing and giving advice during the implementation and finalization of the experimental work.

I would like also to appreciate the staff of the Institute for Water at Birzeit University, Dr. Munther Barakat, Dr. Ziad Al-Mimi, Dr. Maher Abu Madi, and Miss Nadine Sinokrot, who helped me not only during the implementation of this research but also throughout my MSc. Study.

I would like to express my appreciation for the help and support of the laboratory technicians of the MSc. Program at Birzeit (Eng. Abdel Haleem Foqaha, and Riad Bsharat) for their technical support and for the laboratory assistance.

My thanks to Eng. Marwan Ayyad for his technical and scientific support, mainly in the start-up period of this work. I am especially grateful also to eng. Jamal Burnat for his valuable discussions and assistance.

Thanks to my sincere colleagues Mohammad Al-Shayah, Subhi Samhan, and Haitham Lahaleeh for their help and advice during the implementation and finalization of the experimental work.

I gratefully acknowledge Al-Bireh Municipality specially Eng. Munther Hind for providing all possible support to conduct the research at the site of Al-Bireh Wastewater Treatment Plant and many thanks go to the employees of Al-Bireh Wastewater Treatment Plant.

I would like to express my appreciation for the help and support of the Palestinian Hydrology Group (PHG) members especially Ms. Samar Husary, Dr. Sae'd Jasser, Dr. Abdel Rahman Al-Tamimi, Wae'l Awadallah and for my colleagues Yousef Salah and Safa' Sama'nah for their assistance and help.

I would like to express my gratitude to the Dutch Agency for International Cooperation (DAIC) for their financial support within the framework of "Wasteval II" (capacity building) project headed by Dr. Nidal Mahmoud for financially supporting this research.

Finally, my deeply felt thanks go to my mother, father, sisters, brothers and my friends Rani Daoud, Rae'd Salameh, Rami Ghannam and Odai Salameh. There are no words that express my thanks and appreciation towards them.

Abstract

Upflow anaerobic sludge blanket (UASB) is one of the anaerobic systems which is known for its low construction, operation and maintenance cost, high efficiency in removal of organic material, small land requirement, low operation and maintenance cost as well as low sludge production. Nevertheless, UASB is known for low efficiency in nitrogen (N) and phosphorus (P) removal. However, nitrogen is a major nutrient in wastewater that must be reduced to acceptable levels because the uncontrolled release of nitrogen to the environment is known to cause serious problems such as infant methaemoglobinaemia and eutrophication. Therefore, an appropriate post-treatment unit has to be installed after UASB to comply with effluent guidelines for reuse. Waste stabilization ponds (WSP) which are also considered as low cost system, are able to meet effluent standards for reuse. However, these systems are characterized by high land requirement.

Duckweed-based ponds (DBPs) are at their core, modified type of WSP. Furthermore, the WSP are not only a low cost and easy to build and operate, but they also produce tertiary quality effluents and offer the possibility of resource recovery by producing high quality duckweed protein, which can be of further use. DBPs as well as WSP systems are efficient as post-treatment units after UASB system. Moreover, land requirement for WSP and DBPs systems can be considerably reduced when influent sewage is pretreated by UASB. Nevertheless, the use of DBPs is more promising because of the aforementioned cost recovery. Moreover, land requirement and hence cost can be further reduced by using deeper pond systems. However, the effect of depth on ponds performance especially in nitrogen removal has so far not been investigated especially under Palestinian conditions.

The main objective of this thesis was to investigate the effect of depth variation on DBPs and ABPs performance and nitrogen removal efficiency. In order to perform this study, a pilot-scale treatment plant was constructed at Al-Bireh Wastewater Treatment Plant site (AWWTP), 15 Km northeast of Jerusalem-Palestine. The pilot plant consisted of a UASB-septic tank operated under hydraulic retention time (HRT) of 4 days. It was followed by three parallel lines of stabilization ponds with three equal ponds each of similar total HRT of 28 day per line. The depth of the ponds in the first, second and third lines were respectively 90, 60 and 30 cm. The study was divided into two periods; the first period was conducted between May, 2, 2004 and August, 18, 2004 to investigate the effect of depth variation in DBPs. As the growth of duckweed and hence the duckweed cover was not maintained due to unfavorable conditions of treated sewage, the

same as the first period was investigated for algae based ponds (ABPs) in the period between August, 18, 2004 and November, 1, 2004.

The pilot plant was operated for six months, the average ambient temperature throughout the experimental period was 24.5 ^{o}C, while the average water temperature in the first and second periods were 23.6 and 22.9 ^{o}C, respectively. Influent total COD to the system was 1275 ± 84 mg/L, while average COD concentration in DBPs and ABPs period was 701.6 ±241.5 and 330.9 ± 69 mg/L, respectively. The corresponding volumetric loading rates were 25.0 and 11.8 g COD/m^3.d, respectively. The results of this research revealed that total COD, TP and N removal efficiencies were inversely proportioned to depth when equal total HRT was applied for the three lines. COD removal efficiency for the shallowest and deepest DBPs were 75.4 ± 4.1%, 62.5 ± 5.7%, while its removal efficiency in the shallowest and deepest ABPs was 54 ± 1.1%, and 51.6 ± 3.2%, respectively. Moreover, total phosphorus (TP) removal in DBPs and ABPs increased by the decrease in depth. The removal efficiencies of TP in the shallowest and deepest DBPs were 48.5 ± 9.2% and 38.5 ± 4%, respectively. While TP removal efficiencies in the shallowest and deepest ABPs were 57.6 ± 5.6% and 37.6 ± 6.4%, respectively. Furthermore, total suspended solids (TSS) removal efficiency was higher for deeper DBPs; it was 64.4 ± 11.8%, and 58 ± 11.2% in shallowest DBPs, however negative removal efficiency was achieved in ABPs due to algal growth. Higher ammonium (NH_4^+) removal efficiencies were achieved in the shallowest compared to deepest DBPs were 46.6 ± 5.2% , 30.9 ± 1.6%, while, in the shallowest and deepest ABPs, they were 64.5 ± 2.8% and 51.2 ± 1.9%, respectively. Furthermore, the removal efficiencies of total Kjeldahl nitrogen (TKN) in the shallowest and deepest DBPs were 44.5 ± 6.3% and 29.4 ± 6.8%, respectively. While they were in the shallowest and deepest ABPs 45.4 ± 3.1% and 61.1 ± 4.5%, respectively. Significantly higher value of sedimentation (higher contribution to treatment) was found in ABPs compared to DBPs. Finally, even though better removal efficiency was achieved in shallowest ponds for most of the tested parameters, however, they showed higher land requirement (11.7 m^2/capita and 7.8 m^2/capita in DBPs and ABPs, respectively) compared to deepest ponds (6.9 m^2/capita and 3.9 m^2/capita in DBPs and ABPs, respectively) to comply with WHO guidelines for restricted irrigation.

Table of Contents

List of Tables

List of Figures

List of Photos

List of Abbreviations

ABPs	Algae-based ponds
AiLj	i^{th} algae-based pond in the j^{th} line
AV	Ammonia volatilization
AWWTP	Al-Bireh Wastewater Treatment Plant
BOD	Biological oxygen demand
COD	Chemical oxygen demand (mg /L)
DBPs	Duckweed-based ponds
DiLj	i^{th} duckweed-based pond in the j^{th} line
DO	Dissolved oxygen (mg/L)
eff	Effluent
HRT	Hydraulic retention time (d)
inf	Influent
L	Liter
N	Nitrogen
n	Number of samples
NH_3	Ammonia
NH_4^+	Ammonium
TKN	Total Kjeldahl-nitrogen
NO_2^-	Nitrite
NO_3^-	Nitrate
P_iL_j	i^{th} pond of the j^{th} line
Q	Flow rate (m^3/d)
S.E.	Standard error
T	Temperature (^oC)
t	Time (d)
TDS	Total dissolved solids
TS	Total solids
TSS	Total suspended solids (mg/L)
UASB	Upflow anaerobic sludge blanket
WSP	Waste stabilization pond

Greek:

α	unionized ammonia fraction
ρ	significance level

Chapter One

Introduction

1.1 Background

The Palestinian territories face serious problems in the field of water supply, sanitation and wastewater treatment. Water supply problems originate mainly from the scarcity in water resources, but wastewater problems originate mainly from the lack of finance for construction and running cost of conventional wastewater systems. Besides, the environmental problems created by wastewater (WW) production is increasing as a result of increasing population density which necessitate a great challenge to develop and introduce sustainable sewage collection and treatment systems. On the other hand, providing these essential services especially for poorer localities of the world are hindered by the financial limitations and weakness of water supply and sanitation management, correspondingly providing necessary funds for capital operation and maintenance costs is the key issue for development of the sanitation sector especially in low income regions (Zimmo, 2003).

About 20% of the total Palestinian population in the urban areas is served by a central urban sewerage system, while only 5% of the collected municipal wastewater experienced partial treatment. About 73% of the households in the West Bank have cesspit sanitation and almost 3% without any sanitation system (Abu Madi, 2000; MOPIC, 1998). Lack of effective planning in water and sanitation infrastructures in rural communities has led to poor environmental health, ineffective wastewater treatment, exacerbated by under-funded national institutions and lack of skilled staff and appropriate management and coordination (Al-Sa`ed, 2000). Beside applied research studies a number of biological treatment systems for small rural communities were introduced and installed (Mustafa, 1996; PHG, 1999; Zimmo *et al.,* 1999).

Waste stabilization ponds are low cost and efficient systems for wastewater treatment, producing high quality effluent that enables wastewater reuse in irrigation (Mara, *et al*., 1992). There is an international continuous interest for waste stabilization pond systems that are inexpensive and are known for their ability to achieve good removal of organic pollutants and pathogens, however, high algal concentrations of about 100 mg TSS/L may be reached in the effluent (Middlebrooks, 1995), causing sever clogging problems in advanced drip irrigation systems (Person *et al.,* 1995). Correspondingly, increasing interest in modifications of conventional algae based ponds into duckweed-based ponds (DBPs) is observed (Zimmo, 2003).

Waste stabilization ponds can be used to treat secondary quality effluent; it can polish an anaerobic enhanced pre-treated effluent and, with appropriate retention time, can remove pathogens to an acceptable level before discharge into receiving stream or before reuse for irrigation or groundwater recharge (Journey and Scott, 1996). If the ponds are to be designed such that the effluent complies with WHO (1989) microbiological quality guidelines for restricted irrigation (<1,000 faecal coliforms/100 ml), then the major limitation for the application of stabilization pond technology for wastewater treatment at large scale is the large area needed, which is estimated at about 5-7 m^2/capita depending on influent strength of wastewater and effluent guideline requirements (Al-Sa'ed, 2004). However, when stabilization ponds receive their effluent from UASB land requirement may be reduced to the half or more (Jabari, 2003). HRT is an important parameter that plays a fundamental role in determination of the effluent quality and correspondingly the land requirement. The increase in ponds' depth permits the same quantity of influent to be collected in the smaller area of land with smaller water losses through evaporation. The feasibility of conducting wastewater treatment in deep ponds has been proven in several studies carried out in the Region of Murcia, SE of Spain which is characterized by a Mediterranean climate (Soler *et al.*, 1991).

Anaerobic wastewater treatment systems were found to be efficient in reducing the organic loads considerably. However, these systems leave mineralized compounds like ammonium and phosphorus in the solution (Mahmoud, 2002). Upflow anaerobic sludge blanket (UASB) technology is considered also as another low cost alternative because of its' low cost required for construction, operation and maintenance. Moreover, UASB system showed promising results as a wastewater pretreatment unit in tropical countries like Columbia, India, Brazil, and Indonesia (Journey and Scott, 1996). Several UASB pilot plants were constructed there, and many researches for different anaerobic treatment technologies were discussed, however, at low temperatures, more sophisticated reactors are needed (Elmitwalli, 1999). Therefore, modifications of UASB and upgrading the treated effluent still need more investigation at moderate and low temperature circumstances to achieve more reliability and applicability. Principally, anaerobic treatment plant is a mineralization process, and so it is a pretreatment method and always needs a proper post treatment method to upgrade the anaerobic effluent to standards for reuse or for discharge. This applies particularly for pathogens, remaining BOD and mineralized compounds (Jules *et al.,* 2001). There still quite number of attractive new developments can be expected for post-treatment of pretreated sewage by UASB (Lettinga *et al*., 1999).

Nitrogen is a major nutrient in wastewater that must be reduced to acceptable levels. Its removal in wastewater treatment is very important because its existence in the effluent causes eutrophication in the receiving water bodies and groundwater contamination (Horne and Goldman, 1994). Besides, ammonia is toxic to aquatic organisms, especially the higher forms such as fish, at concentrations as low as 0.5 mg/L (Barnes and Bliss, 1983 as quoted by Javed (1995)). On the other hand nitrogen in wastewater should be handled as a nutrient resource rather than a pollutant that only has to be disposed off (Gijzen and Mulder, 2001). Nitrogen removal could be achieved by nitrification and denitrification within activated sludge systems and by break point chlorination or by ion exchange, however, the investment and operational costs are high. Recently, removal of nitrogen in low cost conventional algae-based ponds (ABPs) and duckweed-based ponds (DBPs) is given some attention (Zimmo, 2003). According to preceding studies, a combination of UASB and waste stabilization ponds proved to be promising in terms of land requirement reduction. However, no previous studies investigated the effect of changing ponds' depth (with maintaining the same ponds' area) on removal efficiency of different pollutants, and no reliable data is available for explanation the basic changes in the water column environment of the pond and consequently, the effect of depth variation on efficiency of removal for different pollutants especially nitrogen compounds.

1.2 Objectives

Increasing depth of waste stabilization ponds and the issue of anaerobic pretreatment is expected to play a significant role in decreasing the per capita land required to approach the guidelines. The effect of depth of stabilization ponds with anaerobic pretreatment on the performance of the ponds was not studied mainly for the issue of nitrogen removal efficiency.

The main objectives of this thesis can be summarized as follows:

- ⊙ Determination of the effect of depth on nitrogen transformation in duckweed and algae-based waste stabilization ponds.
- ⊙ Determination of the effect of depth on removal of major constituents of pollutants in duckweed and algae-based waste stabilization ponds.
- ⊙ Determination of the differences between duckweed and algae-based waste stabilization ponds in terms of performance and efficiency.

The aforementioned research objectives can be achieved when conducting the following research questions:

- ⊙ Investigate the ability of duckweed to survive in an anaerobically pretreated municipal wastewater.
- ⊙ Identify influent wastewater characteristics (effluent of UASB-septic tank) in the pilot plant.
- ⊙ Identify wastewater characteristics changes after each pond.
- ⊙ Identify the number of ponds needed (land requirement) to achieve reclaimed water guidelines at each depth.
- ⊙ Identify the (best) expected performance of suggested system as one unit during the summer time of highest ambient temperature.
- ⊙ Compare the performance of duckweed and algae-based ponds as two methods for post treatment of anaerobically pretreated wastewater.

1.3 Contents of the thesis report

This thesis report contains five chapters; chapter two is “literature review” which represents the main outcomes of other preceding related researches. Chapter three “materials and methods”, reviews the materials and methods used in this experimental work including sampling and analysis. In chapter four “results and discussion”, results are discussed based on laboratory results with a comparison between the achieved results and previous results from earlier related studies. Finally, chapter five “conclusions and recommendations”, shows the main outcomes and the recommendations for other complementary researches.

Chapter Two

Literature Review

2.1 General

Nitrogen is a major pollutant in wastewater that must be reduced to acceptable levels. Its removal in wastewater treatment is very important because of eutrophication in receiving bodies and groundwater contamination (Zimmo, 2003). Nitrogen removal is achieved via several methods like ion exchange, nitrification and denitrification in activated sludge systems, or by break point chlorination. However, the common problem in these systems is the high operational and maintenance cost which makes these systems hard to be implemented in poor countries. A high attention was recently given to the low cost stabilization ponds (algae-based ponds (ABPs), and duckweed-based ponds (DBPs)).

2.2 Algae and duckweed-based waste stabilization ponds

Waste stabilization ponds (WSPs) are usually the wastewater treatment method of choice in warm climates where land is available at reasonable costs, (Mara 1976, Arthur 1983). WSP systems are inexpensive and are known for their ability to achieve good removal of pathogens and organic pollutants. However, stabilization ponds systems are characterized by the high land requirement (5-7 m^2/capita) with a retention time from 15-18 days, depending on temperature and required effluent quality (Al-Sa'ed, 2004).

2.2.1. Algae based ponds

ABPs are the simplest form of waste stabilization ponds. ABPs are low-cost and efficient systems for wastewater treatment, producing high quality effluent that enables water reuse in irrigation (Mara *et al.,* 1992). Worldwide, there is a continuous interest for ABPs systems that are inexpensive and are known for their ability to achieve good removal of pathogens and organic pollutants. An advantage of ABPs compared to other technologies such as activated sludge is that besides suspended solids and COD, also pathogens are removed (Zimmo, 2003). However, high algal concentrations may be occasionally reached in the effluent, causing severe clogging problems in advanced (drip) irrigation systems (Pearson *et al.,* 1995).

2.2.2. Duckweed based ponds

Duckweed is a floating aquatic macrophyte belonging to the botanical family *Lemnaceae* that can be found worldwide on the surface of nutrients rich fresh and brackish waters. Duckweed plants are characterized by an excellent potential as a commercial crop plant because of their high growth rate, high nutritional value and low fiber content. Duckweed (*Lemnaceae*) is a family of floating monocoteledons consisting of four genera (*Lemna, Spirodela, Wolfia* and *Wolfiella*) and twenty-eight species (Sculthorpe, 1967). However, *Lemna gibba* proved to be the best duckweed species out of three examined genera; (*Lemna gibba, Wolfia and Spirodela*) regarding treatment efficiency and growth rate (Oron and Porath, 1987).

DBPs are a modified type of stabilization ponds, covered with floating mat of plants. However, they differ from conventional stabilization systems by achieving higher nutrient removal levels, removing organic matter and other oxygen consuming substances compared to ABPs (Skillecorn *et al.*, 1993), suppressing algae growth so that no large amount of algae is washed out of the system as suspended matter (Steen, 1998). Evapotranspiration is lower than evaporation from an open water surface under the same meteorological conditions (Oron *et al.*, 1985). DBPs systems have been applied at full-scale in Taiwan, China, Bangladesh, Belgium and the USA (Edwards, 1980; Zirschky and Reed, 1988; Alaerts *et al.*, 1996). Moreover, the Palestinian Environmental Quality Authority is developing, since 1999, duckweed-based ponds receiving mixed domestic and agricultural wastewater (at Al-Arroub – Hebron - Palestine). It is the first such system operating at full-scale in the country (Awadallah, in preparation).

2.2.2.1 Advantages and limitations of DBPs systems

DBPs are not only characterized by low cost and simple equipment, low energy or unskilled labor input (Awadallah, in preparation), but they also aim at resource recovery in the light of sustainability of wastewater treatment. On the other hand, DBPs were recommended in literature to be used as secondary or tertiary treatment unit. Bonomo (1996) reported based on experimental results and the information deduced that duckweed has to be used for secondary treatment (SS and BOD removal) and for tertiary treatment (algae control and N-Removal). Additionally, duckweed-based systems must be preceded by adequate pre and primary treatments in order to avoid accumulation of sludge and floating matter in the pond. Table 2.1 compares the two types of pond systems.

Table 2.1. Comparison between algae based ponds and duckweed based ponds.

Criterion	ABPs	DBPs
Robustness	• Extremely robust. • High ability to absorb organic and hydraulic shocks.	High BOD loads needs appropriate pretreatment
Capital costs	Low	25% higher than WSPs (Source: PRISM)
Labor requirements for operation and maintenance	• Low labor requirement. • Unskilled but supervised labor is sufficient. • Extreme simplicity of O&M.	• Highly labor intensive. • Requires skilled labor. • Sophisticated management necessity.
BOD removal efficiency	> 90%	> 90%
Nutrients removal efficiency	$N_{tot.}$: 70-90%, $P_{tot.}$: 30-50%.	N_{tot} and P_{tot} 70%.
TSS removal efficiency	Low because of the algae in the final effluent.	High due to inhibition of algae.
Pathogen removal efficiency	High	Mainly unknown, but good preliminary results.
Valorization of biomass	None	• Use as animal feed. • Revenue generation.

Source: WHO (1989), Alaerts *et al.* (1996) and Asano (1998).

2.2.2.2 Wastewater treatment by DBPs

Duckweed mat has the capability to purify wastewater in collaboration with both aerobic and anaerobic bacteria. The duckweed mat, which fully covers the water surface, results in three zones in DBPs; these are the aerobic zone which locates within 10 cm below the duckweed mat (skillicorn *et al.,* 1993), the anoxic zone and the anaerobic zone. In the aerobic zone, organic materials are oxidized by aerobic bacteria using atmospheric oxygen transferred by duckweed roots. Nitrification takes place in the aerobic zone. However, denitrification takes place in anoxic zones, where organic nitrogen is decomposed by anoxic bacteria into ammonium and *ortho*-phosphate, which are the intermediate products used as nutrients by the duckweed (Metcalf and Eddy, 1991). Organic matter in the bottom of the ponds is decomposed by anaerobic bacteria and this produces gases such as carbon dioxide (CO_2), hydrogen sulfide (H_2S) and methane (CH_4) (Metcalf and Eddy, 1991). Nutrients are removed from the wastewater by several processes, including volatilization of NH_3 and sedimentation of suspended solids with organic nitrogen. Part of the nutrients is removed by conversion into plant proteins and by harvesting the biomass. The nutrient uptake by the duckweed plants was reported to be 60-80% of the nitrogen and phosphorus load (0.5 gTKN-N/m^2.d) and (0.09 gTKN-N/m^2.d), respectively (Alaerts *et al.,* 1996).

2.3 Anaerobic Pretreatment

Anaerobic treatment is an effective enhanced primary treatment option for developing countries, particularly those with mild climates. This treatment technology is characterized by efficient removal of organic material, low construction cost, small land requirement, low operation and maintenance cost, lowest sludge production compared to other physical or chemical systems, useful biogas production which can be used for energy generation. However, some limitations for the anaerobic treatment were reported in literature like the need for warm climates, as the optimal reactor temperature is 20^o C and above. Moreover, longer startup time is required in anaerobic systems because of the slow growth rate of anaerobic bacteria. Furthermore, additional treatment for the effluent from anaerobic systems is required to meet secondary quality standards in terms of oxygen consuming substances and odor problems. Moreover, chemical buffering may be required to maintain alkalinity in reactor (McCarty, 1981; Giraldo, 1993; Vieira, 1992).

2.3.1 UASB technology

UASB is a cheap anaerobic wastewater treatment technology that is characterized by low land requirement, ease of operation and maintenance, and low sludge production. However, its effluent always needs to be post-treated in order to achieve reuse guidelines. Moreover, UASB is a high rate suspended growth type of reactor in which a pre-treated raw influent is introduced into the reactor from the bottom and distributed evenly. UASB is one of the most popular anaerobic systems that showed acceptable and encouraging results as a pre-treatment method in many countries (e.g. India, Brazil, and Indonesia) (Journey and Scott, 1996). Lettinga and colleagues developed the UASB process in the late 1970's at the Wageningen University (The Netherlands). The UASB concept was born out of the recognition that inert support material for biomass attachment was not necessary to retain high levels of active sludge in the reactor. Instead, the UASB concept relies on high levels of biomass retention through the formation of sludge granules. The main features for achieving granular sludge development are firstly to maintain an upward-flow regime in the reactor selecting for microorganisms that aggregate and secondly to provide for adequate separation of solids, liquid and gas and preventing washout of sludge granules.

From a hardware perspective, a UASB reactor is at first appearance nothing more than an empty tank (thus an extremely simple and inexpensive design). Wastewater is distributed into the tank at appropriately spaced inlets. The wastewater passes upwards through an anaerobic sludge bed where the microorganisms in the sludge are exposed to wastewater-substrates. The sludge bed is composed of microorganisms that naturally form granules (pellets) of 0.5 to 2 mm diameter that have a high sedimentation velocity and thus resist washout from the system even at high hydraulic loads. The upward motion of released gas bubbles causes hydraulic turbulence that provides reactor mixing without any mechanical parts. At the top of the reactor, the water period is separated from sludge solids and gas in a three-phase separator (also known the gas-liquid-solids separator). The three-phase-separator is commonly a gas cap with a settler situated above it. Below the opening of the gas cap, baffles are used to deflect gas to the gas-cap opening.

2.3.2 UASB-septic tank

The UASB-septic tank system is a promising alternative for the conventional septic tank (Bogte *et al.*, 1993; Lettinga *et al.*, 1993). It differs from the conventional septic tank system by the upflow mode in which the system is operated, resulting in both improved physical removal of suspended solids and improved biological conversion of dissolved components. The most important difference with the traditional UASB system is that the UASB-septic tank system is also designed for the accumulation and stabilization of sludge. Therefore, a UASB septic-tank system is a continuous system with respect to the liquid, but a fed-batch or accumulation system, with respect to the solids. Anaerobic pretreatment requires an appropriate post treatment to enhance effluent quality for further reuse. Al-Juaidy (2001) discussed the performance of UASB-septic tank that was constructed at Birzeit University campus as a pilot plant during the summer period. The influent was black wastewater from the Faculty of Commerce. He showed that UASB-septic tank presented an effective on-site treatment (COD removal was 76% and the TSS removal was 58 %). He suggested that another post treatment should be added to reach the acceptable requirements for reuse of wastewater and to eliminate nutrient (NH_4^+, PO_4^{-3}) and pathogens. Ali *et al.* (2004) also conducted a different study using domestic wastewater. COD and TSS removal efficiencies were 90% and 50 %, respectively. Recently, Al-Shayah (2005) investigated the performance and feasibility of using two UASB-septic tank reactors (HRT = 2 days for R1 and 4 days for R2) under the conditions that arise at the community level in Palestine. He used the wastewater form Al-Bireh wastewater treatment plant with COD_{tot} 1189 mg/L. He reported that the removal efficiencies for COD and TSS for R1 54 and 79%, likewise, the removal efficiencies in R2 for the same parameters were 58 and 80%. He recommended the use of the 4 days HRT UASB-septic tank reactor under Palestine conditions.

2.3.3 Effect of anaerobic pretreatment on ponds' performance

Ciacedo *et al.* (2000) tested the effect of anaerobic pretreatment for stabilization ponds in a system fed with institutional wastewater on several parameters. They reported that Oxygen levels are significantly higher in the stabilization ponds with anaerobic pretreatment, especially in the top layers. Additionally, the latter found also that pH levels are very stable in ABPs and DBPs systems with and without anaerobic pretreatment. Temperature gradients are present during daytime but not as high as they may be in conventional stabilization ponds. In an attempt to reduce the area requirements, they found that anaerobic pretreatment will also change environmental and physicochemical characteristics in the ponds, as the organic matter will be greatly reduced in the anaerobic reactor. Besides, due to hydrolysis of organic matter, nutrients will be present in a soluble form ready to be used by the plants. Consequently, the combination of deep stabilization ponds with UASB seems to have promising results especially for the land requirement issue. However, some aspects of such systems like the issue of depth variation have not been studied sufficiently in literature and therefore further research is required to test this concern.

No difference in removal efficiency was reported for the effect of anaerobic pretreatment on nitrogen removal efficiency. Caicedo *et al.* (2002) compared the nitrogen removal between stabilization ponds with anaerobic pretreatment (UASB) and stabilization ponds without pretreatment. She reported that nitrogen removals were slightly higher in the system without pretreatment but no significant difference was found between the two systems, nitrogen removal percentages were 35–46% and 40–46% in the lines with and without pretreatment, respectively. Pond system with pretreatment influent nitrogen was mainly (90%) ammonium, since organic nitrogen was hydrolyzed in the UASB reactor. The nitrate concentration in the ponds was therefore low and most of the nitrogen available to the duckweed was in the form of ammonium.

2.4 Effect of depth on pond performance

The increase in the ponds' depth (maintaining constant surface area) increases the pond volume, and then increase the hydraulic retention time (HRT).

2.4.1 Effect of depth on environmental conditions

The biological and physicochemical processes occurring in conventional stabilization ponds are complex. Depending on the organic loading aerobic, anaerobic and facultative zones can be presented (Metcalf and Eddy, 1991). Dissolved oxygen (DO) is an important parameter that significantly affects the environmental conditions and consequently it affects the removal processes in the treatment system. For example, oxygen concentrations higher than 1 mg/l are needed to have an efficient nitrification process (Metcalf and Eddy, 1995). DO concentrations in both algae and duckweed ponds decreased rapidly with the distance from the water surface, DO also decreases by the increase in organic surface loading. Literature and previous reports shows that DO in the water column was higher in ABPs than DBPs, this is justified by the photosynthesis of algae species (Zimmo, 2003; Caicedo, 2002; Steen, 1998).

The sizes of facultative and anaerobic zones are highly affected by pond depth value. Zimmo (2003) who carried out his researches in 0.9 m depth ponds found that the three mentioned zones were within the following three depths of each pond: 0 to 37, 37 to 73 and 73 to 90 cm. He reported also that the pH was highest near the surface of the water column and slightly decreased with the distance from the water surface. Additionally, Smith and Moelyowati (1999) found that water temperature was highest at the surfaces than below in all ponds. However, he concluded that differences with depth were not significant. The effect of depth on the environmental conditions was also studied by Javed (1995) who used the batch flow reactors. He found that the growth rate of duckweed was bit lower in 95 cm depth than lesser depths.

2.4.2. Effect of depth on removal of pollutants

2.4.2.1 Effect of depth on nitrogen removal

The effect of depth on nitrogen transformation was not discussed sufficiently in literature. In comparing the effect of depth in the available literature, the results at lower depth are more promising for N-removal than that at higher depth (Steen, *et al*., 1998; Awadallah, in preparation; Smith and Moelyowati , 1999). Small ponds depth of 30 to 65 cm results in high surface area/volume ratios. It is likely that in DBPs higher surface area per volume ratio will result in higher nitrogen removal via duckweed uptake (Körner and Vermaat, 1998; Steen *et al*., 1998). This will enhance also the surface and/or volume related processes of nitrogen removal, such as ammonia volatilisation, denitrification and sedimentation. However, for the case of ABPs, in shallower ponds, the amount of light available per pond volume is higher compared to deeper ponds. This would result in higher algae growth and consequently in an increase in oxygen produced via photosynthesis. This would also favour nitrification, the limiting step for denitrification and nitrogen removal by sedimentation (Reddy and DeBusk, 1987). These results were also reported in several studies. Javed (1995) reported that there was significant difference in nitrogen removal in different depths; he reported a removal efficiency of TKN of about 86% and 33% at a depth of 10 and 95 cm, respectively.

Similarly, Körner and Vermaat (1998) reported 73 to 97% removal of the initial TKN within 3 days in laboratory scale duckweed-covered systems (18.5 cm diameter and 4.5 cm depth, respectively). Buddhavarapu and Hancock (1991) reported NH_4^+ removal efficiency of 70% in duckweed-based pond receiving secondary treated domestic wastewater with NH_4^+ concentration of 0.1-10 mg/l. This experiment was conducted in a pond of 1.8 m depth and a HRT of 13 days. Oron (1985) reported a total NH_4^+ removal in the range of 40-70% in semi-continuous duckweed ponds. These authors used raw sewage with initial NH_4^+ concentrations of 47.5 ± 16 mg/l and COD concentrations of 318 ± 69 mg/l. These experiments were conducted in mini-ponds with water depths of 20 and 30 cm and a HRT of 3 and 10 days. Silva (1982) obtained ammonium removal efficiency of 81% in a system of depth 1.0 m. Reddy and DeBusk (1985) conducted outdoor experiments with *Limna Minor* using simulated wastewater having depth of 40cm, under solar radiation of 500-930 $\mu E/m^2.s$ and ambient temperature of 14-27 ^0C. Nitrogen removal was 2.92 Kg-N/ha.day by duckweed uptake. The total removal by their system was 9.46 Kg-N/ha.day.

2.4.2.2 Effect of depth on total phosphorus removal

Total phosphorus (TP) is the sum of the organic and *ortho* phosphorus. TP is normally reduced by plant uptake, adsorption onto clay particles and organic matter, chemical precipitation and sludge removal (Iqbal, 1999). Few data related to effect of depth on total phosphorus removal is available in literature. Javed (1995) reported that an 88% removal of total phosphorus in batch flow reactors at a depth of 10cm, whereas at a depth of 95cm, the removal efficiency was reported to be 24%. However, Zimmo (2003) reported that TP was respectively 74-79% and 74-92% in 0.9 m depth ABPs and DBPs during the warm seasons. Moreover, Reddy and DeBusk (1985) observed phosphorus removal by duckweed uptake at depth of 40 cm to be 0.87 Kg-P/ha.day. Alerts (1996) reported that the TP uptake by the duckweed plants was reported to be 60-80% of phosphorus load 0.9 kg TP /ha.day. Finally, Caicedo (2002) who tested the effect of anaerobic pretreatment for 40 cm depth DBPs fed with institutional wastewater, found that TP removal efficiencies was 40–46% in the lines with and without pretreatment, respectively. However, the comparison between the above-mentioned results is inconvenient due to wide differences in experimental conditions.

2.4.2.3 Effect of depth on COD removal

Javed (1995) reported that the removal efficiency of COD in the 10 cm depth reactor is 63% whereas in the 95 cm reactor 56%. He concluded that depth is not playing an important role in the reduction of wastewater COD because other natural activities could also take place such as sedimentation and biodegradation by microbes. Moreover, Caicedo (2002) reported an 82% of COD was removed in DBPs (0.7m depth and HRT = 21days) fed with anaerobically pre treated wastewater. Moreover, Bonomo (1997) found that COD removal efficiency was higher than 75% in 2.3 m depth DBPs fed with primary settled wastewater. However, Steen (1998) found a 55 ± 26% COD removal in a combined 0.29 m depth-DBPs and ABPs fed with anaerobically pretreated domestic sewage. Al-Jabari (2003) who tested COD removal in 90 cm depth- DBPs and ABPs fed with the effluent of UASB, achieved 80% and 70% removal efficiency in DBPs and ABPs, respectively.

2.4.2.4 Effect of depth on heavy metals removal

Few data and previous reports discussed the effect of depth on heavy metals removal from wastewater. Abdel Wahab (1995) reported that heavy metals removal does not depend on water depth but it is dependent on duckweed uptake and biomass-water contact surface, which are both defined by the water surface area. However, for two ponds of same surface area with different depth, heavy metals removal is expected to be higher for shallower pond.

Direct comparison of results from the above studies is not possible due to differences in HRT, water depths, initial nitrogen concentrations and duckweed species, densities and harvesting regimes. On the other hand, some studies considered that the behaviour of ponds that have more than particular depth is similar. For example, Smith and Moelyowati (1999) suggested in their model for calculating the effluent quality from DBPs the following equations. The latter assumed the validity of these equations for the ponds of depth ≥ 60cm. actually, considerable error values could be gained if these "empirical" formulas were used for deep ponds (>>60 cm depth) because difference in performance is expected to be observed when deep and shallow ponds are compared. Table 2.2 shows the suggested formulas presented by Smith and Moelyowati (1999) for calculating effluent quality from DBPs.

Table 2.2. Calculation of effluent quality from DBPs systems.

Parameter	**Equation**	**Rate Constants (for depth≥ 0.6 m)**
COD	$Le = Lie^{-kt}$	$K_1 = 0.131(1.065)^{T-20}$
TSS	$Se = Si((-1.18/T)Ln(t)+(6.5/T))$	----
NH_4^+	$Ce = Ci0.640e^{-knt}$	$K_n = 0.137(1.009)^{T-20}$
PO_4^{3+}	$Pe = Pi0.8485e^{-kpt}$	$K_p = 0.012(1.491)^{T-20}$

Source: Smith and Moelyowati (1999).

Where,

Si and Se: Influent and effluent concentration (mg/L).
T: Water temperature (^oC).
t: Time (days).

2.5 Capacity of ABPs and DBPs in nitrogen removal

Higher nitrogen removal in ABPs compared to DBPs was reported in literature. This was mainly attributed to ammonia volatilization during periods of high temperatures and pH (Pano and Middlebrooks, 1982). For example, Silva (1982) obtained ammonium removal efficiency in ABPs of 81%. Similarly, Zimmo (2003) who tested the effect of low, high temperatures and organic load on nitrogen removal reported that during the low organic loading period and warm temperature, nitrogen removal efficiency was higher in ABPs (80%) than in DBPs (55%) despite the fact that approximately one third of the influent nitrogen to the DBPs is removed via duckweed harvesting. Table 2.3 shows Nitrogen uptake by duckweed as reported in preceding studies. However, the algal biomass in the system effluent cannot be easily harvested and consequently nutrients are released again in the environment upon degradation. On the other hand, one of the major limitations which have been mentioned in literature for DBPs is the reduced organic matter removal capacity compared with that of conventional ponds, because of lower oxygen concentrations (Reed *et al.*, 1995). Consequently, an integrated system combining ABPs and DBPs could be a proper solution for nutrients and Faecal Coliform (FC) removal at reasonable land requirement (Steen *et al.*, 1998).

Nitrogen removal from the ABP system and DBP system was described in many studies by the exponential relation: $N = N_I e^{-kt}$, (N_I: initial nitrogen concentration, N: nitrogen concentration at any time (t) along the line of treatment, *k*: removal coefficient). Table 2.3 shows the result of some previous studies on nitrogen and phosphorus removal in different countries. Zimmo (2003) found that N-removal rates in ABPs seem to correlate with BOD loading rates with a correlation coefficient (R^2) of 0.65 and with N-loading rates (R^2=0.80), while in DBPs N-removal rates were relatively constant irrespective of BOD loading rates or N-loading rates. Overall nitrogen removal was higher during warm temperature in both ABPs and DBPs.

Table 2.3. Nitrogen uptake by duckweed as reported in preceding studies.

Region	Species	Daily removal ($g/m^2.d$)	Reference
Louisiana	Duckweed	0.47	Culley *et al.* (1978)
Italy	*L.gibba/L.minor*	0.42	Corradi *et al.* (1981)
USA	*Lemna*	1.67	Zirschky and Reed (1988)
India	*Lemna*	0.5-0.59	Tripathi *et al.* (1991)
Minnesota	*Lemna*	0.27	Lemna corporation
CSSR	Duckweed	0.2	Kvet *et al.* (1979)
Bangladesh	*Spirodela polyrrhiza*	0.26	Alaerts *et al*. (1996)
Yamen	*Lemna*	0.05-0.2	Al-Nozaily (2001)
Palestine	*Limna gibba*	1.31	Zimmo (2003)

Source: (Zimmo, 2003)

2.6 Main nitrogen removal pathways and mechanisms in ABPs and DBPs

2.6.1 Nitrogen removal pathways

Nitrogen compounds are reduced from DBPs and ABPs by sedimentation of SS with organic nitrogen, ammonia volatilization and nitrification and denitrification. Moreover, Zimmo (2003) reported that, N-removal by different mechanisms in both systems (ABPs and DBPs) appeared more dependent on the pH variations than on oxygen variations. At high pH range (7-9), N-removal efficiencies were significantly higher compared to low pH range (5-7) in both systems irrespective to oxygen condition. Moreover, at high pH values of 7 to 9, the increase in N-removal by sedimentation and volatilisation in the algae system and the decrease in N-uptake by duckweed in the duckweed system which result in significantly higher N-removal efficiency in the algae system than the duckweed system (Zimmo, 2003).

Sedimentation is considered as one of the important fluxes for nitrogen transformations. Literature shows that the amount of sediment is dependent on pH value (Zimmo, 2003). Sedimentation in the form of particulate organic nitrogen (probably decaying algae biomass), was considered as the largest nitrogen flux in ABPs. The amount of algae growth is therefore an important factor in determining the amount of nitrogen accumulated in the sediment (Zimmo, 2003). Ferrara and Avci (1982) found that sedimentation was the main removal pathway. On the other hand, in DBPs with anaerobic pre-treatment, sedimentation was of minor importance because almost 90% of the nitrogen in the influent was in the NH_4^+ form (Caicedo, 2002). Consequently, the total contribution of sedimentation to the removal can be calculated as about 6% (Steen *et al.,* 1998). Therefore, the smaller amount of sediment produced in duckweed ponds in comparison with algae ponds will result in less frequent desludging requirements (Zimmo *et al*., 2000).

2.6.2 Nitrogen removal mechanisms

2.6.2.1 Nitrification and denitrification in ABPs and DBPs

Nitrification and denitrification processes are considered as major nitrogen removal mechanisms. Autotrophic nitrification is achieved in two steps that are catalyzed by different species of nitrifying microorganisms:

$$NH_4^+ + 1.5\ O_2 \xrightarrow{\textit{Nitrosomonas}} NO_2^- + 2H^+ + H_2O \qquad (1)$$

$$NO_2^- + 0.5\ O_2 \xrightarrow{\textit{Nitrobacter}} NO_3^- \qquad (2)$$

Nitrification represents the bottleneck for nitrogen removal via denitrification. According to literature, many conditions affect the efficiency of microorganisms responsible for the nitrification and denitrification processes; autotrophic nitrification for example, is affected by temperature, pH and DO. Nitrification takes place in all locations where DO values were higher than the critical threshold of 0.5 mg/L (Taylor and Bishop, 1989). The optimum temperature for occurrence of nitrification is between 25 to 35 ^oC and the optimum pH is between 7 to 8 (Metcalf and Eddy, 1991). However, heterotrophic nitrification in the sediment could take place by heterotrophic nitrifying bacteria that can oxidize both inorganic and organic nitrogen compounds (Van Luijn 1997; Verstraete and Alexander, 1973) at low oxygen concentration (Laurent, 1971).

Nitrifiers are known to prefer attachment to solid surfaces (Focht and Verstraete, 1977; Underhill and Prosser, 1987; Verhagen and Laanbroek, 1991). In DBPs, the presence of a duckweed mat provided additional surface for the attachment of nitrifiers that in role enhances nitrification process. Consequently, low volume to surface/mat ratio has also similar effect (Zimmo, 2003). Practically, this fact has to be considered in the design of DBPs, because, the intensive harvesting can reduce the possibility that nitrifying bacteria would thrive on plants' surface or root zone (Alaerts *et al*., 1996).

For anaerobic pretreatment conditions, complete ammonification of the organic nitrogen occurs in the anaerobic reactor (Steen *et al.*, 1998). Nitrification and ammonia volatilization can occur in the oxidized root zone of duckweed, while denitrification occurs in the reduced environments in the water column or in the sediments. Because of low depth of the aerobic layer, the nitrification step of the coupled nitrification /denitrification process is the rate limiting one (Caicedo, 2002). Conversely, Zimmo *et al*. (2002) mentioned that the presence or absence of DO was associated with the presence of NO_3^-. For the ABPs systems, pH is correlated with high algae growth that eventually contributes to removal, either via sedimentation or by providing substrate for attachment of nitrifying bacteria in pond water enhancing nitrification and subsequently denitrification.

Contradictory results were reported about the importance of this process as a nitrogen removal mechanism. Ferrara and Avci (1982) stated that nitrification does not occur in stabilization ponds under normal conditions. In some studies on macrophyte systems, denitrification was referred to as unaccounted for nitrogen and found to contribute for 10-20% to the nitrogen loss (Alaerts *et al.,* 1996; Hanif, 1998; Steen *et al.,* 1996). However, Zimmo (2003) claimed that low concentrations of both nitrite and nitrate (3.1 mg-N/L and 3.6 mg-N/L respectively) do not prove that nitrification does not occur in ponds and in fact this may be due to efficient denitrification. Similarly, Van Luijn (1997) stated that this mechanism could account for considerable N-removal (65% of the nitrogen removal by denitrification).

2.6.2.2 Nitrogen volatilization in ABPs and DBPs

The optimum pH and temperature for nitrogen volatilization occurrence are 9.4 and 20^0C, respectively (Awadallah, in preparation). Contradictory conclusions were drawn on the importance of ammonia volatilisation. Some researches considered ammonia volatilization is the main pathway for nitrogen. For example, Pano and Middlebrooks (1982), in accordance with King (1978), Silva *et al*. (1995) and Soares *et al*. (1996), argued that ammonia volatilisation largely explains total nitrogen removal from ponds. Steen *et al.* (1998) reported that in ABPs, the raised pH shifts the equilibrium between ammonium and NH_3 towards the latter. Nevertheless, no complete nitrogen mass balance was carried out in these studies. Recently, Zimmo (2003) found through experimental evidences that ammonia volatilization in ABPs and DBPs was found to be a minor N-removal mechanism responsible for less than 1.1 % of total influent nitrogen. However, the main pathway for nitrogen was through nitrification and denitrification processes.

2.6.2.3 Incorporation of nitrogen into biomass

In pond system, N-removal can occur via uptake by biomass attached to container wall and in the sediment. However, DBPs are able to remove nitrogen through new pathway, which is uptake of nitrogen by duckweed species (Iqbal, 1999). Nitrogen removal through uptake by duckweed ranges from 19-33% of total nitrogen, depending on the organic loading rate and temperature (Zimmo, 2003). Nitrogen removal in DBPs is therefore influenced by the plant species and density of plants. Frequent harvests are necessary to sustain high levels of nitrogen removal (Reed, 1988; Zimmo, 2003).

2.6.2.4 Nitrogen fixation

Zuberer (1982) and Duong and Tiedje (1985) demonstrated nitrogen fixation associated with duckweed covers. In this process, atmospheric nitrogen is reduced to NH_4^+, which is subsequently incorporated in biomass. Nitrogen fixation usually does not occur if ammonia is present in the environment because in the presence of ammonia nitrogenase synthesis is suppressed by a phenomenon called the "ammonia switch-off" effect (Brock *et al.*, 1991). Duong and Tiedge (1985) reported N-input in naturally occurring duckweed-cyanobacterial associations of 1-2 mg-N /m^2.d. This value is low compared to the total amount of nitrogen fed to the treatment ponds via the wastewater (possibly 1-2 g-N/m^2.d). N-fixation therefore is not likely to affect the overall nitrogen balance in wastewater treatment ponds (Zimmo, 2003).

2.7 ABPs and DBPs removal capacity for other parameters

2.7.1 (Bio) chemical oxygen demand (BOD&COD) removal

The major factor responsible for BOD and COD removal is O_2 production, which is the same for facultative stabilization ponds. Duckweed plants create the environment for treatment but contribute very little directly to the removal of BOD (Reed, 1988). Conversely, Zimmo (2003) who compared the efficiency of ABPs and DBPs reported that better removal of BOD and COD in DBPs. Similarly, Körner *et al.* (1998) concluded from laboratory studies with *Lemna gibba* that heterotrophic uptake of small organic compounds is not important. Nevertheless, they found that COD removal was significantly faster in the presence of duckweed than in uncovered controls. This result could be attributed to lower algal development and better sedimentation due to effect of shading and quiescent conditions provided by duckweed cover. The effect of anaerobic pretreatment on organic matter removal was investigated by Caicedo *et al.* (2002) who found that the stabilization ponds with and without anaerobic pretreatment were efficient in removing organic matter. However longer HRT was required for the ponds without pretreatment in order to reach a removal efficiency equals to the effluent of the anaerobic pretreatment unit (UASB reactor) which was 82% of COD and 92% of BOD.

2.7.2 Total suspended solids (TSS) removal

TSS is mainly reduced by the process of sedimentation and biodegradation of organic matters. For achieving effective sedimentation, it is important to increase the retention time of slowly degradable organic matter. Suspended solids removal in a duckweed-covered basin should be more effective as compared to conventional stabilization ponds due to the control of algal growth and the improved quiescent conditions under the surface of mat (Reed, 1988). Moreover, TSS can also be removed in DBPs by the adsorption on the root system. Fractions of adsorbed solids are degraded and part of degraded products may be assimilated by the plants. DBPs system can consistently bring final effluent (TSS) to below 5mg/L (Skillicorn *et al,* 1993). Velocity and HRT plays an important role in removal of suspended solids, consequently the purpose of sedimentation pond prior to duckweed based wastewater treatment lagoons is to reduce the suspended solids in the raw wastewater, and this prevents excessive sludge accumulation in duckweed pond. Bonomo *et al*. (1996) tested TSS removal in DBPs. They reported that TSS removal efficiency was good in the whole experimental period (50-80%) TSS effluent concentration ranged between 26 to 54 mg TSS/L. Similar removal efficiency (80%) of TSS has been reported in literature for DBP systems (Mandi, 1994; Bonomo *et al.*, 1996; Zirschky and Reed, 1988).

2.7.3 Pathogen removal

Pathogen die-off results from complex interactions of several factors such as light radiation, depletion of nutrients, microbial antagonism. The presence of antibacterial substances produced by algae and high oxygen concentrations (Polprasert *et al.,* 1983; Pearson *et al.*, 1987; Saqqar and Pescod, 1992). Nevertheless, bacterial pathogens would likely be removed to somewhat lesser degree in duckweed ponds than algae ponds because of the restricted sunlight penetration and the absence of very alkaline conditions, which occur during the daytime in algae ponds. On the other hand, ABPs systems can effectively reduce pathogen counts to a level low enough for the effluent to be used for restricted irrigation (WHO, 1989). Similar result was achieved by Zimmo (2003) who reported that the environmental conditions in the DBPs were not favorable for pathogen decay, due to reduced light penetration. Consequently, DBPs are able to satisfy the WHO guideline at higher HRT (21 days during the summer and more than 28 days during winter) in comparison with ABP system (Zimmo, 2003; Iqbal, 1999; Steen *et al.,* 1999). Accordingly, if the effluent from the duckweed ponds must comply with microbiological guidelines for reuse for agricultural irrigation, then algae ponds should usually be included in the treatment system, which could reduce land required to achieve guidelines for reuse (Steen *et al.,* 1999).

2.7.4 Heavy metals removal

Heavy metals can be reduced from DBPs by sedimentation as sludge, plant uptake (copper and arsenic) and absorption during polishing processes. However, heavy metals can influence the performance of treatment in DBPs if concentrations of iron, zinc, aluminum, chromium, and copper in the wastewater are higher than 20 mg/L, 20 mg/L, 0.1 mg/L and 1 mg/L, respectively (Boniardi and Rota, 1998). Duckweeds could be specifically applied as bio-accumulators for heavy metals, although the information about their capacity to remove heavy metals from wastewater is still limited (Doyle, 1977; Clark *et al.,* 1981; Charpentier *et al*., 1987). However, Clark *et al*. (1981) found that after bioaccumulation duckweeds are also able to release Cu and Cr again when placed in a metal free solution. The maximum heavy metals removal capacity depends on local aqueous concentrations as well as on growth and harvesting conditions of duckweed, because equilibrium is established between the metal concentrations outside and inside the plants. Cr can be better removed from the solution than Cu (75-100% against 35-40%). Nevertheless, bioaccumulation is toxicity exerted by the heavy metal, especially Cu (III). Therefore, this limitation for duckweed has to be taken into consideration when the harvested duckweed is to be used as fodder for fish and animals (Abdel-Wahhab *et al*., 1995).

2.8 Design considerations

Adequate primary treatment of raw wastewater is indispensable prior to duckweed treatment (Bonomo, 1996). Anaerobic pretreatment in earthen sedimentation ponds with a clay lining or closed settlement tanks are a good option for primary treatment. Duckweed treatment systems can both be designed and operated as plug-flow or batch systems. Continuous flows through lagoons are suggested for medium-scale applications at community or urban level. Ponds operated as batch reactors are commonly encountered at village-level. Optimum water depths are reported between 0.4 and 1 m (Smith and Moelyowati , 1999). Plug-flow design should allow a HRT of at least 20 days with a length to width ratio of 1:10 or more. In general, a narrow pond design is more suitable as it allows operational work to be carried out from the pond perimeter and avoids direct contact of workers with the wastewater (Awadallah, in preparation). A floating bamboo or plastic containment grid system is required to prevent the plants from drifting to the shore by the action of wind and water current (Gijzen, 1997).

2.8.1 Hydraulic retention time

HRT is a determining factor for the removal efficiency in WSP systems (Iqbal, 1999). Increasing the retention time of slowly degradable organic matter was reported to be important for achieving effective sedimentation and nitrogen removal. HRT for reducing organic matters depends on the influent BOD, but 10 to 20 days is acceptable to reduce BOD to 30 or 20 mg/l. Best results for filtered BOD_5 removal were obtained at HRT of 20 days. However, effluent with a HRT of 10 days was also suitable for irrigation purposes (Oron and Porath, 1987). Furthermore, Silva (1982) obtained ammonium removal efficiency of 81% in a system of (1.0 m) depth and HRT of 29 days. Similarly, Middlebrooks *et al.* (1982) reported higher removal values in systems with very long hydraulic retention times of 227 days. Zimmo (2003) found that annual nitrogen removal efficiencies in ABPs and DBPs of 0.9 m depth were respectively 73% and 54% after 28 days retention time. HRT in WSPs is a function of the seasonal and climatic conditions. Experiments on duckweed for example showed that needed HRT for nutrient removal in winter is 3 times more than in summer (Wildschut, 1983). Zimmo (2003) reported that DBPs is able to satisfy the WHO guideline at 21 days during the summer and more than 28 days during winter.

On the other hand, the longer the retention time causes more anaerobic conditions and lower protein content of duckweed produced which deteriorates the nutritional value of the harvested duckweed. Oron and Porath (1987) reported that Protein contents of 25% was found at a HRT of 5 days were large, dark plants with small roots were observed. This was the indication of favorable growth conditions. However, this percentage was reduced to 15% at HRT of 20 days where small but pale green plants with long roots were found probably due to shortage of nutrient supply. Longer HRT in DBPs system requires fewer days to obtain steady state conditions. Once the system reaches steady state, at shorter HRT the appearance of the plant is healthier. Consequently, an optimization is required between the HRT, effluent quality and nutritional value of duckweed. In fact, these variables depend also on the regional climatic conditions, type of reuse and the legislations for effluent quality.

2.8.2 Organic loading rate

Organic loading rate (OLR) plays a major role in determination of the biological and “complex” physicochemical processes occurring in conventional stabilization ponds (Metcalf and Eddy, 1991). Organic loading rate was reported to affect FC and TSS removal efficiency. However, organic loading rate was reported to have no effect on total phosphorus (TP) and total nitrogen (TN) removal efficiency in DBPs and ABPs (Zimmo, 2003). Moreover, Steen (1998) reported that no effect was noticed on duckweed growth rate when COD_{tot} was changed. Likewise, Caicedo (2002) found that efficiency of DBPs was almost constant in removal of COD and BOD regardless to OLR variation and anaerobic pretreatment.

2.8.3 Land Requirement

The major limitation for the application of algae-based or duckweed-based ponds technology for wastewater treatment at large scale is the large area needed, which is estimated at about 5-7 m^2/capita (excluding associated facilities) depending on influent strength of wastewater and effluent guideline requirements (Mara and Pearson, 1998). However, optimization of the ABPs and DBPs systems may reduce the area requirement considerably making these systems more attractive (Zimmo, 2003). In reality, ponds are designed such that the effluent complies with WHO (1989) microbiological quality guidelines for restricted irrigation (<1,000 FC/100 ml). One of the methods suggested to maintain smaller HRT with smaller land requirement, is using deeper ponds. The advantages of treating urban wastewaters in deep ponds (>3m), as opposed to traditional shallow ponds, lie in the smaller area of land needed, the greater capacity for water storage and regulation and the smaller water losses through evaporation, which is an important consideration when the treated wastewater will be used for irrigation. However, this issue still needs to be further studied. The feasibility of conducting wastewater treatment in deep ponds has been proven in several studies carried out in the region of Murcia, SE Spain. This region is characterized by a Mediterranean climate (Soler *et al.*, 1981, 1988, 1991; Moreno, 1984; Berna *et al.*, 1987; Moreno *et al.*, 1988).

2.9 Contribution of microorganisms in treatment

There is a variety of microorganisms in wastewater treatment systems that contribute in treatment. DBPs and ABPs have the capability to purify wastewater in collaboration with both aerobic and anaerobic bacteria, depending on the organic load and oxygen input (Steen *et al.,* 1998). In WSPs three zones in the water column could be distinguished, these are the aerobic zone (Skillicorn *et al*., 1992), the anoxic zone, and the anaerobic zone. In the aerobic zone, organic materials are oxidized by aerobic bacteria using atmospheric oxygen transferred by duckweed roots. Heterotrophic nitrification and denitrification take place in anoxic zones, where organic matter is decomposed by anoxic bacteria into ammonium and *ortho*-phosphate, which are intermediate products used as nutrients by the duckweed (Metcalf and Eddy, 1991). Organic matter in the bottom of the ponds is decomposed by anaerobic bacteria and this produces gases such as carbon dioxide (CO_2), hydrogen sulphide (H_2S) and Methane (CH_4) (Metcalf and Eddy, 1991).

2.10 Duckweed growth conditions

Growth rates of duckweed systems are function of water temperature, wastewater composition (Landolt, 1986), procedure for plant harvesting and HRT (Oron and Porath, 1987). Conditions affecting the growth of duckweed accordingly, can be divided into environmental conditions and wastewater characteristics. Nevertheless, stress factors on duckweed growth are nutrient scarcity (Edwards *et al.,* 1992) toxins, (DeBusk and Ryther, 1981), extremes of pH (Zirusky and Reed, 1989) and temperature (Oron and Porath, 1987), crowding by overgrowth of the colony (DeBusk and Ryther, 1981) and competition with other plants for light and nutrients (Wildschut, 1993).

2.10.1 Environmental conditions affecting growth of duckweed

2.10.1.1 Temperature

Duckweed is more tolerant to temperature than water hyacinth. A minimum temperature of 7 ^{0}C has been suggested as practical limit for growth of duckweed (Reed, 1988). *Lemna gibba* can be grown at lower temperature (1 to 3 ^{0}C) (Wildschut, 1983). Some species of *Lemnaceae,* such as *Limna gibba,* can vegetate at temperature from -3 ^{0}C up to 30 ^{0}C (Oron *et al.*, 1985). Optimal temperature range for maximum growth is reported to be 17.5 to 30 ^{0}C with upper tolerance limit of 34 ^{0}C .A mat of duckweed heats up in sun faster than a water column below it, sometimes difference at several centimetres below approaches 8 ^{0}C (Javed, 1995). A full thick mat of duckweed plants may have a temperature of about 10 ^{0}C above the ambient air temperature due to radiation effects (Zischky and Reed, 1989). Nutrient content in duckweed are lower in summer due to high growth rates (more biomass/unit area) and higher in winter due to slow growth rate and luxurious uptake (Reddy and DeBusk, 1985). Acute heat stress can be managed by spraying water or physically immersing the crop. Shading with bamboo and banana trees, or taro plants can also moderate temperature (Javed, 1995).

2.10.1.2 Light intensity

The plants can grow in full sunlight as well as in dense shade. Different species of duckweed have different behaviour with respect to the light conditions. *Wolffia* does better under dark conditions, whereas *Limna gibba* does better in sunlight (Zirschky and Reed, 1988). Moreover, it has been reported that *Lemnaceae* are able to use carbohydrates as its energy source under non-saturated light condition (Landolt, 1986). It is also known that *lemnaceae* can be grown in complete darkness if organic substances such as sugars are added to the nutrient solution (Landolt, 1986). Moreover, Steen (1998) reported that the very intense radiation of the Negev desert (300-600 W/m^2 average daily global radiation) did not reduce the duckweed production.

2.10.1.3 Wind diffusion

The main problem of wind diffusion with duckweed is the fact that wind may blow it to the sides of the pond and algal growth would takeover in the cleared water. As duckweed mats are susceptible to the wind, floating booms or cells hold the plants in place.

2.10.1.4 Algal activity and toxins

Algae are the primary competitor of duckweed for nutrients as it grows faster. Algae dominance results in diurnal high pH and production of free ammonia which can become toxic to duckweed. Filamentous algae are more harmful to duckweed as they wrap themselves around the plant roots causing the fronds of duckweed to shrivel and finally to die (DeBusk *et al.*, 1976; Lin, 1982). Extracts of some blue green algae are known to inhibit growth (Entzoroth *et al.*, 1985). It was also observed that cyanobacterin which is released by the blue green algae inhibits the growth of *Lemna* (Landolt, 1986). One of the practical solutions for inhibition of growth of algae is to maintain relatively high density of duckweed with daily harvest reducing open areas of open water.

2.10.2 Wastewater characteristics

Duckweed grows on a wide range of quiescent or slow-current waters, and relatively polluted waters, saline waters and eutrophic water bodies (Oron *et al.,* 1986). However, in addition to the aforementioned external factors, duckweed growth is highly affected by the concentrations of pollutants in wastewater. The following discussion comprises criteria of wastewater characteristics in which duckweed can survive normally.

2.10.2.1 Nutrients concentration

Growth rate of duckweed decreases when TKN and P concentration were less than 3 mg/L and 0.3 mg/L, respectively. However, the growth rate of duckweed was shown to be independent of N and P if higher than 10 and 2-3 mg/L, respectively (Edward and Hassan, 1992; Rejmankova 1982; Skillicorn *et al.*, 1993). Economically, the lack of nutrients affects the nutritional value of duckweed as plant protein drops to 10% (Javed, 1995). On the other hand, High loadings of nutrients (ammonia in particular), surfactants and compounds with herbicidal properties affect the growth rate. Especially surfactant can dissolve duckweed protective waxy coating making plants more vulnerable to fungal infection (Skillicorn *et al.,* 1993).

2.10.2.2 pH value for wastewater

Zirschky and Reed (1988) reported that typical pH range is 4.5-7.5, though duckweed growth is completely inhibited only at pH greater than 10, while Landolt (1986) found that duckweed could survive at pH of 5 to 9 but grows best in 6.5 to 7.5. When pH is less than 7, ammonia can be in its ionized state as NH_4^+ which is the preferred form of nitrogen for the plant. According to Wildschut (1983) duckweed can tolerate a wide range of pH from 3 to 10 but for optimal growth rate it should be 5 to 7. The complete duckweed cover suppresses the algae which suppresses the elevating effect on pH.

2.10.2.3 Ammonium and ammonia concentration

NH_4^+ is the preferred form of nitrogen for duckweed over NO_3^- as their source of nitrogen (Landolt, 1986). NH_4^+ is available in the wastewater as about 65% of TKN. It is produced as a result of hydrolysis of organic nitrogen at anaerobic conditions. In aerobic conditions, NH_4^+ is oxidized to NO_3^-, while in anaerobic conditions; the nitrogen balance is transformed in favor of NH_4^+ over NO_3^-. Toxicity of NH_4^+ is pH dependent; as pH goes high, the % fraction of NH_3 of total (NH_4^+ + NH_3) in the water increases, for instance at pH of 8, only 5% is NH_3, while at pH of 9.2 results 50% of NH_3 (Koning *et al.,* 1987). Moreover, Caicedo (1995) reported that the NH_4^+ itself appeared to be the inhibitor specially at low pH, because when the plant absorb NH_4^+ it breaks into NH_3 to be absorbed and H^+ stay out of the cell in the solution, and hence not the NH_3 is the inhibitor but H^+ (low pH).

Rejmankova, (1979) stated that duckweed can tolerate N- concentration up to 375 mg/L, however, it is not specified that the nitrogen was in the form of NH_4^+. However, Zimmo (2003) claimed that NH_4^+ concentration of 200 mg/L is too high for *Lemna Gibba*. High concentration of NH_4^+, makes the colour of plants darker, however the plants and roots get smaller at high concentration of NH_4^+ (Wildschut, 1983). At NH_3-N concentrations of 3 mg/L, the growth rate was inhibited by 20% and at 7.16 mg/L about 50% (Wang, 1991). Plants in prolonged direct contact with NH_3 gas will usually die rapidly (Ghosh, 1994). Likewise, Zimmo (2003) found that NH_3 concentrations of 5 mg-N /L and higher caused the death of duckweed within 2 days of its exposure to that concentration.

Comparable values of ammonia toxicity to duckweed were reported in studies of Wang (1991) and Clement and Marlin (1995) whereas Caicedo *et al*. (2000) reported ammonia toxicity for *Spirodela polyrrhiza* already at lower values. The latter reported that a negative effect on growth of *Spirodela* at higher concentration than 50 mg (NH_4^+ + NH_3)-N/L.

2.10.2.4 COD concentration

Mandy's experiment proved that DBPs systems tolerate maximum influent COD concentrations from 300 to 500 mg/L (Mandi, 1994) but lacks the ability to grow on undiluted domestic wastewater or industrial wastewater (COD 1667 mg/L).

2.10.2.5 Dissolved oxygen (DO) concentration

DO in wastewater is one of the most important parameters that determines the wastewater physicochemical and biological characteristics, correspondingly aeration may increase the treatment performance, but unless it is managed carefully it will disturb the duckweed mat in DBPs and generate open areas, which will reduced the removal efficiencies (Smith and Moelyowati , 1999).

2.10.2.6 Heavy metals concentrations

Duckweeds can be applied as bio-accumulators for heavy metals, although the information about their capacity to remove heavy metals from wastewater is still limited (Doyle, 1977; Clark *et al.,* 1981; Charpentier *et al.,* 1987). Abdel Wahaab *et al*. (1995) reported the uptake rates of heavy metals by duckweed mat were 80-333 and 250-667 mg/d.m^2 for Cu and Cr (III), respectively. He concluded also that Cu is toxic at 1.0 mg/L, resulting in plant death after 8 days. Clark *et al*. (1981) reported that exposure to the high Cu level, resulted in plants started decaying (Yellowing of fronds) releasing part of the Cu again.

2.11 Harvesting of duckweed

The duckweed biomass can be easily harvested by skimming the plants from the surface. In large systems, this could be mechanized, but in smaller ponds, it could be done manually by a simple kind of fork. Seed stock should have a density of 600 to 900 gm wet-weight/m^2 (Skillicorn et *al.,* 1993). PRISM (1992) also suggests based on extensive field experimentations for an initial stock density of at least 600 gm wet weight/m^2. However, it is important to avoid crowding of duckweed after passing the startup period because crowding decreases the doubling rate of the colony. Crowding reduces not only the crop growth rate but also the average age of the frond population, which weakens the resistance of the colony to be attacked by the predators like aphids, snails or fungi. The dense crop average also reduces DO and suppresses nitrifying bacteria. However, once the duckweed recovers its optimum density, the algae disappear rapidly (Oron and Porath, 1987). The aim of biomass production system is to generate a valuable by-product. This could increase the economic feasibility of treatment schemes, especially in developing countries. It is therefore essential to optimize the duckweed production per unit area, rather than nutrient removal. Ponds with long retention times are expected to have low nutrient concentration, and therefore reduce duckweed production (Whitehead *et al*. (as quoted by Steen, 1998), 1987; Alaerts *et al*., 1996).

2.12 Mosquitoes and odor control

Mosquitoes, larvae and odor development are considered also as other limitations for WSPs. The large volume porous foliage of the water hyacinth plant above the water surface produces excellent conditions for mosquito larva development (Oron *et al.,* 1985). However, in case of duckweed the situation is entirely different. As long as a thick surface mat is maintained, mosquito larvae will not be able to penetrate a fully developed duckweed mat and cannot survive in the anaerobic water beneath the surface cover. Consequently, it cannot create a problem (Anon, 1989; Reed, 1988; Wildschut, 1983). This fact is considered on of the advantages of DBPs over ABPs. However, the dense cover of duckweed prevents oxygen from entering the water by diffusion. The fact, together with the lack of photosynthetic oxygen production by phytoplankton, makes the water largely anaerobic (Brix, 1991; Culley and Epps., 1973) which may increase the opportunity for odor creation.

Chapter Three

Materials and Methods

3.1 Experimental set-up

In order to perform this study a pilot scale treatment plant was constructed in Al-Bireh Wastewater Treatment Plant at Al Bireh city 15 Km north-east Jerusalem-West Bank-Palestine. This study was performed during the period from May to November 2004. The study was intended to test the effect of depth on nitrogen removal in DBPs only. However, after detection of duckweed death in the ponds, the system was altered into ABPs system. The first period extended from the beginning of the experimental period until the fourteenth week of the research (2/5/2004 to 18/8/2004). However, the second period was from the fifteenth to the twenty-fourth week of the experimental period (18/8/2004 to 31/10/2004). The selected duckweed species was *Lemna gibba*. The climatic conditions in the two experimental periods were considered as mostly summer climates (range of temperature was from 20 to 30 ^{o}C), humidity 60-70%, radiation 150-260 Watt/m^2, and wind speed is 2.3 to 2.6 m/s (ARIJ, 1996).

3.2 Source of duckweed

Limna gibba species were collected from a pilot DBP system located in the campus of Birzeit University-Palestine. The mentioned DBPs system consists of four ponds laid out in series. The dimensions of each pond are 1m width, 3m length and 0.9m depth. They were fed with institutional wastewater from the university campus. *Lemna gibba* was collected from these ponds equally throughout the periods from May to the beginning of July. These ponds were used as pilot treatment plants for previous studies, but they were left since at least one year without operation or maintenance.

3.3 Source of wastewater

The influent wastewater to the pilot plant was pumped from the grit removal chamber of Al-Bireh wastewater treatment plant (AWWTP). This wastewater is conveyed via sewer system from Al-Bireh city. This wastewater is classified as municipal wastewater because it is conveyed by the sewer system from Al-Bireh city mixed with some industrial wastewater from small factories and from sewage vacuum tankers that dispose their collected load from different sources in the near manholes leading by sewer line to the mentioned treatment plant.

COD concentration in the influent sewage from the grit chamber at AWWTP was measured several times during the research experimental period, grab samples were analyzed every month. The average value for COD was 1275 ± 84 mg/L. Nevertheless, industrial wastewater shock loads were received by the treatment plant either through sewer system or through sewage vacuum tankers that discharge their loads in the near manholes of the main sewer line leading to the grit chamber at AWWTP. The impact of industrial wastewater shock loads were clearly noticed through the notice of change of wastewater color, and the sudden temporarily shift in COD value and other parameters values of the effluent from the UASB-septic tank and the receiving stabilization ponds.

3.4 System description

The pilot scale treatment plant (Figure 3.1) consists of a UASB-septic tank of 4 days HRT (250 cm height, 64 cm diameter, and 0.8 m^3 volume). This reactor acted as a pretreatment unit for the influent wastewater from the influent line of AWWTP, it had been started up by Al-Shayah (2005). Three parallel lines of aluminum circular tanks, each consists of three equal tanks with a total HRT of 28 days. The pretreatment and post treatment units were interconnected by a (160 L net volume) holding tank in which the effluent of the UASB is received. The upper side of the holding tank is connected to the overflow line from which the excess pretreated wastewater returns back to the grit chamber. The bottom of the tank is connected to the pumping unit (two pumps) "MasterFlex Consule analog L/S". These pumps have a variable speed; they have peristaltic motor drive, used with pump head and peristaltic tubing. One pump was pumping to the deepest ponds line (with a flow rate of 59.0 L/day). The second pump was calibrated to pump to the other two lines at a flow rate of 39.0 and 20.0 L/day, respectively. The flow rates of the pumps were recalibrated twice a week in order to maintain the same preset flow in the corresponding lines.

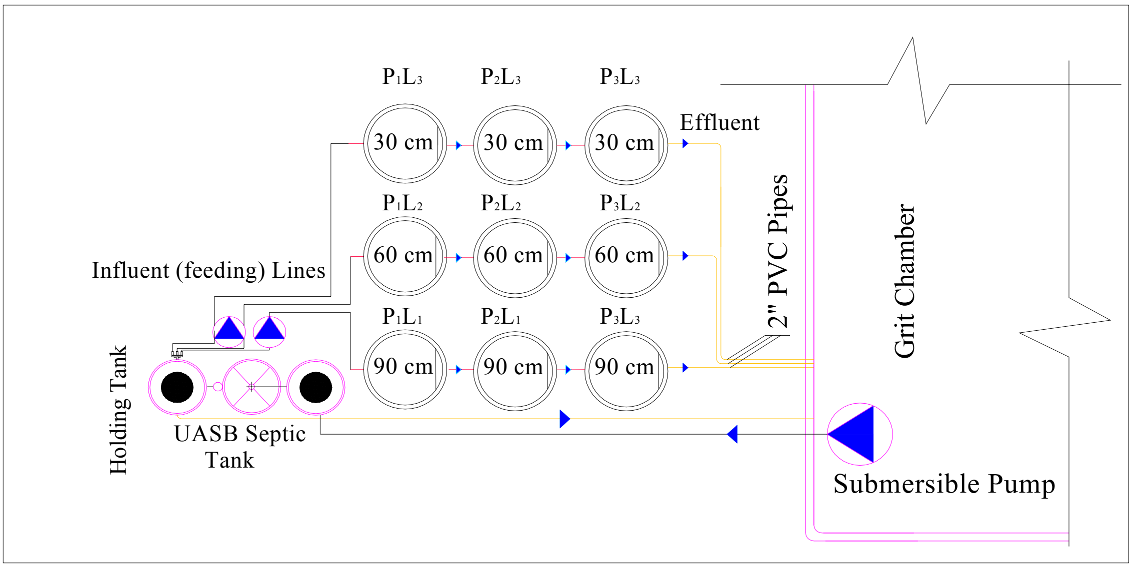


Fig. 3.1 Schematic diagram for the stabilization ponds based pilot treatment plant; the figure shows the suction location, required pumps, holding tanks, UASB-septic tank, and the three lines of stabilization ponds. The figure shows also each pond with its' corresponding code (which is used later to describe the pond) and depth.

The circular tanks (ponds) were made from a metal in order to maintain water tightness. They were arranged in three parallel lines with three ponds of the same size (depth) for each line. The depths of the first, second and third line ponds were 90, 60, and 30 cm, respectively. All of the ponds were made in a similar diameter (88 cm); the selection of the ponds dimensions based according to the available materials in the market. Their material was selected in order to simplify their transportation after manufacturing. The range of error in dimensions was considered in order to maintain a similar HRT (28 days). Baffles at the outlet of each pond were installed in order to reduce short-circuiting and prevent the transfer of floating materials to the consecutive ponds. Moreover, the inlet pipe to each pond was mounted to feed at 15-20 cm above the bottom of the pond. Finally, the effluent of the ponds is conveyed by PVC pipes back to the grit chamber (by gravity). Figure 3.2 shows a typical pond with some key details.

Batch experiments

In parallel with period one and two, batch experiments were performed for testing duckweed survival in different dilutions and for investigating the effect of thermal and radiation stress. These tests were performed using a number of similar white color buckets of 27 cm length and 26 cm diameter and 15L volume, moreover two cultures were maintained during the experimental periods of 40 cm length and 34 cm diameter, these cultures were maintained at 1:1 and 3:1 pretreated wastewater concentrations (wastewater: fresh water), respectively.

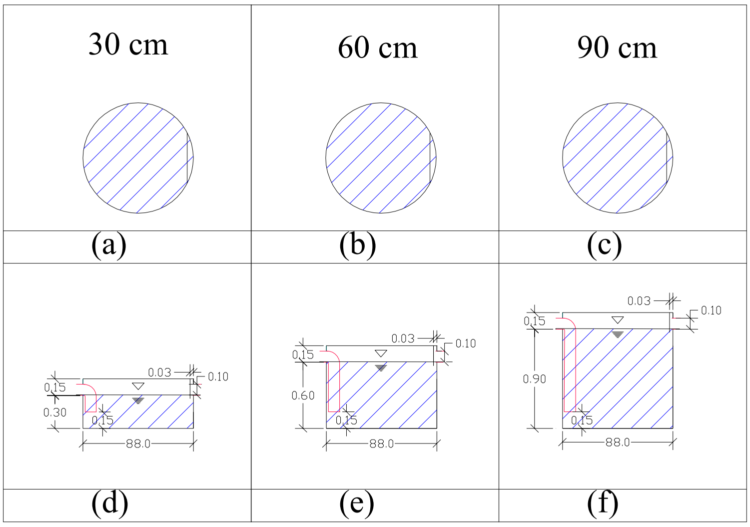


Fig. 3.2 Top view for the ponds of depth 30, 60, and 90 cm, respectively (a,b and c) and the dimensions of the above mentioned ponds(d,e and f). All dimensions are in meter otherwise mentioned.

3.5 Main research periods

The overall research period was 24 weeks. In the first 14 weeks, the ponds were operated as duckweed based ponds, while in the other 10 weeks, the ponds were converted to algae based ponds. The operational conditions were maintained the same for the two periods, the meteorological factors were almost similar for the two periods. In the first experimental period, duckweed ponds were covered with plastic sheet for 12 weeks (5 days/week). The reasons for covering were that algae blooms were noticed at the surface of the stabilization ponds that expected to be the reason for death of duckweed. Moreover, to avoid probable stress caused by direct exposure to solar radiation, (or a combination of the two factors) that was expected to cause death of duckweed.

Therefore, the role of the mentioned cover was to decrease direct radiation on the ponds surface and therefore to prevent algal growth and thermal stress. Moreover, sufficient aeration was maintained to the ponds surface for replenishment of oxygen for the duckweed and maintaining aerobic conditions at the surface of water column.

3.6 Design parameters of stabilization ponds

The stabilization ponds were operated under continuous flow, at ambient temperature conditions with temperature variation between 15 ^{o}C and 35 ^{o}C. The operated flow rates were calibrated to maintain equal total HRT of 28 days in each line. Applied flow rates for the three lines (90, 60 and 30 cm) were 59, 39 and 20 L/d, respectively A considerable difference was found in the volumetric loading rate for DBPs compared to ABPs, the organic loading rate for the three lines in each line was 25.0 and 11.8 g COD/m^3.d in DBPs and ABPs respectively. Table 3.1 shows the volumetric loading rates for each pond in the two experimental periods.

Table 3.1. Organic loading rates (g COD/m^3.d) for each pond during DBPs and ABPs periods.

Parameter	**P_1L_1**	**P_1L_2**	**P_1L_3**	**P_2L_1**	**P_2L_2**	**P_3L_2**	**P_3L_1**	**P_2L_3**	**P_3L_3**
OLR_{DBPs}	75.0	75.1	75.0	47.9	40.9	32.1	31.6	22.8	26.7
OLR_{ABPs}	35.5	35.5	35.4	21.2	24.7	23.0	19.9	22.3	21.4

3.7 Wastewater sampling

Wastewater grab samples of 300 ml were collected weekly from the influent (effluent of the UASB-septic tank) and the effluents of the nine ponds. The samples were collected for conducting all the chemical analyses each time in the laboratory. However, the other phisico-chemical analyses as temperature, pH, and dissolved oxygen were directly measured in situ in the ponds and the influent. On the other hand, the volume of the samples that were collected from the batch flow buckets was 50 ml for each sample. All of the tests were performed on the same day of sampling.

3.8 Wastewater sampling point

The test wastewater samples were collected from the top 5 cm of the wastewater surface. The samples were collected from the nearest point to effluent pipes (Figure 3.3). The influent samples were collected from the pipes connecting the holding tank with the ponds. The samples from the buckets were collected also at a depth of 5 cm beneath the surface level.

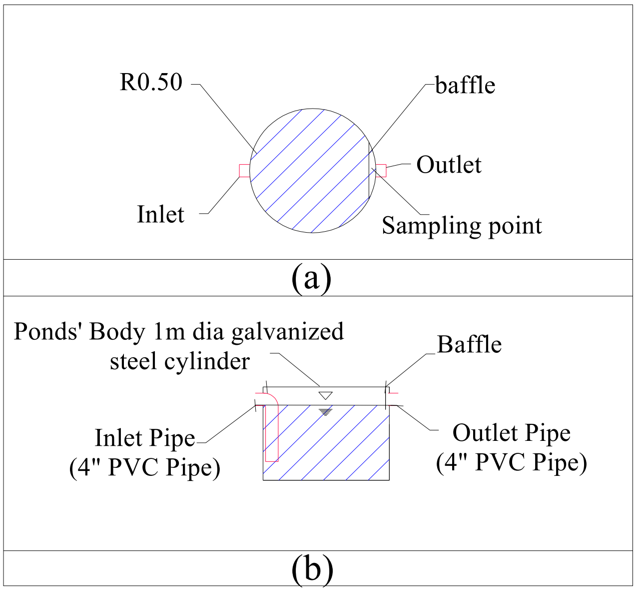


Fig. 3.3 Schematic diagram for a typical stabilization pond; (a) shows the top view and (b) shows a longitudinal cross section for the pond.

3.9 Wastewater analysis

3.9.1 Chemical analysis

3.9.1.1 Chemical oxygen demand (COD)

COD measurements were conducted every week during the research period, Measurements were carried out using the closed reflux (acid destruction at 150 ^{o}C for two hours) and then colorimetric at 600 nm wavelength method as described in standard methods (APHA, 1995).

3.9.1.2 Total Kjeldahl nitrogen (TKN)

TKN measurements were conducted weekly, the measurements were performed using Macro-Kjeldahl method, samples were (digested, distillation, and titration) according to APHA (1995).

3.9.1.3 Ammonium (NH_4^+)

NH_4^+ concentration was measured every week. $NH4^+$ was measured colorimetrically at 425 nm after nesslerization as described in APHA (1995). The experiments were conducted using spectrophotometer using 1 cm cell against distilled water.

3.9.1.4 Total Phosphorus (TP)

TP measurements were conducted every week throughout the research period. Measurements were carried out using the ascorbic acid spectrophotometric method, according to the standards methods (APHA, 1995) and measuring absorbance was conducted by spectrophotometer at wave-length of 880 nm.

3.9.2 Physical analysis

3.9.2.1 Total suspended solids (TSS)

TSS was measured according to the standards methods (APHA, 1995) by drying at 105 ^oC degree oven.

3.9.2.2 Total solids (TS)

TS was measured according to the standards method (APHA, 1995) by drying raw wastewater samples 105 ^oC oven.

3.9.2.3 pH

pH was determined for all the samples by pH meter (HACH).

3.9.2.4 Temperature

Temperature was determined in situ by alcohol thermometer for each grab sample in the location. Ambient temperature was measured also by alcohol thermometer.

3.9.2.5 Color

Color was determined by visual appearance.

3.9.2.6 Dissolved Oxygen(DO)

Dissolved oxygen was measured with a DO meter. The WTW microprocessor oximeter OX 196 with electrode E 096 was used to measure dissolved oxygen in the wastewater. The calibration of the electrode had been checked before each measurement.

3.9.3 Sediment collection

Cylindrical cups were used to collect sediment from the DBPs and the ABPs. The first group of cups was installed in DBPs in the 4^{th} week of the experiment (one cup per pond), and they remained until the end of the research period. The second group of cups was installed in the first week of the second experimental period until the end of the research (one cup per pond in parallel with the initially installed cup). The dimensions of the cups were, (diameter = depth = 8cm). Sediment depth was measured at the end of the two experimental periods after evacuating the supernatant water carefully using meter. The tests conducted for the sediment were according to the aforementioned methods after dilution of sample to 1:25 of the initial concentration.

3.10 Calculations

Removal efficiency (%)

The removal efficiency of different components of pollution can be calculated via equation 3.1

$$\% = \frac{(X_{\inf} - X_{eff})}{X_{\inf}} * 100\% \tag{3.1}$$

Where:

% = Removal efficiency;

X_{inf} = Concentration of component in the influent (mg/L);

X_{eff} = Concentration of component in the effluent (mg/L).

Hydraulic retention time (HRT)

HRT can be calculated by equation 3.2

$$HRT = \frac{C}{OLR} \tag{3.2}$$

Where:

HRT = Hydraulic retention time (d);

C = COD concentration in the influent (g COD/m^3);

OLR = Organic loading rate (gCOD/m^3.d)

Flow rate (Q)

Flow rate can be calculated via equation 3.3

$$Q = \frac{V}{t} \tag{3.3}$$

Where:

Q = flow rate (m^3/d);

V = volume of reactor (m^3);

t = retention tine (d).

Nitrogen mass balance

The terms of nitrogen mass balance represent the influent nitrogen and the nitrogen pathways and the nitrogen concentration in the effluent. The difference between the two fluxes represents the unaccounted nitrogen which is the losses and/or error in measurements in a complete mass balance. The following equation represents nitrogen mass balance.

$$N_{\inf} = N_{eff} + N_{sed} + N_{DW} + N_{AV} + N_{Den} + N_{unacc} \qquad (3.4)$$

where,

N_{Inf} and N_{Eff} = Influent and effluent nitrogen (TKN) concentrations (mg/L)

$N_{Sed.}$ = Nitrogen accumulation in the sediment (mg/L).

$N_{Dw.}$ = Nitrogen recovered (mg/L) via duckweed harvesting (for DBPs only).

$N_{AVol.}$ = Nitrogen leaving the system via ammonia volatilisation (mg/L).

$N_{Denit.}$ = Nitrogen leaving the system via denitrification (mg/L).

$N_{unAc\ for}$ = Unaccounted fraction of nitrogen (mg/l).

Ammonia and ammonia volatilization

The unionized ammonia fractions (α) in the pond water were calculated using equation 3.5, by Clement and Merlin (1995):

$$\alpha = \%\ Unionised\ NH_3 = \frac{100}{1 + 10^{(pK_a - pH)}} \qquad (3.5)$$

Unionized NH_3 is relatively volatile and can be removed from solution to the atmosphere via diffusion through water to the surface and through mass transfer from the water surface to the atmosphere. The ammonia volatilization rate was found to depend on pH, water temperature (Jørgensen, 1989; Stratton, 1968, 1969 as quoted by Zimmo, 2003) and mixing conditions (Pano and Middlebrooks, 1982). The mass transfer equation (equation 3.6) with the assumption that the concentration of the ammonia gas in the atmosphere is zero was used to estimate the average ammonia volatilization rates from each pond of the two systems:

$$N_{NH_3} = -K_l NH_3 \tag{3.6}$$

Where;

N_{NH_3} : The mass transfer rate of ammonia (mg/L.d);

K_l : The convection mass transfer coefficient in the liquid phase (d^{-1});

NH_3 : The concentration of ammonia in the liquid phase (mg /L).

This first order equation for ammonia mass transfer rate has been supported by Stratton (1969) who obtained the following expression for the mass transfer coefficient:

$$K_l = \frac{0.0566}{d} \exp[0.13(T - 20)] \tag{3.7}$$

Where,

d : The depth of water column in the pond (m);

T : The water temperature (^{0}C).

3.11 Data analysis

Statistical analyses for data were carried out using Microsoft Excel 2003 (Microsoft Corporation) software package. With this software most of data analyses (including arithmetic averages, standard deviations, removal equations and correlations between different variables) and graphs were carried out. Furthermore, since the system composed of three lines of ponds then the statistical comparison among removal efficiencies of these lines and the different ponds was carried out at a level of significance (ρ) of 0.05 using SPSS software for windows. Release 12.0, SPSS© Inc. (2003). A good comparison between the aforementioned lines in each period could be carried out since these lines were operated in parallel and fed with similar influent wastewater and HRT. Nevertheless, the variation in some parameters in the influent with time between the two periods was taken into consideration.

Chapter Four

Results and Discussion

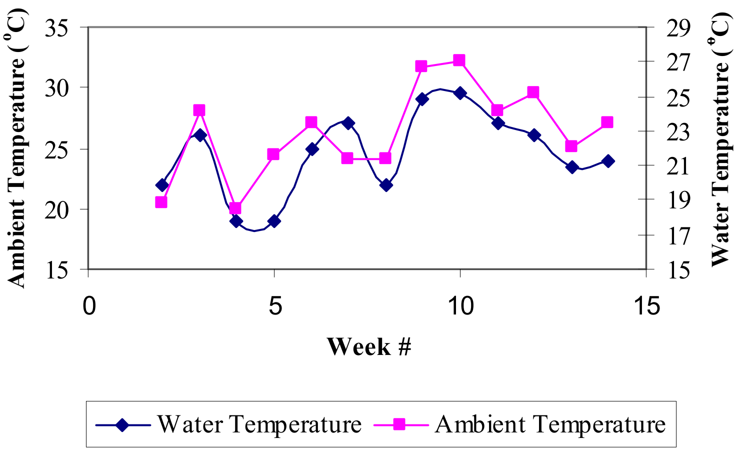
4.1 General

The effect of depth on nitrogen transformation was studied in a pilot-scale treatment plant. The research was divided into two main periods; in the first period, the effect of depth on the performance of DBPs (in nitrogen and other major pollutants removal) was investigated. In the second period, the effect of depth on the performance of ABPs in nitrogen and other major pollutant forms removal were tested. Duckweed death (as discussed later) was noticed most of the time for the first experimental period. This may justify the obtained data that sometimes disagree with the previous studies. The results related to the pretreatment unit (UASB-septic tank) were discussed else where (Al-Shayah, 2005). Moreover, the results for pathogenic related issues in our system were discussed in another research work (Samhan, 2005).

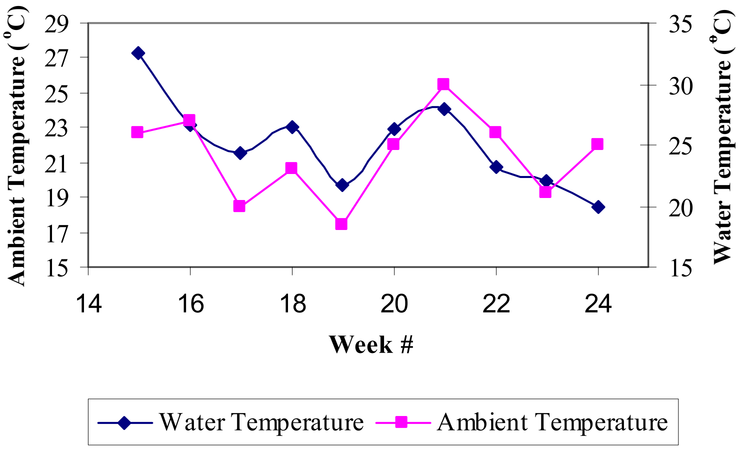
4.2. Physicochemical properties of the system

4.2.1 Temperature

The ambient temperature during the two experimental periods is known to be the highest throughout the year. The average ambient temperature in the first and second experimental periods were 24.5 ± 3.4 ^{o}C and 24.2 ± 3.5 ^{o}C, respectively. The average wastewater temperatures in the ponds of the first, second, and third lines were similar throughout the two experimental periods. The average temperature in the DBPs and ABPs were 22.9 ± 2.6 ^{o}C and 22.1 ± 2.5 ^{o}C, respectively. No significant difference was found in ponds temperature as a result of depth variation. Moreover, the effect of duckweed cover on water temperature was noticed most of the time (see Figure 4.1). However, the difference between ambient and water temperature did not exceed 5 ^{o}C. However, water temperature in ABPs, was most of the time higher than ambient temperature due to direct exposure of water column to solar radiation. The optimal temperature range for maximum duckweed growth was reported to be 17.5 to 30 ^{0}C with upper tolerance limit of 34 ^{0}C (Javed, 1995). This means for the current system that the temperature values are within the favorable range for duckweed growth conditions. Figure 4.1 shows the average ambient and wastewater temperatures throughout the two experimental periods. Temperature has also a substantial effect on the microorganism's performance in the ponds which plays a major role in determination the required HRT. In the light of literature, the reported experimental temperatures are expected to be optimal for the performance of the UASB-septic tank and pond systems (Mahmoud, 2002; Zimmo, 2003).



(a)



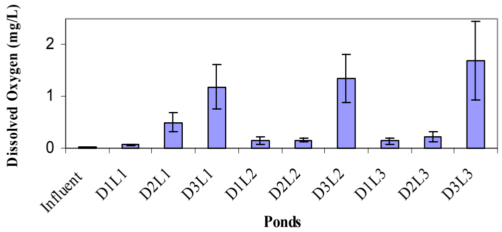
(b)

Fig. 4.1. Ambient and wastewater temperatures during the DBPs period (a) and ABPs period (b). The number of measurements for the DBPs and ABPs were 13 and 10, respectively.

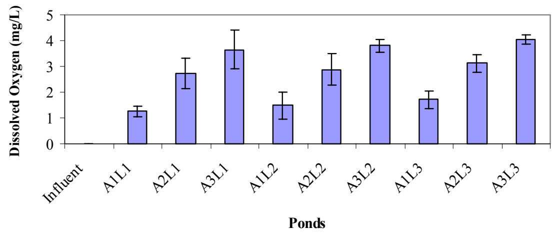
4.2.2 Dissolved oxygen (DO)

DO concentration in the effluent of UASB-septic tank was found to remain zero throughout the experimental period. Throughout the ponds, gradual increase in DO concentration was found at the effluent of the successive ponds in each line for the two periods ($\rho \leq 0.05$). However, no significant change was found in DO concentrations among the corresponding ponds (for example, A_1L_1, A_1L_2 and A_1L_3) in each line ($\rho \geq 0.05$). Average DO concentrations in the effluent of the three lines in the first period (90, 60 and 30 cm) were, 1.19 ± 0.43, 1.36 ± 0.47 and 1.69 ± 0.77 mg/L, respectively. Furthermore, DO concentration in the effluent of ABPs was significantly higher compared to DBPs ($\rho \leq 0.05$). Average DO concentrations in the effluent of the three lines in the second period (90, 60 and 30 cm) were 3.7 ± 0.70, 3.8 ± 0.30 and 4.0 ± 0.2 mg/L, respectively. Figure 4.2 shows the average value for DO concentration (vertical columns) at the effluent of the nine ponds in the two experimental periods. The bars represent the value of standard error of the mean ($\pm\sigma/n^{1/2}$).

The zero concentration of DO in the influent (effluent of the holding tank) indicates that the HRT of wastewater in this tank was not enough for oxygen replenishment from the anaerobic pretreated wastewater. Moreover, the significant increase in DO concentration for the successive ponds reflects the decrease in the oxygen consuming substances in the successive ponds with continuous flux of oxygen to the ponds wastewater. This result is in agreement with literature (Zimmo, 2003; Al-Jabari, 2003; Caicedo *et al.,* 2002; Steen, 1998). Furthermore, the insignificance in the values of DO concentrations in the effluent of corresponding ponds in different lines, suggests that natural oxygen replenishment was insufficient to create a significant change in DO concentration in these ponds. This could be justified by the high concentration of oxygen consuming substances in the treated wastewater. The high concentration of total COD was reported by Al-Shayah (2005) who reported that the total COD concentration in the effluent of the four-days HRT, UASB-septic tank ranged from 266 to 810mg/L with an average of 493 ± 95 mg/L.



(a)



(b)

Fig. 4.2. DO concentration in DBPs (a) and ABPs (b). The vertical columns represent the average DO concentration in the nine ponds effluent (mg/L); the bars represent the standard error ($\pm\sigma/n^{1/2}$). D stands for duckweed and A stands for algae. Note that (D or A)iLj means the i^{th} (D or A) pond in the j^{th} line (e.g. D_3L_1, means the third DBP in the first line).

The higher DO concentration in ABPs compared to DBPs suggests the production of DO through photosynthetic production of oxygen by phytoplankton in the water column. On the other hand, the lower DO values observed in DBPs compared to ABPs may be the result of reduced diffusion of oxygen from the air into the water column by the duckweed cover and of reduced photosynthetic production of oxygen resulted from lack of light crucial for photosynthesis process. Similar results were also reported in literature (Zimmo, 2003). He reported a DO concentration was over-saturation during warm seasons and 3.5-5.7 mg/l in ABPs and DBPs operated at the same HRT, respectively. The low oxygen concentration achieved in the pilot plant (for both periods) could be explained by the zero concentration of DO in the effluent of UASB-septic tank and the high organic loading rates.

4.2.3 pH

pH values were monitored throughout the two periods. pH values in the influent was found stable throughout the two experimental periods ($\rho \geq 0.05$), this was reflected from the equal average values of pH for the two experimental periods (7.3 ± 0.1). Similar value of pH was reported by (Al-Shayah, 2005) who mentioned that pH in the effluent of the UASB-septic tank, ranged between 7.12 and 7.70 with an average value of 7.4 ± 0.14. Moreover, a slight decrease in the influent pH (data not shown) at the end of the research period (7.2) when compared to the influent pH at the beginning of the research period (7.5). As shown in Figure 4.3, pH values were stable also in each pond. This is reflected by the small values of standard error for each pond. A significant difference in pH value was noticed among the influent and effluent of the first pond of each line during the two experimental periods. Furthermore, significant difference in pH values for the first experimental period, was found among corresponding ponds with different depths (for example, A_2L_1, A_2L_2 and A_2L_3), ($\rho \leq 0.05$), except for the values of pH in the first pond of each line ($\rho \geq 0.05$). However, no significant difference in pH values during the second experimental period, among corresponding ponds with different depths ($\rho \leq 0.05$) except for the values of pH in the third pond of each line ($\rho \geq 0.05$).

No significant difference was found in pH values throughout the successive DBPs of the three lines ($\rho \geq 0.05$). However, significant increase in pH in the aforementioned ponds was noticed between the influent and the first pond in each line ($\rho \leq 0.05$). The average values of pH at the effluent of the three lines (90, 60 and 30 cm) in the first experimental period were 7.40, 7.53 and 7.73, respectively. However, significant difference in pH was found among the successive ABPs ($\rho \leq 0.05$). pH values in the effluent of the three lines (90, 60 and 30 cm, respectively) in the second experimental period were, 8.0 ± 0.03, 8.1 ± 0.03 and 8.3 ±0.03. Furthermore, significant increase in pH values was noticed for ABPs when compared to DBPs ($\rho \leq 0.05$). Similar results were obtained in literature concerning the higher pH value in the ABPs when compared to DBPs. High pH values are not likely to occur in DBPs. The increase of pH in ABPs resulted from intense photosynthetic activity (CO_2 uptake). However, the maintained complete duckweed cover suppressed the algae that suppressed the elevating effect on pH (Wang, 1991; Caicedo., 2002; Zimmo, 2003; Al-Jabari, 2003).

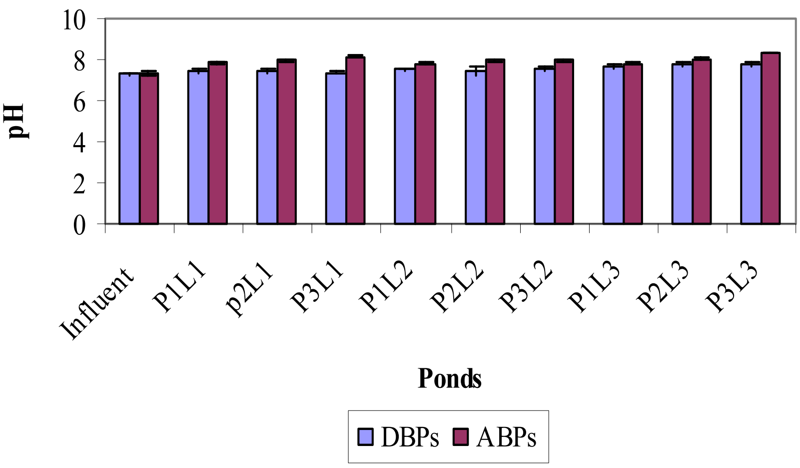


Fig. 4.3. Average pH values in DBPs and ABPs. The vertical columns represent the average pH values for the influent and the nine ponds of the pilot plant, the bars represent the standard error ($\pm\sigma/n^{1/2}$). n for first and second experimental periods were = 9 and 7, respectively.

Finally, lower pH values were measured (the values of pH did not exceed 8.3 in both system even in the ABPs) in these stabilization ponds compared to other previously reported values in preceding researches. Actually, this is could be justified by the anaerobically pretreated influent which is characterized by high concentration of CO_2 and the low concentrations of DO which resulted in more reduced environment compared to other stabilization ponds without anaerobic pretreatment. The same justification is also could be given to the lower variation in pH values between ABPs and DBPs compared to other studies.

4.3. Removal efficiencies of pollutant forms

4.3.1 Removal efficiency of solids

4.3.1.1 Removal efficiency of total suspended solids (TSS)

Total suspended solids (TSS) were measured 24 times; 14 times during the DBPs period and 10 times during the ABPs period. Grab samples were tested weekly from the influent (effluent of the UASB-septic tank), and the effluent of the nine ponds.

I. TSS removal efficiency in DBPs

Average TSS concentration in the influent was 172.1 ± 5.9 mg/L. However, Significant TSS removal efficiency between the successive ponds of the same line was observed. TSS concentrations at the effluent of the three DBPs lines (90, 60 and 30 cm) were 60.8 ± 18.4, 70.9 ± 17.9 and 66.1 ± 22.1 mg/L, respectively. Therefore, no significant difference was observed among TSS removal rate from the effluent of the three lines ($\rho \geq 0.05$). Furthermore, maximum removal efficiency was achieved in the deepest ponds (64.4 ± 11.8%), and the minimum removal efficiency was achieved in the second line of the intermediate ponds (58 ± 11.2%), whereas intermediate removal efficiency was achieved for the third line of the shallowest ponds (61.2 ± 13.8%). No significant difference was found in TSS removal efficiency among the three lines. In other words, no effect was observed for depth in TSS removal efficiency during the first period.

Figure 4.4 presents the TSS concentration at the effluent of each pond in the DBPs period. TSS removal efficiency of from the DBPs (depth ≥ 60 cm) was modeled by an equation reported by Smith and Moelyowati (1999). The latter considered initial TSS concentration at the influent, water temperature and HRT are the main parameters affecting TSS removal from DBPs. According Smith and Moelyowati model the TSS removal efficiency can be expressed as follows:

$$Se = Si\left(\left(\frac{1.18}{T}\right)\ln(t) + \frac{6.5}{T}\right) \tag{4-1}$$

Where,

Si: Initial concentration of TSS (mg/L).

Se: Final concentration of TSS (mg/L).

T: Water temperature (^{o}C).

t: Time (days).

Considering the average water temperature (24 ^{o}C) and HRT (28 days) in our system, the equation then reduces to

$$Se = 0.436\ Si \tag{4.2}$$

However, higher removal efficiency was achieved in our system (58 ± 11%) Se and (61 ± 14%) Se in the first and second lines, respectively. The result points out that no decomposition for the duckweed occurred in the system and/or the decomposition of duckweed (which was not observed) did not deteriorate the quiescent conditions of the water column in the DBPs.

The insignificant difference found in removal efficiency among the three DBPs lines could be attributed to the low difference in pond depth and long HRT. A difference could be achieved therefore in ponds with shorter HRT and larger difference in depth. No literature was found discussing the effect of depth variation on TSS removal efficiency in

DBPs. However, similar results were reported in literature concerning the achieved removal efficiencies of TSS by Bonomo *et al*. (1996) who achieved a removal efficiency of 50-80%. Comparable results were reported by Zimmo *et al.* (2001) who reported TSS removal efficiency of 71% in DBPs operated at HRT of 28 days with depth of 90 cm. However, higher TSS removal efficiencies were reported in other studies (Mandi, 1994; Bonomo *et al.*, 1996; Zirschky and Reed, 1988; Al-Jabari, 2003). The difference in removal efficiency could be attributed to difference in HRT and wastewater characteristics. Moreover, TSS removal by the adsorption (one of the mechanisms for TSS removal in DBPs systems) on the duckweed roots (Skillicorn *et al,* 1993) probably did not occur in our system.

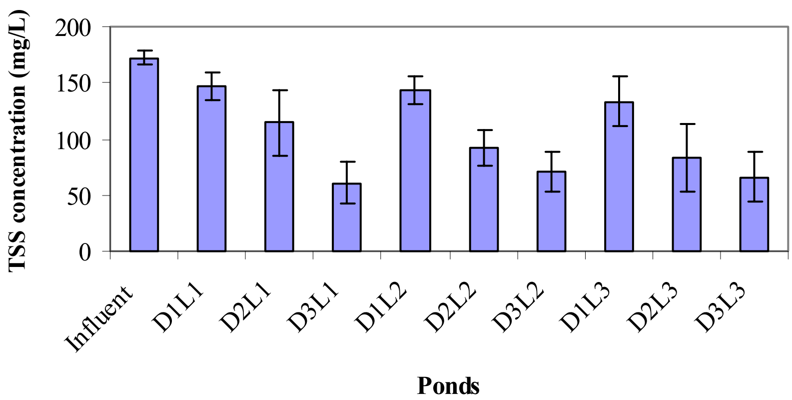


Fig. 4.4. The average values of TSS concentration in the influent and effluent of DBPs, n = 5. The vertical columns represent the average TSS concentration values for the influent and the nine ponds of the pilot plant, the bars represent the standard error ($\pm\sigma/n^{1/2}$) at a level of significance $\rho \leq 0.05$.

II. TSS removal efficiency in ABPs

Total suspended solids concentrations in ABPs period were monitored throughout the second experimental period. Lower concentration of TSS was found in the influent to ABPs throughout the second experimental period compared to the first experimental period. Average concentration for TSS in the influent was 94.7 ± 5.8 mg/L. However, negative removal efficiencies were achieved in most of ABPs ponds. The average TSS concentrations in the effluent of the three lines (90, 60 and 30 cm) were 133.8 ± 10.4, 88.5 ± 19.9 and 153.1 ± 15.8 mg/L, respectively. Therefore, no relation was found between TSS removal efficiency and depth. Figure 4.5 shows a graphical representation for TSS concentration in the influent and effluent of the nine ABPs.

Significant difference in TSS removal efficiency was found among the first two ponds and the third pond of the first line (90 cm). Significant difference in TSS removal efficiency was found also among the three ponds of the second and third lines (60 and 30cm, respectively). In order to have a good understanding for this result it is essential to differentiate between the influent TSS and the generated TSS in the system by the algal growth. An increase in the concentration of algae was noticed (visual test) for the second and third ponds of each line.

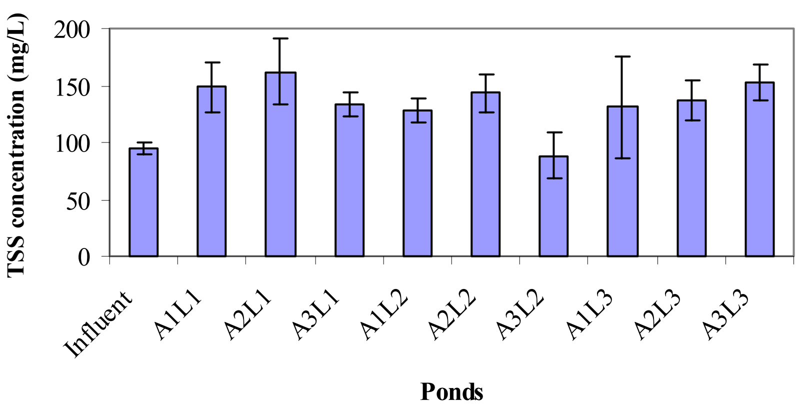


Fig. 4.5. The average values of TSS concentration in the influent and effluent of ABPs, n = 5. The vertical columns represent the average TSS concentration values for the influent and the nine ponds of the pilot plant. The bars represent the standard error at a level of significance $\rho \leq 0.05$.

The removal efficiency of TSS in DBPs ranged from 64.4 to 58.8%. Whereas, negative removal efficiency was found in ABPs, which was due to the intensive growth of algae in the ABPs system because of luxurious conditions for algal grow which is represented by availability of nutrients and light. Therefore, optimal growth was observed in the shallowest ponds where light is available throughout the water column. This justification can explain the highest TSS concentration found in the effluent of this line. According to literature, Zimmo *et al.* (2001, as cited by Al-Jabari, 2003) reported TSS removal efficiency of 36.9% in 90 cm-depth ABPs operated at a HRT of 28 days. However, Al Jabari (2003) reported a 71% TSS removal efficiency in 90 cm-depth ABPs with HRT of 32 days. Therefore, the large difference between the two reported values from literature could be justified by the difference in HRT, type of pretreatment, wastewater characteristics.

The better removal efficiency of TSS in DBPs compared to ABPs was reported in other studies (Zimmo, 2003; Al-Jabari, 2003; Mara and Pearson, 1998; Steen, 1998). This result is attributed to lower algal development and better sedimentation due to effect of shading (lack of light penetration) and quiescent conditions provided by duckweed cover. Reed *et al.* (1995) found that the effluents from ABPs systems are characterized by a high concentration of suspended solids, which in some occasions exceeded 100 mg/l. Similar results were found by Smith and Moelyowati (1999) and Skillicorn *et al.* (1993).

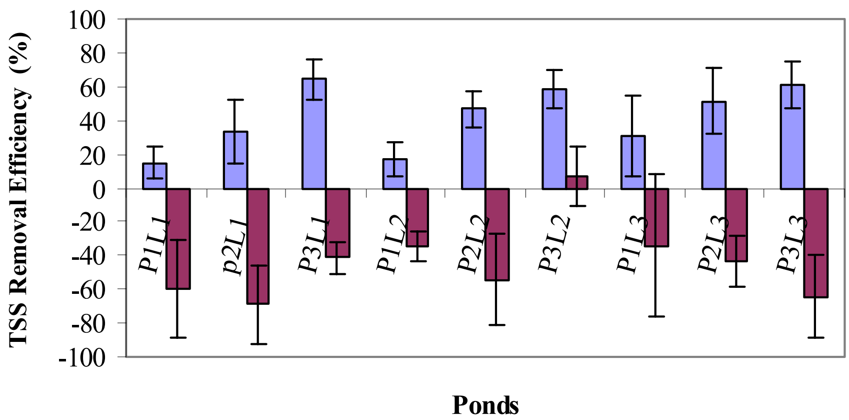


Fig.4.5. Average values for the removal efficiencies of TSS in the nine ponds of the two experimental periods. The vertical columns represent the average values TSS removal efficiency. The bars represent the standard error ($\pm\sigma/n^{1/2}$) at a level of significance $\rho \leq 0.05$.

4.3.1.2 Total dissolved solids (TDS) removal

I. TDS removal efficiency in DBPs

TDS concentration in the influent and effluent of DBPs was measured four times during the first experimental period. Influent average TDS concentration was 973 ± 43.1 mg/L. However, TDS concentrations in the effluent of the three DBPs line (90, 60 and 30 cm) were 818.5 ± 29.9, 828.5 ± 20.6 and 841.0 ± 58.8 mg/L, respectively (Table 4.1). Therefore, negligible removal efficiency (12-13%) for TDS was found in this period. Moreover, no effect for depth can be concluded on removal efficiency for TDS and no clear trend for an increase or decrease in the TDS concentration in the successive pond for any line.

Table 4.1. Average values of TDS concentration for the influent and effluent of the nine ponds for the DBPs experimental period (mg/L) and the value of standard error in the concentration for each pond, n = 4. The table shows also the removal efficiencies (%) of the nine ponds with the corresponding standard error values.

Parameter	Influent	D_1L_1	D_1L_2	D_1L_3	D_2L_1	D_2L_2	D_3L_2	D_3L_1	D_2L_3	D_3L_3
Average	973.0	912.5	826.0	794.5	942.5	852.0	828.5	818.5	850.5	841.0
St. Error	43.1	48.0	2.0	46.1	10.8	37.2	20.6	29.9	40.2	58.8
%	---	13.6	15.8	14.8	6.2	15.1	12.4	3.1	18.2	12.6
St. Error	---	2.2	3.4	1.7	0.8	3.6	0.1	3.2	8.4	0.3

II. TDS removal efficiency in ABPs

TDS in ABPs was also monitored throughout the second experimental period. TDS was measured every week by using the grab sampling method. Average influent TDS concentration was found slightly lower in this period compared to the first period. Average TDS concentration was 849.2 ± 8.8 mg/L and the average TDS concentration in the effluent of the three lines (90, 60 and 30 cm) were 773.6 ± 6.1, 780.6 ± 14.3 and 760.4 ± 7.4 mg/L, respectively (Table 4.2). Moreover, TDS concentrations in the effluent of the nine ponds were almost equal; there was no significant difference in removal efficiencies for this parameter among the successive ponds of APBs lines ($\rho \geq 0.05$). Likewise, no significant effect for depth was detected on removal efficiency of TDS ($\rho \geq 0.05$).

Table 4.2. Average values of TDS concentration for the influent and effluent of the nine ponds for the ABPs experimental period (mg/L), and the value of standard error in the concentration for each pond ($\rho \leq 0.05$), n = 9. The table shows also the removal efficiency (%) with the corresponding standard error values.

Parameter	Influent	A_1L_1	A_1L_2	A_1L_3	A_2L_1	A_2L_2	A_3L_2	A_3L_1	A_2L_3	A_3L_3
Average	849.2	800.8	796.8	783.8	769.4	778.2	780.6	773.6	764.8	760.4
St. Error	8.8	14.9	3.6	7.4	4.8	3.9	14.3	6.1	10.6	7.4
%	---	5.7	6.2	7.7	9.4	8.3	8.1	8.9	9.9	10.4
St. Error	---	1.7	0.7	1.6	1.3	1.1	1.1	1.0	1.0	1.5

4.3.1.3 Total solids (TS) removal efficiency in ABPs

TS concentration in the effluent of ABPs was measured weekly throughout the second experimental period. Average TS concentration in the influent for this period was 1143 ± 106 mg/L and the average values for TS concentration in the effluent of the three lines (90, 60 and 30 cm) were 1096 ± 25, 1068 ± 42 and 1160 ± 31 mg/L, respectively. No relation was detected between removal of TS and depth variation. Moreover, no effect was also detected between HRT and removal efficiency of TS. Furthermore, small and/or negative removal efficiencies were observed in most of the ponds in this period. Figure 4.6 shows the removal efficiencies of TS in the ABPs.

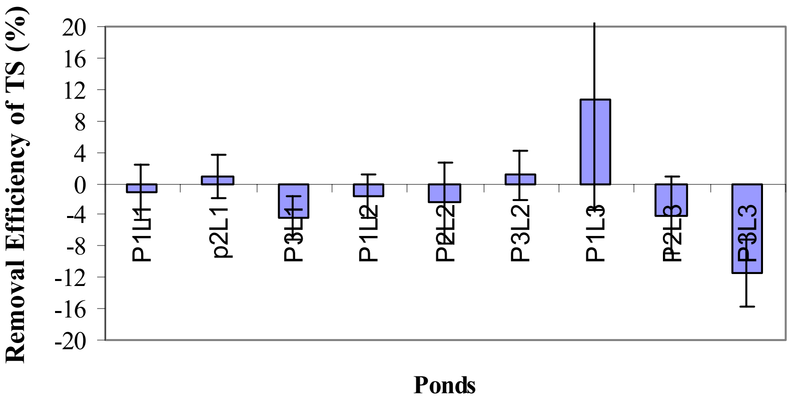


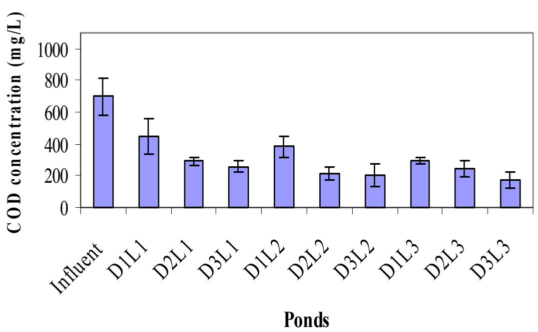
Fig. 4.6. TS removal efficiency from the nine ABPs, n = 9. The columns represent the removal efficiency and the bars represent standard error ($\pm\sigma/n^{1/2}$).

The results obtained for TDS and TSS in ABPs were reflected in the total solids readings. It is clear that the removal efficiency for TS in the second experimental period was very low for the first two lines and for the first pond of the third (shallowest) line. This also gives an indication for the concentration of algae in the ponds; the negative removal efficiency in the last two ponds of the shallowest line indicates a high algal concentration in the effluent of these ponds. This result was obtained before when TSS was discussed. Moreover, the results obtained for the TDS, TSS, and TS removal efficiencies indicate that there is no relation between the depth of the pond and the removal efficiency of solids. However, it was shown that in ABPs the generation of TSS was due to the algal growth in the ponds where favorable conditions (light essential for photosynthesis and nutrients) were present. The availability of light increase as the depth of the pond decrease, so it was mentioned that the shallowest ponds has the highest algal concentration and lowest removal efficiency of TSS, and consequently the TS. Finally, according to the aforementioned results, it is highly recommended to polish ABPs effluent by a DBPs unit for significantly reduce TSS concentration in the effluent by removal of algae and provide quiescent conditions suitable for proper sedimentation (Steen, 1998).

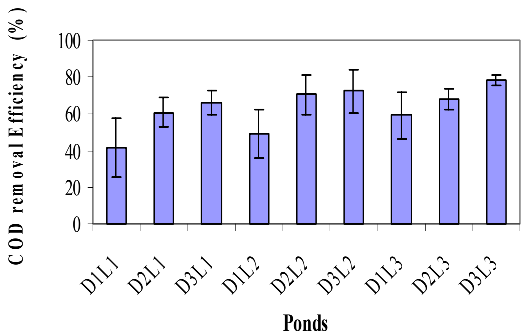
4.3.2 COD removal efficiency in DBPs and ABPs

4.3.2.1. COD removal in DBPs

Grab sampling was used to measure COD concentration in the influent and effluent of the ponds of different lines. COD was measured weekly throughout this period. Average COD concentration in the influent for this period was 701.6 ± 241.5 mg/L. However, effluent COD concentration from the three lines (90, 60 and 30 cm) were 258.6 ± 39.3, 205.0 ± 72.7 and 174.8 ± 49.2 mg/L, respectively (Figure 4.7). Likewise, the removal efficiencies in the aforementioned lines were 62.5 ± 5.7% (least removal efficiency), 70.6 ± 11.6 and 75.4 ± 4.1% (maximum removal efficiency), respectively. Nevertheless, average COD removal rates (Figure 4.6) were observed significantly higher in the line of deepest ponds compared to line of shallowest ponds; the average COD removal rates in the first, second and third ponds of the first line (90 cm depth) were 24, 14.5 and 34 $g/m^2.d$, respectively. However, the average COD removal rates in the aforementioned ponds in the third line (30 cm depth) were 17, 13, and 66.5 $g/m^2.d$, respectively.



(a)



(b)

Fig. 4.7. Average values of COD concentration at the influent and effluent of DBPs (a) and COD removal efficiency from the nine DBPs (b). The vertical columns represent the average COD concentration values for the influent and the nine ponds of the pilot plant, the bars represent the standard error.

As shown in Figure 4.7, COD concentration shows a significant decrease at the effluent of the first pond ($\rho \leq 0.05$), this could resulted from the settleable and suspended COD particulates that enter the system and settle down in the first pond of each line. This issue was mentioned in (Al-Shayah, 2005). The latter mentioned that intermittent washout events of sludge were observed during the study period with a concentration of 2-3g/L. Moreover, he reported an average TSS concentration of 117 ± 19mg/L in the effluent of the UASB-septic tank. Therefore, the difference in TSS could be presented as suspended COD. One of the main reasons that explain the high value of solids that enter the system is that the holding tank which is connected directly to the ponds and receives all the UASB-septic tanks' effluent. This flow rate (200 L/d) was more than the pumped quantity of water to the ponds (118 L/d), so most of solids from the UASB effluent settle down and concentrated in the influent to the ponds. This also justify the difference observed in the measured COD values reported for the UASB-septic tank effluent and the reported values in this report. Moreover, the dissolved COD represented 61% of total COD load (Al-Shayah, 2005). This could also play a role in enhancement COD removal in the first pond because HRT required to consume dissolved COD is expected to be lower compared to other COD forms.

Removal efficiency of COD from the system was inversely proportioned to the depth. However, the difference in removal efficiency was not significant for the first and second ponds of each line ($\rho \geq 0.05$). In other words, significant difference in removal efficiency with respect to depth was observed only for the third pond in each line (HRT 28 days). On the other hand, COD removal rate was significantly higher in the deepest ponds (direct proportionality between COD removal rate and depth). This result was achieved because the flow rate (and therefore rate of COD enters the system) of the deepest line is 1.5 and 3 time the flow rate for the second and third lines, respectively. However, COD removal rate over rate of COD enters the deepest ponds was not the highest compared to other shallower ponds. However, the considerable difference in removal rate for the deepest line in addition to the highest capacity (highest flow rate), makes it most attractive option among the three lines for large-scale application in terms of land requirement (discussed later). No comparable studies were found in literature discussing the effect of depth on COD removal. Nevertheless, this result can be justified by the fact that other natural activities that also took place in the control reactors are dominant such as sedimentation, biodegradation by microbes, etc.

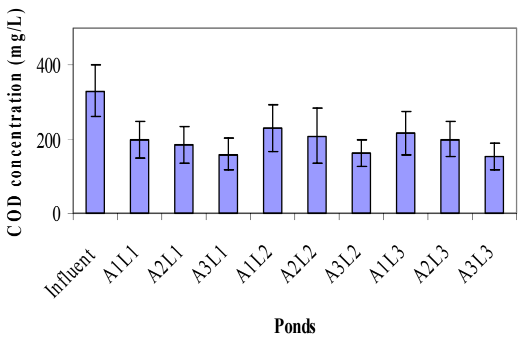
Considering the achieved removal efficiencies from the three lines, similar results reported by Steen (1998). He reported 55 ± 26% COD removal efficiency in a combined DBPS and ABPs of 0.29 m depth fed with anaerobically pretreated domestic sewage. Caicedo (2002) reported 82% of COD was removed in DBPs (0.7m depth and HRT = 21days) fed with anaerobically pretreated wastewater. However, the influent sewage was institutional wastewater. Moreover, similar results were also recently reported by Awadallah (in preparation) who reported a COD removal of 67 ± 34.0% in Al-Arroub DBPs of 1.8 m depth. Similar result was achieved also by Javed (1995) who reported that no significant difference was noticed in the concentration of COD in batch reactors at different depths (HRT = 20 days). He concluded that the removal efficiency of COD in the 10 cm depth reactor was 63% whereas in the 95 cm reactor was 56%. Moreover, Al-Jabari (2003) reported also a similar removal efficiency of COD (80%) in DBPs (depth = 0.9 m) fed with anaerobically pretreated wastewater. Nevertheless, wastewater used in the latter's system was collected from latrines and cesspits which means that there is a large difference in influent wastewater characteristics which was pretreated twice before post-treated by DBPs. However, Mirzapur (Bangladesh) duckweed based lagoons (DBL) system showed extremely high COD removal of 95-97% as reported by (Zimmo, 2003). The large difference could be justified by different environmental conditions, wastewater characteristics and higher temperature.

4.3.2.2 COD removal in ABPs

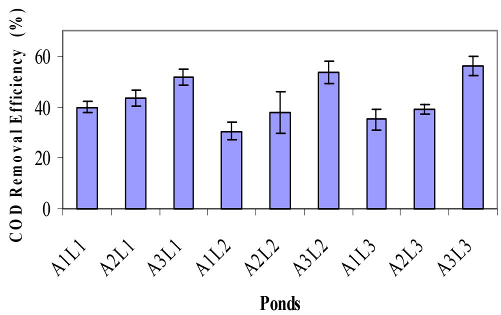
The same method of sampling (grab sampling) was used to measure COD of the influent and effluent of the ponds of different lines. Ten weeks were committed to test COD removal efficiency of ABPs. Steady state conditions (stable readings) were achieved after five weeks from the beginning of this experimental period. Lower COD concentration was noticed for the effluent of the UASB-septic tank in this period (Figure 4.8) compared to the first period. Average COD concentration in the influent for this period (fifth to ninth week of this period) was 330.9 ± 69 mg/L. This indicates that the UASB-septic tank efficiency was stable throughout this period. Al-Shayah (2005), reported that in the aforementioned period, no shock loads were received by the system and better removal efficiency was achieved. Moreover, COD concentrations (± S.E.) in the effluent of the three lines (90, 60 and 30 cm) were 158.8 ± 43.5, 162.6 ± 35.9 and 152.1 ± 55.9 mg/L, respectively.

COD removal efficiencies in ABPs were inversely proportioned to depth (Figure 4.8). The maximum removal efficiency was achieved in the shallowest ponds (55.9 ± 3.7%), whereas the least removal efficiency was achieved in the deepest ponds (51.6 ± 3.2%)

and the removal efficiency was intermediate in the effluent of 60 cm – depth ponds (53.4 ± 4.3%). The average COD removal rate in the second period was significantly higher in the deepest ponds (direct proportional relationship was observed between depth and removal rate). Figure 4.9 shows the average COD removal rates (± S.E.) in the nine ABPs. Average COD removal rates in the three ponds (first line) were 54% and 189% higher than that in the second and third lines, respectively. This could actually be attributed to the highest flow rate (and therefore rate of COD enters the system) of the deepest ponds compared to other shallower pond because total HRT for the three lines was maintained equal.



(a)



(b)

Fig. 4.8. Average values of COD concentration at the influent and effluent of ABPs (a) and COD removal efficiency from the nine ABPs (b). The vertical columns represent the average values for COD concentration in the influent and the nine ponds of the pilot plant, the bars represent the standard error.

The relatively fast achievement of steady state conditions that was observed in our system is supported with literature. The same observation was reported previously by Mara and Pearson (1998) who stated that three to four weeks are sufficient in summer time for WSPs to reach steady state conditions because of higher rate of microbial population development. Moreover, the ponds operated before as DBPs, this also contributed to acceleration of the steady state conditions achievement because part of the microbial populations was already developed. According the statistical analysis, significant decrease in removal efficiency was found among the three successive ponds in the first and third ABPs lines ($\rho \leq 0.05$). However, no significant difference in COD removal efficiency was found among the three successive ponds of the second line ($\rho \geq 0.05$). The difference in removal efficiency among the three lines was sufficiently low (difference between highest and lowest removal efficiencies was 2.4%) which means that the use of deep ponds for COD removal is the most economically feasible compared to other lines of lower depths. This result is also supported by the significantly higher removal rate for COD observed in the aforementioned deepest ponds compared to other ponds.

4.3.2.3 Comparison and discussion

Similar climatic and operational conditions were observed and maintained during the ABPs period and DBPs. Therefore, proper comparison can be carried out between the three lines in the two experimental periods. The influent COD concentration in the second experimental period was lower (COD concentration influent for the first experimental period was 701.6 ± 241.5 mg/L, whereas, it was 330.9 ± 69 mg/L in the second experimental period). Consequently, COD concentrations in the successive ponds were also lower than that in the corresponding ponds of the first experimental period. However, COD removal efficiency in ABPs period was found to be lower than the removal efficiency achieved in DBPs. Nevertheless, effluent COD concentration in the three lines were in compliance with discharge standards of 150 – 200 mg/L established by Ministry of Environmental Affairs on treated wastewater. However, this concentration was occasionally achieved in the second and third lines (60 and 90 cm depth, respectively) of DBPs due to higher concentration in the influent.

Lower COD concentration in the effluent of ABPs compared to DBPs. Nevertheless, the removal efficiency of COD in DBPs was higher than that in ABPs. This could be attributed to higher COD concentration in the influent for the first experimental period and lower algal development and better sedimentation due to effect of shading and quiescent conditions provided by duckweed cover. Similar results were reported by other studies (Zimmo, 2003). Moreover, algae concentration (which is added to the COD load) in the ponds increased significantly (visual test) for the second and third ponds with respect to the first pond of each line, the concentration increased as the depth of the pond decrease. The reason behind this result is that higher density of light could penetration to the water column for the shallower ponds.

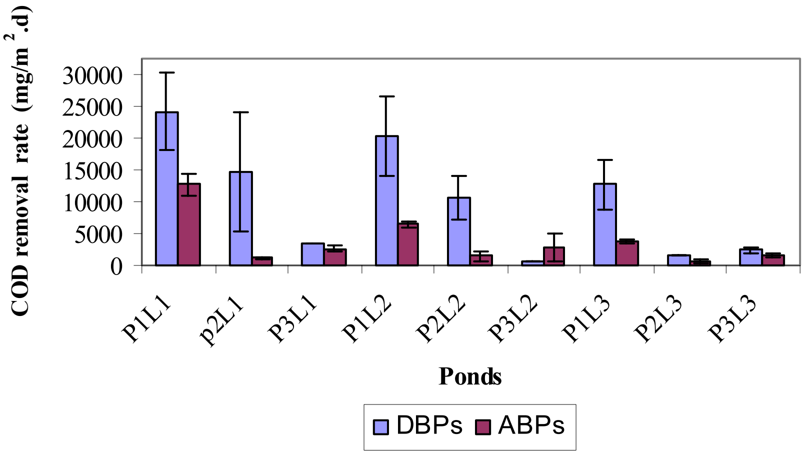


Fig. 4.9. Average values for COD removal rates (mg/m^2.d) for the nine ponds in the two periods. The vertical columns represent the average values COD removal rates. The bars represent the standard error ($\pm\sigma/n^{1/2}$).

No significant difference in COD removal efficiency was found observed between the first two ponds of the three lines in the first period. Similar result was reported by Javed (1995) in 90 cm reactor. Furthermore, the removal efficiencies achieved in DBPs were almost equal to the reported removal efficiencies (81%) in DBPs (70 cm depth and 21 days HRT) by Caicedo (2002). Likewise, similar removal efficiency (75%) in DBPs (2.3 m) was reported by Bonomo (1997). However, higher removal efficiency was reported by Al-Jabari (2003) who achieved 80% and 70% removal efficiency in DBPs and ABPs, respectively. DO concentration which plays an important role in COD removal, was significantly lower in the first period compared to the second period. However, COD removal efficiencies were significantly higher in DBPs. This result could be attributed to lower algal development and better sedimentation due to effect of shading and quiescent conditions provided by duckweed cover. Similar results were reported by other comparable researchers (Smith and Moelyowati, 1999; Caicedo, 2002; Zimmo, 2003; Körner *et al.* 1998 (as quoted by Zimmo, 2003); Al-Jabari, 2003). Finally, deepest ponds in both systems proved to be the most feasible in terms of COD removal rate (in the reported conditions), therefore the use of such ponds is recommended for large scale application. Nevertheless, increasing the number of ponds (or HRT) can achieve higher removal efficiency with maintaining land requirement for these ponds lower compared to other shallower ponds. However, further research is required to reinforce this result especially in the winter time when the efficiency of the UASB and ponds system is expected to be lower (Zimmo, 2003; Iqbal, 1999; Steen *et al.,* 1999; Lettinga *et al.,* 1993; El-Metwalli, 1999).

4.3.3 Total phosphorus (TP) removal

TP is the sum of the organic and *ortho*-phosphorus. In our system, the UASB-septic tank was inefficient in removal of nutrients, the average influent TP concentration to the UASB-septic tank was 14.0mg/L. However, "TP accumulation" and negative removal efficiency were observed in the effluent; the average TP concentration (± standard deviation) at the effluent of UASB-septic tank throughout the two experimental periods was 14.2 ± 1.1 mg/L (Al-Shayah, 2005). Consequently, most of the removed quantity from the whole system is achieved in the pond system. TP removal from the pond system was monitored throughout the two experimental periods. TP concentration at the influent and effluent of each pond was measured every week.

4.3.3.1 TP removal in DBPs

Stable values of TP were found in the effluent of the UASB-septic tank. This can be observed by the small values of standard error. The average value of TP concentration at the influent was 15.3 ± 0.5 mg/L. Significant difference was found between TP concentration in the influent and effluent of the first pond in each line ($\rho \leq 0.05$). TP concentrations at the effluent of the three lines (starting with the deepest to the shallowest) were 9.4 ± 0.4, 9.0 ± 0.2, and 7.8 ± 1.2 mg/L, respectively. This result indicates that there is an inverse proportionality relation between the ponds depth and the removal of TP in the effluent. Significant difference in TP removal efficiency was found between the successive ponds in all DBPs lines ($\rho \leq 0.05$). Furthermore, according to the statistical analyses, no significant difference was found in TP removal efficiency between the effluent of second and third lines in this period. The removal efficiencies of TP for the three mentioned lines (starting with the deepest to the shallowest) were 38.5 ± 4%, 40.8 ± 3% and 48.5 ± 9.2%, respectively. The average TP removal rates in the nine ponds are shown in Table 4.3. Moreover, average TP concentrations at the effluent of each pond in the first experimental period are shown in Figure 4.10.

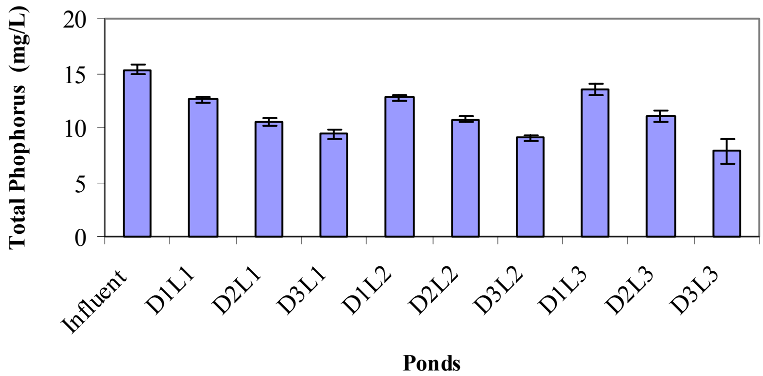


Fig. 4.10. The average value of TP removal efficiency in DBPs and the standard error bars ($\pm\sigma/n^{1/2}$) at a confidence level of 95%.

The highest removal efficiency achieved in the shallowest ponds could be attributed to the higher surface/volume ratio in addition to the lower surface loading rate (lower flow rate) and lower water velocity (settling velocity) which plays a major role in the process of sedimentation. However, the insignificance in TP removal efficiency found between the second and third lines could be attributed to the long HRT and low difference in depth between the mentioned lines. Deepest ponds achieved the highest values for TP removal rates (Table 4.3) compared to other lines. Moreover, TP concentration at the effluent of the second pond of this line (HRT =18.6 days) was lower than 15mg/L (TP concentration according to Jordanian guidelines for restricted irrigation). Accordingly, these ponds are more feasible for large-scale application compared to the other ponds.

Moreover, TP concentration in the effluent of the first pond (HRT = 9.3) in line one (15.3 mg/L) complied with the mentioned guidelines. This concentration is expected to decrease with normal duckweed status, because in duckweed systems besides the higher reduction of TP in DBPs is normally attributed to duckweed uptake and subsequent removal by harvesting. However, this mechanism of TP removal was eliminated in our system. The achieved removal efficiencies were in line with other reported results in literature. Javed (1995) reported a removal efficiency of 88% for the reactors of 10 cm depth, and 56% for the reactors of 95 cm in batch flow reactors. Nevertheless, it is inconvenient to compare two systems with different flow configurations. However, TP removal efficiencies in both systems showed inverse proportionality with depth. Recently, Zimmo (2003) reported P-removal efficiency in one meter depth ponds was 67-68%. However, the duckweed status was normal in the latter’s system. Moreover, the achieved results were significantly lower than the “extremely” high removal efficiencies (94% and 96%) reported by Al-Jabari (2003) at HRT equals 16 and 32 days, respectively.

4.3.3.2 TP removal in ABPs

TP was measured weekly throughout the second experimental period. The average value of TP concentration at the influent was 12.8 ± 0.7 mg/L. Significant difference in TP concentration was found between the influent and the effluent of the first pond in each line ($\rho \leq 0.05$). TP concentrations at the effluent of the three mentioned lines (starting with the deepest to the shallowest) were 8.0 ± 0.8, 6.8 ± 1.1, and 5.4 ± 0.6 mg/L, respectively. Consequently, as the case in DBPs, the concentrations of TP in the effluent shows that there is an inverse proportionality relation between the ponds depth and the removal of TP in the effluent of the ponds. No significant difference was found in the removal efficiency of the second and third lines ($\rho \geq 0.05$). The removal efficiencies of TP for the three mentioned lines (90, 60 and 30 cm) were 37.6 ± 6.4%, 46.4 ± 10.5%, and 57.6 ± 5.6%, respectively. Figure 4.11 shows the average TP concentration in the effluent of the nine ponds in the second experimental period. The columns represent the average value of the concentration for the influent and effluent of each pond, the error bars represents the standard error. Moreover, average TP removal rates in the nine ponds for this period are depicted in Table 4.3.

TP concentration in the influent to the pond system was inline with the reported values of TP by (Al-Shayah, 2005). He reported that TP range in the effluent in the whole experimental period was 12.5-16.8 mg/L with an average (± standard deviation) of 14.2 ± 1.1 mg/L. Similar justification (as the case in DBPs period) could be given to the insignificance in TP removal efficiency between the second and third lines (high surface area/depth ratio and long HRT). Moreover, as the case in the first experimental period, TP removal rate was highest in deepest ponds compared to other ponds. Furthermore, TP concentration at the effluent in the first pond in the deepest line (HRT = 9.3) complied with Jordanian guidelines for restricted irrigation. Therefore, these ponds are the best alternative for application in large scale.

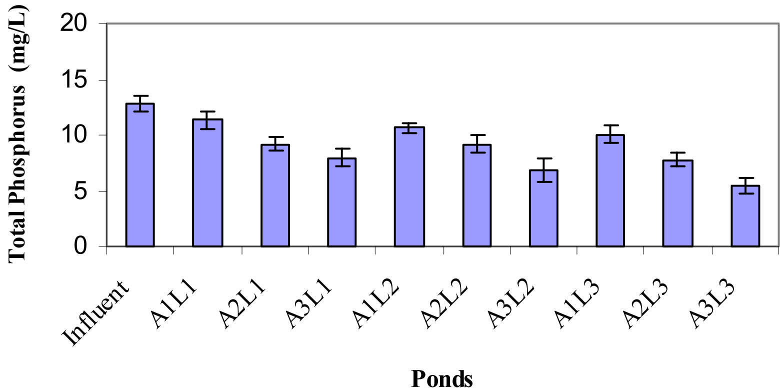


Fig. 4.11. The average values of TP concentration (mg/L) in the influent and effluent of the nine ABPs of the pilot plant of ABPs (represented by columns). The bars represent the values of standard error ($\pm\sigma/n^{1/2}$).

TP removal in ABPs is generally attributed to sedimentation of incorporated phosphorus in particulate with decayed algae and due to biological phosphorus removal. According to preceding researches, TP removal efficiency was higher in DBPs compared to ABPs. However, the result differs in our system; higher removal efficiencies were found in ABPs system compared to DBPs system. This result could be attributed to the die off for duckweed. Therefore, the amount of TP removed by duckweed remained in the water column. Nevertheless, both systems achieved Jordanian guidelines for restricted irrigation. Moreover, deepest ponds in both systems were capable to achieve the aforementioned guidelines after the first pond in line one (HRT = 9.3 days). Therefore, the use of deepest ponds in the two systems is more feasible in comparison with other shallower ponds.

Table 4.3. Average values of TP removal rate in the nine ponds for the two experimental periods ($mg/m^2.d$), and the values of standard error ($\pm\sigma/n^{1/2}$) at each pond.

Period		P_1L_1	p_1L_2	p_1L_3	p_2L_1	p_2L_2	p_2l_3	p_3L_1	p_3L_2	p_3L_3
DBPs	**Average**	258	164	57	199	127	78	109	108	103
	S. E.	36	36	7	34	30	26	33	12	36
ABPs	**Average**	140	139	88	207	93	71	115	151	76
	S. E.	39	19	13	58	52	15	79	67	30

4.3.3.3 Comparison and discussion

No comparable studies were found in literatures that discuss the effect of depth on TP removal. However, the effect of depth on TP removal efficiency was clear in both systems; maximum removal efficiency was found for both systems in the shallowest ponds (48.5 ± 9.2% for DBPs, and 57.6 ± 5.6% for ABPs). However, the least removal efficiency was found in the deepest ponds. Moreover, significant decrease in removal rate was found among the three lines in both experimental periods ($\rho \leq 0.05$) as the depth of the pond decrease (Figure 4.12). All the mechanisms of TP removal were similar in both systems because the duckweed has no ability to contribute in TP removal through direct uptake, accordingly, the lower removal efficiency of TP in DBPs could be justified by this fact. Additionally, negligible phosphorus amount could be released back to the water column as the dead mat of duckweed was removed from the system completely.

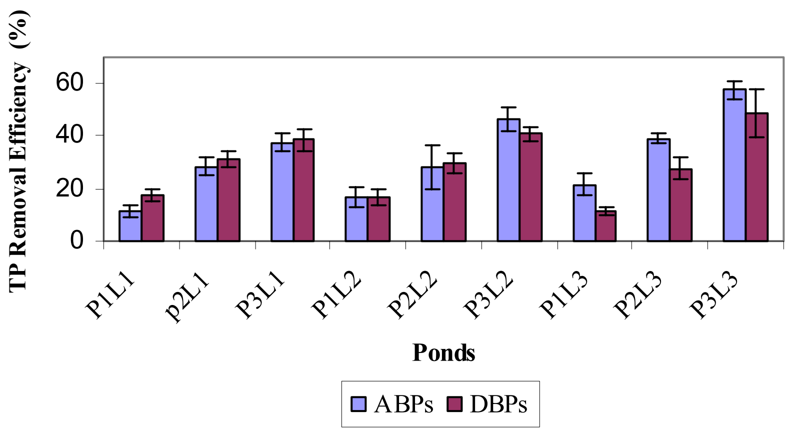


Fig. 4.12. Average values for the removal efficiencies (%) of TP in the nine ponds of the two periods. The vertical columns represent the average values TP removal. The bars represent the standard error ($\pm\sigma/n^{1/2}$) at a level of significance $\rho \leq 0.05$.

It is believed that DBPs with normal status for duckweed are able achieve higher removal efficiency than ABPs systems. This issue was reported in other studies in literature; several researchers reported that the DBPs are more efficient in TP removal (Culley *et al*., 1978; Reddy and DeBusk 1985; Zirschky and Reed, 1988; Al-Nozaily *et al*., 2000; Zimmo, 2003). On the other hand, Javed (1995) reported that the existence of duckweed did not play an important role in phosphorus removal. Nevertheless, this system was batch flow system, and therefore it is inconvenient to consider this result for our continuous flow plant. Similar result was also achieved by Al-Jabari (2003) who reported similar extremely high removal efficiency (94-97%) at 32 days HRT and 90 cm depth for both systems. The difference in these results could be attributed to the fact that the type of wastewater used in that research was different (wastewater from latrines or cesspits) rather than municipal wastewater that was tested in our system.

4.3.4 Ammonium (NH_4^+) removal

NH_4^+ was monitored weekly throughout the two experimental periods. UASB-septic tank was not efficient in NH_4^+ removal. The influent concentration of NH_4^+ was 58.9 mg/L and the effluent NH_4^+ concentration was in the range 51-67 mg/L with an average of 59 ± 4.4 mg/L (Al-Shayah, 2005). Therefore, the removal of NH_4^+ in the system was highly dependent on pond system.

4.3.4.1 NH_4^+ removal efficiency in DBPs

Average NH_4^+ concentration in the influent to DBPs was 60.6 ± 1.9 mg/L. significant decrease in this concentration was observed between the influent and effluent of the first pond in each treatment line ($\rho \leq 0.05$). The average NH_4^+ concentration (± S.E.) at the effluent of the three lines (90, 60 and 30 cm) were 41.8 ± 0.9, 37.6 ± 1.7 and 32.3 ± 2.4 mg/L, respectively. Moreover, according to the statistical analysis, significant decrease in removal efficiency was observed among the three successive ponds in each line ($\rho \leq 0.05$). Likewise, significant decrease in removal efficiency was observed among the effluent of the three mentioned lines as the depth increases. Removal efficiencies achieved in the three lines (90, 60 and 30 cm) were, 30.9 ± 1.6%, 37.9 ± 1.7% and 46.6 ± 5.2%, respectively. This suggests that there is an inverse proportionality between NH_4^+ removal and depth. However, as depicted in Table 4.4, maximum NH_4^+ removal rate was achieved in the deepest pond which is attributed to highest flow rate applied to this line (for maintaining equal overall HRT in the three lines). Figure 4.13 shows average NH_4^+ concentration at the effluent of the nine DBPs.

The reported value for average NH_4^+ concentration in the influent was inline with the reported value at the effluent of the UASB-septic tank by (Al-Shayah, 2005). Limited information is available on the effect of depth on removal of ammonium in ABPs and even less information is available for DBPs. However, the inverse proportionality between ammonium removal efficiency and depth can be observed in previous studies where higher removal efficiencies were reported in shallow ponds compared to deep ponds. Javed (1995) reported NH_4^+ removal efficiency of 94% for a 10 cm depth batch flow reactor of 20 days HRT reactor. Moreover, Oron *et al.* (1987) reported NH_4^+

removal in the range of 40% and 70% in semi-continuous flow duckweed ponds (depth = 20-30 cm and) at HRT = 3 and 10 days, respectively. He used raw sewage with initial NH_4^+ concentrations of 47.5 ± 16 mg/l. However, Awadallah (in preparation) reported removal efficiency of 61 % in DBPs fed with effluent of septic tank (depth = 180 cm, HRT = 3 days). Nevertheless, direct comparison of results with the above studies is not possible due to differences in HRT, water depths, initial nitrogen concentrations and duckweed condition, densities and harvesting regimes.

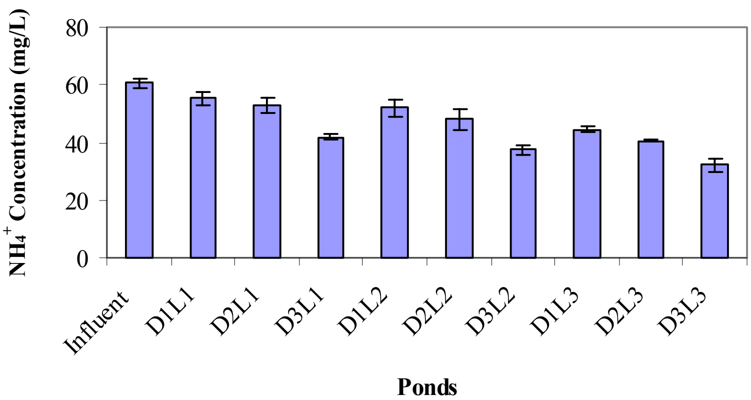


Fig. 4.13, Average values of NH_4^+ concentration in the influent and effluent of the nine ponds of the pilot plant of DBPs (represented by the columns). The bars represent the standard error ($\pm\sigma/n^{1/2}$).

Concerning the values of removal efficiencies in our system, higher removal efficiencies were reported in literature in comparable systems. For example, Steen (1998) reported ammonium removal efficiency 50-80%. Similar results were reported in some researches (Buddhavarapu and Hancock, 1991; Kawabata *et al.,* 1986). Most of these studies attributed the higher NH_4^+ to the ability of DBPs to remove nitrogen through uptake of nitrogen by duckweed species (Iqbal, 1999; Steen, 1998). Therefore, the larger surface and mat/volume favored NH_4^+ consumption from the water column. However, this is not the fact in our system because the effect of duckweed was eliminated. Nevertheless, the higher surface/volume ratio could favor ammonia volatilization process (Körner and Vermaat, 1998; Steen *et al.*, 1998) and/or denitrification that were not affected by the existence of "maintained" duckweed cover (Zimmo, 2003).

4.3.4.2 NH_4^+ removal in ABPs

Average NH_4^+ concentration in the influent was 62.7 ± 5.7 mg/L. Significant decrease in concentration was achieved at the effluent of the first pond in each line ($\rho \leq 0.05$). The average NH_4^+ concentration (± S.E.) at the effluent of the three ABPs lines (90, 60 and 30 cm) were 30.7 ± 3.6, 27.2 ± 3.8 and 22.2 ± 2.1 mg/L, respectively. Moreover, significant decrease was found in removal efficiency among the three successive ponds in each line ($\rho \leq 0.05$). Likewise, significant decrease in removal efficiency was found among the three ABPs lines ($\rho \leq 0.05$). The removal efficiencies found in the aforementioned lines were 51.2 ± 1.9%, 56.9 ± 2.9% and 64.5 ± 2.8%, respectively. Accordingly, (as the case in DBPs period), these results suggests also an inverse proportionality between NH_4^+ removal efficiency and depth. However, highest NH_4^+ removal rates were achieved in the deepest ponds that are attributed to highest flow rate (and therefore highest NH_4^+ rate that enters this line) applied to this line (Table 4.4).

Table 4.4. Average values for NH_4^+ removal rate in the nine ponds for the two experimental periods (mg/m^2.d) and the values of standard error ($\pm\sigma/n^{1/2}$) at each pond.

Period		P_1L_1	p_1L_2	p_1L_3	p_2L_1	p_2L_2	p_2l_3	p_3L_1	p_3L_2	p_3L_3
DBPs	**Avg.**	511	550	508	235	253	129	1044	659	265
	S. E.	101	120	35	145	112	46	175	227	71
ABPs	**Avg.**	1142	422	661	1165	1046	337	750	797	294
	S. E.	154	45	119	256	133	76	198	55	58

The reported influent concentration of ammonium was in agreement with the reported value for the UASB-septic tank effluent (Al-Shayah, 2005). Moreover, the higher removal efficiency achieved in the second period (ABPs) was also reported in preceding-comparable studies (Zimmo, 2003; Al-Jabari, 2003). Furthermore, the higher removal efficiency achieved in the shallowest ponds could be attributed also (as the case in DBPs) to the higher surface/volume ratios in these ponds compared to other ponds that favored denitrification and/or ammonia volatilization process (as discussed later). Moreover, the raised pH shifted the equilibrium between NH_4^+ and NH_3 towards the latter. This justification could be given for the higher removal efficiency achieved in this period. Figure 4.14 shows NH_4^+ concentration in the nine ABPs.

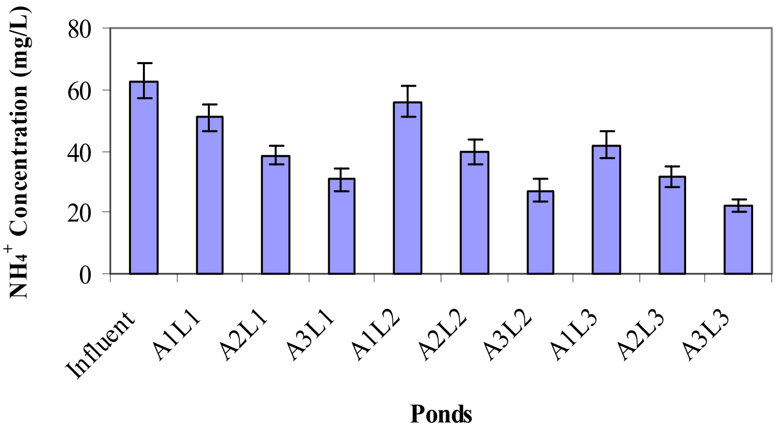


Fig. 4.14. The average values of NH_4^+ concentration in the influent and effluent of the nine ABPs (mg/L). The bars represent the standard error ($\pm\sigma/n^{1/2}$) at a level of significance $\rho \leq 0.05$.

4.3.4.3 Comparison between the two periods in NH_4^+ removal

According to the previous discussion, the removal efficiency of the two systems had an inverse proportionality to the depth of the pond; maximum removal was obtained in the shallowest ponds, while minimum removal was obtained in the deepest ponds. ABPs achieved higher removal efficiency of ammonium for each pond compared to the removal efficiency achieved in the ponds of duckweed system (Figure 4.15). However, the highest NH_4^+ removal rates were observed in the deepest ponds in both systems. This could be attributed to the highest flow rate applied to these ponds (in order to maintain equal total HRT for the three lines) therefore, highest rate of NH_4^+ enters these ponds. However, NH_4^+ removal rate over rate of NH_4^+ enters the system (removal efficiency) was not the highest compared to other shallower ponds. Therefore, these ponds were unable to achieve highest removal efficiency compared to other shallower ponds. However, an optimization can be made between depth and removal efficiency to achieve lowest land requirement.

Nitrogen compounds are removed from DBPs and ABPs by sedimentation of SS with organic nitrogen, ammonia volatilization, nitrification and denitrification. However, sedimentation is not expected to play a major role in NH_4^+ removal because it is in the ionic form (Steen, 1998). However, ammonia volatilization and/or denitrification in ABPs and DBPs are expected to be equal, as the duckweed cover did not provide a physical barrier for volatilization (Zimmo, 2003). Moreover, nitrification could take place in the last pond in each line and in all ponds (where DO concentration ≥ 0.5 mg/L) of the first and second periods, respectively (Taylor and Bishop, 1989), this could favour NH_4^+ removal through volatilization in these ponds. Furthermore, pH and temperature in both periods were optimal for occurrence of nitrification process (Metcalf and Eddy, 1991). However, Nitrifiers are known to prefer attachment to solid surfaces (Focht and Verstraete, 1977; Underhill and Prosser, 1987; Verhagen and Land brook, 1991). But in the first period the repeated removal of duckweed mat could prevent nitrifiers to thrive on the root zone of duckweed (Alaerts *et al*., 1996; Reed, 1988; Zimmo, 2003).

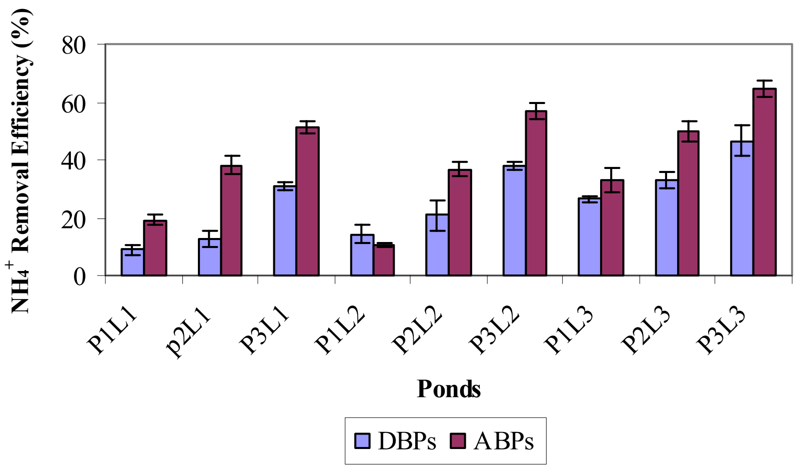


Fig. 4.15. Average values for the removal efficiencies (%) of NH_4^+ in the nine ponds of the two experimental periods. The vertical columns represent the average values NH_4^+ removal. The bars represent the standard error ($\pm\sigma/n^{1/2}$).

Likewise, the smooth ponds surface is not expected to be favourable for attachment of nitrifiers. However, high volume to surface/mat ratio enhances nitrification process (Zimmo, 2003). Nevertheless, the opposite was noticed when the shallowest ponds (lowest volume to surface ratio) achieved the highest removal efficiencies compared to other ponds. This suggests that nitrification did not play a major role in removal of NH_4^+ in both periods. However, heterotrophic denitrification could take place in the anaerobic zones of the ponds. Nevertheless, a complete nitrogen mass balance is highly recommended to approve/ disapprove these predictions.

4.3.5 Total Kjeldahl nitrogen (TKN) removal

TKN was measured weekly throughout the two experimental periods. Average TKN concentration in the influent of UASB-septic tank throughout the experimental period (two periods) was 78 mg/L and the average TKN concentration in effluent of UASB-septic tank was 68 ±6.7 mg/L, the range of its' value throughout the experimental period was 55-78 mg/L (Al-Shayah, 2005). Consequently, UASB-septic tank was not efficient in removal of TKN in our system. Therefore, most of TKN removal was achieved in the pond system throughout the two periods.

4.3.5.1 TKN removal in DBPs

TKN was measured in the first experimental period. The average value for the influent TKN concentration was 84.1 ± 5.7 mg/L. significant decrease in concentration was observed in the effluent of the first pond in each line compared to the influent to these ponds ($\rho \leq 0.05$). Average TKN concentration in the effluent of the three lines (90, 60 and 30 cm) were 59.1 ± 6.1, 53.7 ± 3.3 and 44.8 ± 4.6 mg/L, respectively. Figure 4.16 shows the average values of TKN concentration at the influent and effluent of the nine DBPs. Moreover, significant decrease in removal efficiency was observed among the successive ponds in each line. Likewise, significant decrease was observed in the removal efficiency among the three treatment lines for this period as the depth increase ($\rho \leq 0.05$). The removal efficiencies of the three lines (90, 60 and 30 cm) were 29.4 ± 6.8%, 35.7 ± 5.2% and 44.5 ± 6.3%, respectively. Nevertheless, highest TKN removal rates were achieved in the deepest ponds compared to other ponds (Table 4.5).

TKN concentration in the influent to pond system was slightly higher than the concentration reported in the effluent of the UASB-septic tank. This could be attributed to probable accumulation for organic nitrogenous particles that settled in the holding tank. Removal efficiency did not exceed 50% in the best cases. Furthermore, According to Figure 4.16, TKN removal efficiency had an inverse proportionality to the depth ponds. This result indicates that the main mechanism for organic nitrogen removal is sedimentation. Moreover, in the light of literature, sedimentation is enhanced by increasing surface/volume ratio (Körner and Vermaat, 1998; Steen *et al*., 1998). The latter explains the higher removal efficiencies of TKN in shallower ponds. Additionally, this result is highly supported in literature; similar results were reported by Javed (1995). The latter achieved TKN removal efficiencies of 86%, 33% and 17% in 10 cm and 95 cm plug flow reactors, but the 17% efficiency was achieved in fully mixed reactor (no sedimentation for organic nitrogen).

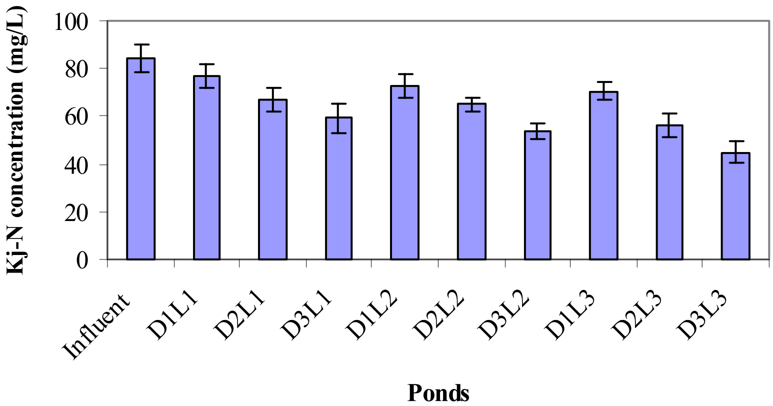


Fig. 4.16. Average values of TKN concentration in the influent and effluent of the nine DBPs. The bars represent the standard error ($\pm\sigma/n^{1/2}$).

Ammonium in this system represented 63-72% of TKN. Therefore, in addition to sedimentation, ammonia volatilization and denitrification (as discussed later) could be played a major role in removal. Moreover, heterotrophic nitrification and denitrification take place in anoxic zones (Zimmo, 2003), where organic nitrogen is decomposed by anoxic bacteria into ammonium and *ortho*-phosphate (Metcalf and Eddy, 1991). Therefore, the produced ammonium from organic nitrogen decomposition could take place by the aforementioned pathways.

4.3.5.2 TKN removal in ABPs

The average value of TKN concentration in the influent during the second experimental period was 88.8 ± 1.8 mg/L. significant decrease in TKN concentrations was observed between the influent and effluent of the first pond in each line ($\rho \leq 0.05$). Likewise, significant decrease was also noticed among TKN concentration in the effluent of the successive ponds in each line ($\rho \leq 0.05$). Average TKN concentrations in the effluent of the three lines (90, 60 and 30 cm) were 47.2 ± 1.7, 43.8 ±2.7 and 33.7 ± 4.0 mg/L, respectively. Therefore, maximum removal efficiency was achieved in the shallowest ponds. Figure 4.17 shows average TKN concentrations in the influent and effluent of the nine ABPs. The removal efficiencies achieved in the three lines (90, 60 and 30 cm) were 45.4 ± 3.1%, 49.3 ± 4.3% and 61.1 ± 4.5%, respectively. However, no significant difference in removal efficiency was observed between line one (90 cm depth) and line two (60 cm depth). Nevertheless, maximum TKN removal rate was achieved in the deepest ponds compared to other shallower ponds ($\rho \leq 0.05$). The values of average removal rates are shown in Table 4.5.

Table 4.5. Average values of TKN removal rates in the nine ponds for the two experimental periods and the values of standard error ($\pm\sigma/n^{1/2}$) for each pond. All values are measured in ($mg/m^2.d$).

Period		P_1L_1	p_2L_1	p_3L_1	p_1L_2	p_2L_2	p_3L_2	p_1L_3	p_2l_3	p_3L_3
DBPs	**Avg.**	511	235	1044	550	253	659	508	129	265
	S. E.	101	145	175	120	112	227	35	46	71
ABPs	**Avg.**	1355	1799	821	977	1045	848	945	430	380
	S. E.	339	391	254	151	325	157	120	121	36

TKN concentration in the influent to the pond system for this period is slightly higher than the reported value for the latter concentration in the effluent of the UASB-septic tank (Al-Shayah, 2005). This could be attributed (as the case in DBPs period) to probable accumulation in the holding tank. An inverse proportionality was observed between depth and removal efficiency of TKN. Similar justification (as previously discussed in DBPs) can be given also to this result; surface to volume ratio in the shallowest ponds was the highest among the other groups of ponds. Consequently, sedimentation and volatilization were enhanced in these ponds compared to other ponds. Moreover, the increase in average pH in the nine ponds in this period (as discussed before) also favored the processes of sedimentation, denitrification and ammonia volatilization (Zimmo, 2003), this could justify for the higher removal efficiency of TKN achieved in this period. Similar results were reported by (Zimmo, 2003; Al-Jabari, 2003; Caicedo, 2002; Reed, 1988; Steen, 1998; Alaerts *et al*., 1996).

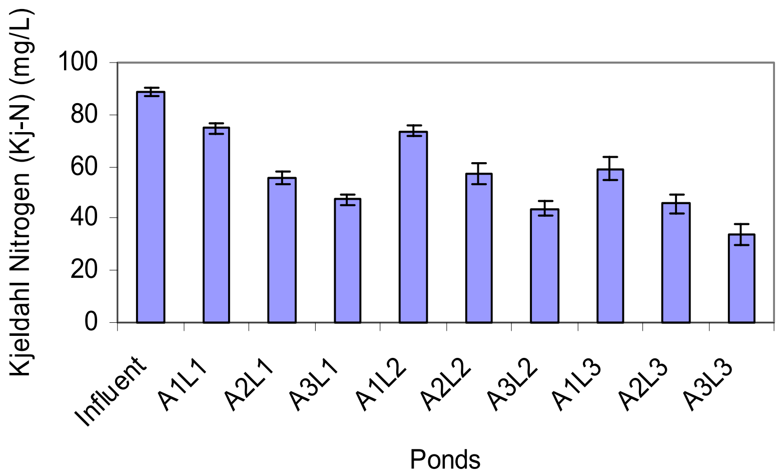


Fig. 4.17. Average values of TKN concentration in the influent and effluent of the nine ABPs. The error bars represent the standard error ($\pm\sigma/n^{1/2}$) at a level of significance $\rho \leq 0.05$.

4.3.5.3 Comparison between the two periods in TKN removal

TKN concentration in the influent for the first experimental period was slightly lower. However, TKN removal efficiency and removal rate in the first experimental period was lower compared to the second experimental period. This could be attributed to the higher values of pH detected in the ponds during the second experimental period compared to that in the first experimental period. Higher removal efficiencies for ABPs compared to DBPs were reported in literature despite the fact that algae remains in the effluent and then organic nitrogen remains in the effluent. Zimmo (2003) reported Annual nitrogen removal efficiencies in ABPs and DBPs were respectively 73% and 54% (depth = 0.9 m and HRT = 28 days). Several studies concluded that shallower ponds were more promising in nitrogen removal (Steen, 1998; Javed, 1995; Awadallah, in preparation).

Most of aforementioned studies attributed this result to the higher quantity of sediment occurred in the shallow ponds, ability of higher uptake by duckweed and higher rates of denitrification and ammonia volatilization. Literature shows that the amount of sediment, nitrification and denitrification and ammonia volatilization are dependent on pH value (Ferrara and Avci, 1982). Sedimentation in the form of particulate organic nitrogen (probably decaying algae biomass), was considered as the largest nitrogen flux in ABPs (Zimmo, 2003). additionally, the difference between some removal efficiencies mentioned in previous researches and the results achieved in our system is the large difference in the environment conditions i.e. aerobic or facultative, climate, HRT, and influent wastewater characteristics.

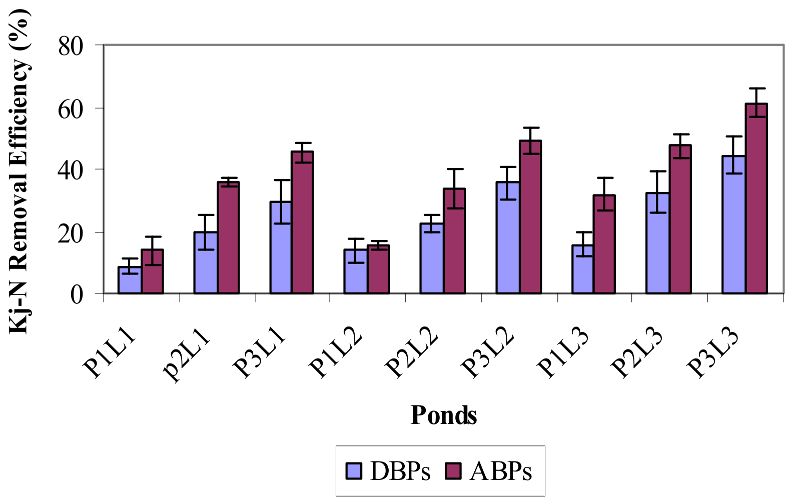


Fig. 4.18. Average values for the removal efficiencies (%) of TKN in the nine ponds of the two periods. The vertical bars represent the average values TKN removal. The bars represent the standard error ($\pm\sigma/n^{1/2}$) at a confidence level.

4.4 Sedimentation

A difference in the colour and shape of sediment of DBPs and that of the ABPs was observed. The colour of DBPs sediment (first group) was brownish while the colour of ABPS sediment (second group) was olive-green which indicates the presence of decayed algae. Table 4.6 shows the main results obtained from sediment testing. The tests that were conducted in this test were TN, COD ands TP. According to Table 4.6, higher quantities of sediment were found in ABPs compared to DBPs. This result was reported also by Zimmo (2003) who reported Moreover, considerable amounts of pollutants were removed via this pathway in our system. N-removal by sedimentation in both algae and duckweed systems increased with increase in pH, and was higher in algae systems than in duckweed systems (Zimmo, 2003).

Table 4.6. Average values of quantities and contents of sediment that was collected from the DBPs and the ABPs. COD, TP and TKN sedimentation rates are measured in (mg/m^2.d).

	Parameter	P_1L_1	p_2L_1	p_3L_1	p_1L_2	p_2L_2	p_3L_2	p_1L_3	p_2l_3	p_3L_3
DBPs	**Sed. Depth (cm)**	1.2	0.9	0.7	0.6	0.5	0.2	0.3	0.2	0.1
	COD (mg/L)	20583	18958	14925	20500	19025	15850	19925	15567	13783
	TP (mg/L)	413	543	303	289	636	349	114	391	274
	TKN (mg/L)	798	476	952	728	2184	700	714	392	322
	Volume (L)	9.42	7.07	5.50	4.71	3.92	1.57	2.36	1.57	0.79
	COD sed. rate†	3529	2438	1493	1757	1359	453	854	445	197
	TP sed. rate†	71	70	30	25	46	10	5	11	4
	TKN sed. rate†	137	61	95	62	156	20	31	11	5
ABPs	**Sed. Depth (cm)**	1.4	0.8	0.8	1.3	1.0	0.7	1.0	0.7	0.4
	COD (mg/L)	18167	15142	12742	18800	15550	11675	15817	14967	13350
	TP (mg/L)	1421	1275	491	423	889	386	612	467	423
	TKN (mg/L)	798	476	952	728	2184	700	714	392	322
	Volume (L)	11	6	6	10	8	5	8	5	3
	COD sed. rate†	3633	1730	1456	3491	2221	1168	2260	1497	763
	TP sed. rate†	284	146	56	79	127	39	87	47	24
	TKN sed. rate†	160	54	109	135	312	70	102	39	18

†: Sed. Depth = sediment depth. Sed. rate = sedimentation rate.

The results obtained in our system are in line with the results reported in literature for most of the parameters. According to literature, the sedimentation in ABPs compared to DBPs is significantly higher (Zimmo, 2003). However, different conclusions were drawn concerning the importance of this pathway for nitrogen removal. For example, Avci (1982) found that sedimentation was the main removal pathway for nitrogen. Similar conclusion was stated by Zimmo (2003) who reported (based on complete nitrogen mass balance) that sedimentation in ABPs and DBPs were 33-40% and 30-33%, respectively. However, the lower water velocity (settling velocity) in his ponds (due to higher surface area 3*1 m) could be the reason for better results for nitrogen removal through sedimentation. Furthermore, Steen (1998) who studied the performance of DBPs (fed by anaerobically pre-treated wastewater) in nitrogen removal reported a total removal of nitrogen by sedimentation of only 8%.However, settling velocity in the latter's plant was higher compared to the pilot plant reported by Zimmo (2003) Table 4.7 shows the percentage of sediment contribution in the pollutants removal for the two experimental periods.

Table 4.7 Percentage of TKN, TP, and COD removed via sedimentation in the three treatment lines in the two experimental periods. The values of pollutants concentration express the average value of concentration of pollutant at the influent throughout the time of each experimental period.

	Line #	Parameter	Concentration in the influent (mg/L)	Contribution of sedimentation in removal (%)
DBPs	First line	TKN	84	4.0
		TP	15	4.5
		COD	801	1.7
	Second line	TKN	84	13.2
		TP	15	8.4
		COD	801	4.2
	Third line	TKN	84	10.8
		TP	15	7.0
		COD	801	5.9
ABPs	First line	TKN	89	4.2
		TP	16	9.1
		COD	331	5.6
	Second line	TKN	89	35.1
		TP	16	24.0
		COD	331	31.1
	Third line	TKN	89	23.8
		TP	16	32.6
		COD	331	42.9

It is clear from Tables 4.6 and 4.7 that sediment depth and therefore the removal of the mentioned pollutants via sedimentation were significantly higher in the second experimental period (4.2-35.1 %) compared to the first experimental period (4.0-13.2 %). Moreover, Nitrogen removal via sedimentation was not dependent on depth. This could be attributed to the low difference in depth among the three lines of ponds. However, increasing the number of cups used to collect sediment and increasing the difference in depth between ponds could achieve better results.

4.5 Nitrogen mass balance

The mass balance equation (equation 3.4) was used to calculate nitrogen budget in the three lines for the two experimental periods. The results for nitrogen mass balance are depicted in Table 4.8. According to the nitrogen mass balance equation, nitrogen removal via ammonia volatilization and denitrification were not measured for our system in the two experimental periods. Therefore, the values for nitrogen removed via these pathways were lumped (AV +D).

Table 4.8. Nitrogen budget for the DBPs and ABPs systems. The listed values are in (mg/L).

DBPs										
Term	**Infl.**	**P_1L_1**	**p_2L_1**	**p_3L_1**	**p_1L_2**	**p_2L_2**	**p_3L_2**	**p_1L_3**	**p_21_3**	**p_3L_3**
TN	84.1	76.8	67.3	59.1	72.5	65.1	53.7	70.6	56.3	44.8
N removed	0	7.3	9.5	8.1	11.6	7.4	11.3	13.4	14.3	11.5
Sedimentation	0	1.6	0.7	1.1	1	2.4	0.3	1	0.4	0.1
AV+D	0	5.7	8.8	7	10.6	5	11	12.4	13.9	11.4
ABPs										
TN	88.8	74.7	55.8	47.2	73.5	57.1	43.8	59.2	45.7	33.7
N removed	0	14.2	18.8	8.6	15.3	16.4	13.3	29.7	13.5	11.9
Sedimentation	0	1.6	0.7	1.1	1	2.4	0.3	1	0.4	0.1
AV+D	0	12.6	18.1	7.5	14.3	14	13	28.7	13.1	11.8

As shown in Table 4.8, most of nitrogen removed could not be accounted for. Therefore, other unmeasured nitrogen removal pathways were the major nitrogen pathways in our system. In the light of literature, contradictory conclusions were found concerning the prediction for the main mechanisms and pathways responsible for nitrogen pathways. Some studies concluded that ammonia volatilization was the main nitrogen removal mechanism (Steen, 1998; Caicedo, 2003). Other studies concluded that denitrification was the main mechanism for nitrogen removal (Zimmo, 2003; Awadallah, in preparation). A prediction for ammonia volatilization and denitrification (as discussed later) was made on the light of previous studies.

4.6 Prediction for the unaccounted nitrogen removal mechanisms

4.6.1 Ammonia Volatilization

Ammonia volatilization values in the ponds were not measured in this research. However, it was mentioned that the proportions of ammonium ion (NH_4^+) and free ammonia (NH_3) are pH and temperature dependent (Erickson, 1985) which were measured throughout the two experimental periods. Therefore, equation 3.5 by Clement and Merlin (1995) was used to calculate the unionized fraction of NH_4^+. Moreover, the mass transfer equation (equations 3.6 and 3.7) with the assumption that the concentration of the ammonia gas in the atmosphere is zero was used to estimate the average ammonia volatilization from each pond of the two experimental periods. Table 4.9 shows the calculated values for NH_3 and ammonia volatilization.

According to the results presented in Table 4.9, ammonia volatilization played a negligible role in nitrogen removal in our system. From a chemical point of view, both forms of ammonia (ionic and gaseous) are available at equal concentration at pH of 9.4 and temperature of 20 ^{o}C. Therefore, ammonia volatilization is of considerable amount at high pH values only (Ferrara and Avci, 1982), which was not available in our system. Accordingly, the low pH in the ponds throughout the two experimental periods could be the reason for the low contribution of ammonia volatilization in nitrogen removal. The effect of depth and HRT were considered in the model used for calculations of ammonia volatilization, the model suggests that ammonia volatilization inversely proportioned to depth. This was reflected by the increase in nitrogen volatilization as the depth decrease.

Table 4.9. results for calculations for the fraction of unionized ammonia (according to equation 3.5) and Ammonia volatilization amounts in the system (according to equations 3.6 and 3.7).

		D_1L_1	D_2L_1	D_3L_1	D_1L_2	D_2L_2	D_3L_2	D_1L_3	D_2L_3	D_3L_3
DBPs	**% of ionized ammonia[1]**	1.5	1.6	1.3	1.8	1.6	2.1	2.8	3.1	3.3
	Ammonia (mg/L)	0.8	0.8	0.5	1.0	0.8	0.8	1.2	1.3	1.0
	AV (mg/L.d)[2]	0.08	0.08	0.05	0.14	0.11	0.12	0.35	0.36	0.29
	AV (mg/L)[3]	0.75	0.78	0.50	1.33	1.07	1.09	3.27	3.34	2.71
ABPs		A_1L_1	A_2L_1	A_3L_1	A_1L_2	A_2L_2	A_3L_2	A_1L_3	A_2L_3	A_3L_3
	% of ionized ammonia[1]	3.9	5.2	7.7	3.7	5.0	5.1	3.6	6.0	11.2
	Ammonia (mg/L)	2.0	2.0	2.4	2.1	2.0	1.4	1.5	1.9	2.5
	AV (mg/L.d)[2]	0.16	0.16	0.21	0.26	0.20	0.17	0.36	0.45	0.77
	AV (mg/L)[3]	1.51	1.52	1.94	2.44	1.86	1.57	3.41	4.17	7.14

1: Calculations based on Clement and Merlin (1995) model.

2, 3: Calculations were based on Stratton (1969) model, AV stands for ammonia volatilization.

Several studies considered ammonia volatilization in waste stabilization ponds plays a major role in nitrogen removal (Ferrara and Avci, 1982; Mara and Pearson, 1986). Steen (1998) studied nitrogen removal in combined DBPs and ABPs with anaerobic pretreatment he reported that more than 50% of the influent nitrogen could not be accounted for in the balance. He assumed that it was volatilized and contributed to 59-73%. Caicedo (2002) considered also similar assumption. However, none of these studies was based on complete nitrogen mass balance. Therefore, their assumptions could not be considered as facts. However, the complete nitrogen mass balance conducted recently by Zimmo (2003) proved that ammonia volatilization in DBPs and ABPs played a minor role in nitrogen removal. He reported that the ammonia volatilization during the study period in any system did not exceed 1.5 % of total influent ammonium. Therefore, it can be said confidently that the calculations for nitrogen volatilization in our system were able to reflect the actual situation. However, for achieving concrete results, it is recommended to conduct ammonia volatilization in further research.

4.6.2 Denitrification and other insignificant nitrogen pathways

According to the nitrogen mass balance equation, the majority of nitrogen in our system was removed occurred via denitrification and other insignificant nitrogen pathways. In the light of literature, the insignificant nitrogen pathways play a negligible role in nitrogen budget. Therefore, most of nitrogen removed could be attributed to denitrification mechanism (Figure 4.19). From chemical point of view, the optimum temperature for occurrence the nitrification and denitrification processes is between 25 to 35^{o}C and the optimum pH is between 7 to 8 (Metcalf and Eddy, 1991). According to the reported conditions in our system, it was shown that most of the conditions crucial for occurrence of nitrification and denitrification processes were available. In reality, nitrification can occur in the upper zone of the water column. However, denitrification occurs into reduced environments in the water column or in the sediments (Caicedo, 2002; Steen, 1998). The relative small ponds depth of 30 to 90 cm resulted in high surface area per volume ratios. It is likely that this will enhance the surface and/or volume related processes of nitrogen removal, such as denitrification and sedimentation. Because the amount of light available per pond volume is higher compared to deeper ponds. This would result in higher algae growth and consequently in an increase in oxygen produced via photosynthesis. This would favour nitrification, the limiting step for denitrification (Zimmo, 2003).

The high contribution of nitrification and denitrification in nitrogen removal via heterotrophic nitrification in the sediment could take place by heterotrophic nitrifying bacteria that can oxidize both inorganic and organic nitrogen compounds (Van Luijn 1997; Verstraete and Alexander, 1973) at low oxygen concentration (Laurent, 1971 as cited by Green *et al.*, 1996). Recently, Zimmo (2003) reported through complete nitrogen mass balance that denitrification was the dominant nitrogen removal mechanism (15-25% of TN). This result supports the result obtained via calculations for nitrogen pathways in our system. On the other hand, several studies concluded that nitrification and denitrification do not play a major role in nitrogen removal (Ferrara and Avci, 1982;

Mara and Pearson, 1986; Caicedo, 2002; Steen, 1998; Javed, 1995). However, these studies based on incomplete mass balance. Therefore, these conclusions cannot be considered in our study. Figure 4.19 shows the overall nitrogen mass balance, the values for ammonia volatilization were calculated by the models of Clement and Merlin (1995) and Stratton (1969) while denitrification was calculated based on nitrogen mass balance (equation 3.4). Insignificant nitrogen pathways were neglected.

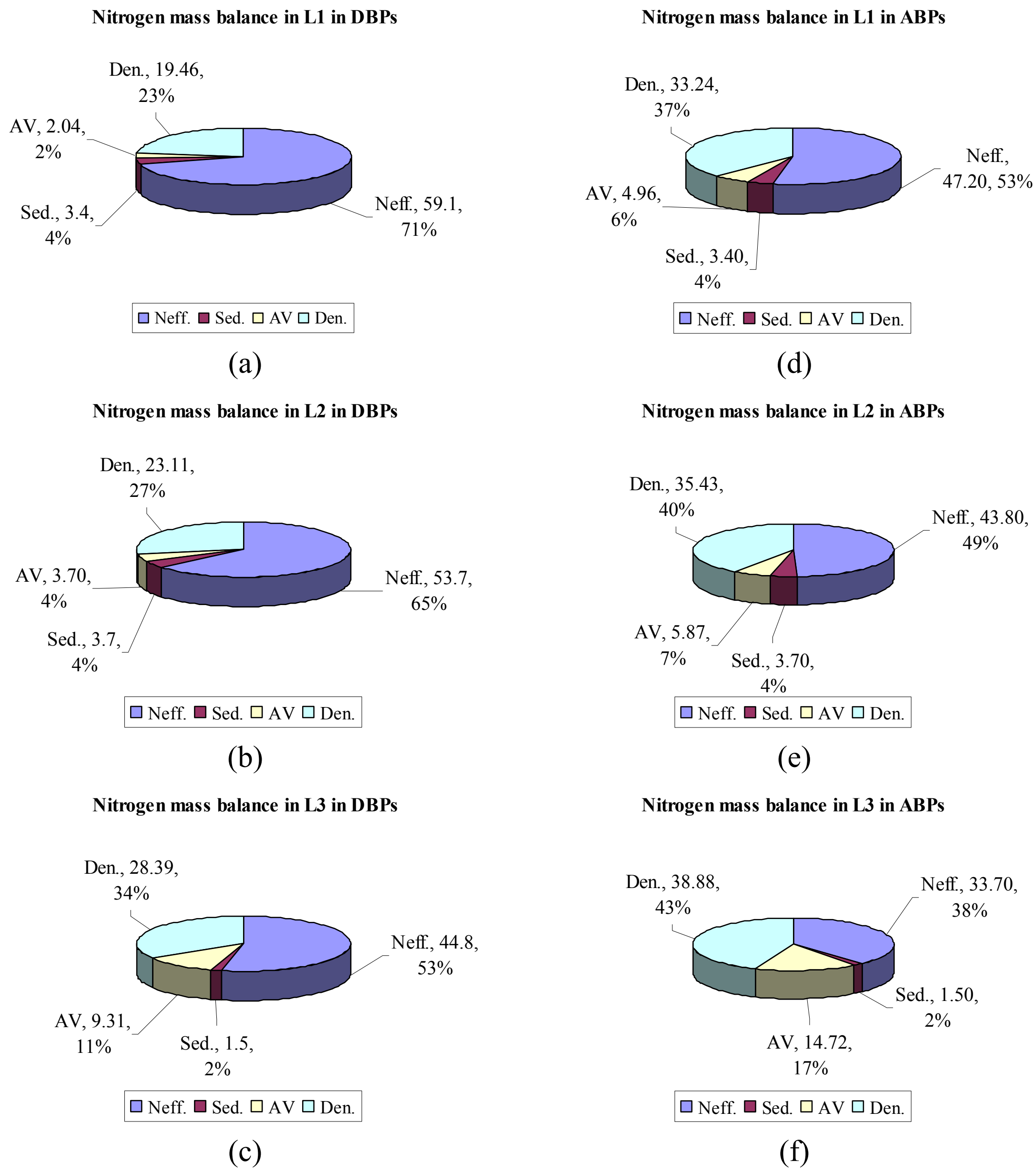


Fig. 4.19. Nitrogen mass balance for the three lines in the two experimental periods each pie chart includes the value of each term (mg/L) and the percentage for this term from TN in the influent. Den, AV, Sed. and $N_{eff,}$ stand for denitrification, ammonia volatilization, sedimentation and concentration of nitrogen in the effluent, respectively. Figures (a,b,c) represent the case in the first, second and third lines of DBPs, respectively, whereas figures (d,e,f) represent the case in the first, second and third lines of ABPs, respectively.

4.7 Land requirement

Different guidelines and regulations are considered worldwide to balance between the reused water quality objective and the financial support that the country has to afford to achieve the desired quality level (Abu Madi, 2004). Table 4.10 shows the effluent quality requirement according to the Jordanian, USEPA and WHO guidelines for restricted irrigation.

Table 4.10. Effluent quality for the three DBPs and ABPs line systems against the Jordanian, USEPA, and WHO standards for restricted irrigation with reclaimed wastewater (mg/L unless otherwise mentioned).

Parameter	DBPs			ABPs			Jordanian	USEPA	WHO
	90	60	30	90	60	30			
pH	7.1	7.6	7.3	7.7	7.8	7.7	6-9	6-9	6-9
COD	258.6	205.0	174.8	158.8	162.6	152.1	500	50-150	200
TSS	60.8	70.9	66.1	133.8	88.5	153.1	200	<30	50
TDS	818.5	828.5	841	773.6	780.6	760.4	2000	500-2000	--
DO	1.19	1.36	1.69	3.7	3.8	4	>2	0.5	--
TKN	59.1	53.7	44.8	47.2	43.0	33.7	50-100	--	--
NH_4^+	41.8	37.6	32.3	30.7	27.2	22.2	25-50	--	--
TN	59.1	53.7	44.8	47.2	43.0	33.7	100	--	10-30
TP	9.4	9	7.8	8	6.8	5.4	15	0.1 -30	--

Source: USEPA, 1992; Al-Lafi, 1996; Bahri, 1998; WERSC, 1998 (as quoted by Abu Madi, 2004).

Jordanian guidelines for restricted irrigation were achieved for the three lines of the treatment system for ABPs. Therefore, the use of the deepest ponds for achievement of these guidelines proved to be more feasible with respect to land requirement which was for the ABPs (according to the exponential-removal curve, $R^2 = 0.8481$) equals to 0.80 m^2 /capita. On the other hand, when USEPA or WHO guidelines for restricted irrigation are considered to determine the quality standards, then the land requirement seems to be high which make the application of such system economically more expensive. The per capita land requirement according to USEPA and WHO guidelines for restricted irrigation in the deepest ponds (measured in m^2 /capita) are respectively 4.3 and 6.9 for DBPs and 1.2, 3.9 for ABPs. Nevertheless, USEPA or WHO guidelines for restricted irrigation, could be considered where land could be available at reasonable price. This does not exclude urban areas where pond systems for nutrient utilisation can be planned outside the city at convenient locations.Table 4.11 shows the land requirement for each depth for DBPs and ABPs against each guideline (excluding land requirement by UASB-septic tank which equals 0.12 m^2/capita). Equal flow rate (75 L/d) was used in calculation land requirement for all the lines.

Table 4.11. Land requirement (in m^2/capita) for each pond depth in DBPs and ABPs to achieve the Jordanian, USEPA and WHO guidelines for restricted irrigation concerning TN, COD and TSS. The flow rate that was used to estimate the land requirement was 75 L/d.

System	Line Depth	Land Requirement according to Jordanian Standards	Land Requirement according to USEPA Standards	Land Requirement according to WHO Standards
TN				
DBPs	90	0.0		6.9
	60	0.0		8.4
	30	0.0		11.7
ABPs	90	0.0		3.9
	60	0.0		5.5
	30	0.0		7.8
COD				
DBPs	90	0.8	3.2	2.6
	60	0.8	3.8	3.1
	30	1.3	7.3	5.9
ABPs	90	0.0[1]	1.2	0.3
	60	0.0[1]	2.4	0.8
	30	0.0[1]	7.1	4.3
TSS[1]				
DBPs	90	0.0[2]	4.3	3.1
	60	0.0[2]	6.8	4.8
	30	0.0[2]	12.4	8.8
land required to comply with all mentioned guidelines for restricted irrigation[3]				
DBPs	90	0.8	4.3	6.9
	60	0.8	6.8	8.4
	30	1.3	12.4	11.7
ABPs	90	0.0[1]	1.2	3.9
	60	0.0[1]	2.4	5.5
	30	0.0[1]	7.1	7.8

1:TSS is increasing with time, therefore it was not included in land requirement calculations, However a tertiary treatment is recommended (a duckweed-covered pond could be a good alternative).
2: TSS concentration in the effluent of the three DBPs lines complied with Jordanian guidelines for restricted irrigation (200 mg/L).
3: The highest value of land requirement among all values listed for each pond was considered.

Land requirement for the deepest line was the least among the three depths in the two experimental periods. It was also shown that a significantly larger land is required to comply with the stringent WHO guidelines rather than the Jordanian guidelines. The consideration of WHO or Jordanian guidelines for determination effluent standards may be referred to the willingness of the country to pay for treating wastewater and to the degree of restriction of types of plants to be irrigated. The reduction of TN to 100 mg/L (Jordanian guidelines for restricted irrigation) did not require any further land after UASB-septic tank, while the reduction of TN to 30mg/L (WHO guidelines for restricted irrigation) required high land 6.9 m^2 /capita (R^2 = 0.994), 3.9 m^2 / capita (R^2 = 0.998) in DBPs and ABPs respectively (excluding associated facilities).

Consequently, land requirement for DBPs is considered higher than that for ABPs in our system. However, land requirement in our system is considerably lower than the reported land requirement reported for waste stabilization ponds by Burka (1996, as quoted by Mara and Pearson, 1998) who mentioned that land required for WSP in Germany ranges between 10 and 15 m^2/capita to comply with EU directive on wastewater treatment. Actually, this area requirement may be reduced through optimization of ABPs and DBPs making these systems more attractive. Zimmo (2003) reported that DBPs require 2-4 m^2/capita (depending on influent strength of wastewater and effluent guideline requirements). He reported a decrease in land requirement by 10% when ABPs are used. However, an integrated system combining ABPs and DBPs could be a proper solution for nutrients and FC removal at reasonable land requirement (Steen, 1998).

4.8 UASB-septic tank followed by pond system

It was mentioned before that, the problem with stabilization ponds system that is considered as a cheap wastewater treatment method is the high land requirement. Therefore, the main objective of this research was to test the ability of UASB-septic tank and increasing the depth of the ponds to significantly reduce land requirement and therefore make this cheap alternative more attractive mainly for the poor regions of the world particularly in Palestine. The main results of this research concerning the removal of main pollutants forms are depicted in Tables 4.12 and 4.13. In the calculations for removal efficiencies of different pollutants in the aforementioned tables, the considered initial concentrations were the influent to the UASB-septic tank. Therefore, there are differences in the removal efficiencies for different pollutant forms.

It is clear from Tables 4.12 and 4.13 that integration in treatment is found between the UASB-septic tank and the pond system in the two experimental periods. Most of the organic load was removed in the UASB-septic tank, while most of the nutrients load was removed in the pond system. This result was found in several studies in literature when additional treatment for the effluent from anaerobic systems was recommended to meet secondary quality standards in terms of oxygen consuming substances and odor problems (Buitrago *et al.,* 1994; Jewell, 1987; Giraldo, 1993; Vieira, 1992). Similar result was reported by Al-Juaidy (2001) who concluded that UASB-septic tank presented an effective on-site treatment. He suggested that another post treatment unit should be added to reach the acceptable requirements for reuse of wastewater and to treat nutrients and pathogens. It is shown in Table 4.12 and 4.13 that the overall efficiency for the system (UASB-septic tank and pond system) throughout any line was almost the same.. For example the overall COD removal efficiencies in the integrated system for the second experimental period (lines with depth 90, 60 and 30 cm) were 87%, 86% and 87%, respectively. Similar results were found for other parameters. This actually encourages the use of deepest ponds rather than shallowest ponds for large-scale application.

Table 4.12 Research results for the average effluent concentrations and removal efficiencies (%) for the overall system (including results for UASB-septic tank) during the first experimental period under the imposed operational conditions. The standard deviations are presented between brackets.

Par.	**Inf.†**	**Effluent† of UASB**		**D_1L_1**		**D_2L_1**		**D_3L_1**		**D_1L_2**		**D_2L_2**		**D_3L_2**		**D_1L_3**		**D_2L_3**		**D_3L_3**	
	Avg.	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%
pH	7.5	7.4 (0.14)	--	7.4 (0.1)	--	7.4 (0.2)	--	7.4 (0.1)	--	7.5 (0.1)	--	7.5 (0.3)	--	7.6 (0.1)	--	7.7 (0.1)	--	7.7 (0.1)	--	7.8 (0.1)	--
DO	nm	0	--	0.06 (0.02)	--	0.5 (0.19)	--	1.19 (0.44)	--	0.15 (0.07)	--	0.16 (0.03)	--	1.36 (0.48)	--	0.15 (0.06)	--	0.22 (0.1)	--	1.69 (0.78)	--
TS	nm	nm	--	nm	--	nm	--	nm	--	nm	--	nm	--	nm	--	nm	--	nm	--	nm	--
TSS	614	117 (19)	80 (4.6)	146 (13)	76 (2)	114 (30)	81 (5)	61 (19)	90 (3)	143 (13)	77 (2)	91 (16)	85 (3)	71 (18)	88 (3)	133 (22)	78 (4)	83 (31)	87 (5)	66 (23)	89 (4)
TDS	nm	973 (31)	--	913 (35)	--	943 (8)	--	819 (30)	--	826 (20)	--	852 (37)	--	829 (21)	--	795 (33)	--	851 (29)	--	841 (42)	--
COD	1185	493 (95)	58 (7)	447 (124)	62 (10)	295 (30)	75 (2)	258 (45)	78 (4)	382 (76)	68 (6)	213 (44)	82 (4)	205 (83)	83 (7)	300 (24)	75 (2)	249 (58)	79 (5)	175 (56)	85 (5)
TP	14	14.2 (1.1)	-2 (9.8)	12.6 (0.3)	10 (2)	10.5 (0.4)	25 (3)	9.4 (0.5)	33 (3)	12.7 (0.3)	9 (2)	10.7 (0.4)	23 (3)	9.0 (0.2)	35 (2)	13.5 (0.6)	3 (4)	11.1 (0.7)	21 (5)	7.8 (1.5)	44 (11)
NH_4^+	58.9	59 (4.4)	-0.4 (8)	55.2 (2.2)	6 (3.7)	52.8 (2.7)	10 (4.7)	41.8 (0.9)	29 (1.5)	51.9 (3.0)	12 (5.1)	47.9 (4.0)	19 (6.8)	37.6 (1.7)	36 (2.9)	44.6 (1.0)	24 (1.7)	40.6 (0.5)	31 (0.9)	32.3 (2.5)	45 (4.2)
TKN	78	68 (6.7)	12 (10)	76.8 (6.6)	2 (8.5)	67.3 (6.6)	14 (8.5)	59.1 (8.2)	24 (10.5)	72.5 (6.8)	7 (8.7)	65.1 (3.7)	17 (4.7)	53.7 (4.5)	31 (5.7)	70.6 (4.9)	9 (6.2)	56.3 (6.3)	28 (8.1)	44.8 (6.2)	43 (7.9)

†: source for all values of parameters in the influent and effluent and removal efficiencies of UASB-septic tank: Al-shayah (2005).
All parameters in mg/L except: pH no unit.
Par. = parameter; Avg. = average; inf. = influent; nm = not measured.

Table 4.13 Research results for the average effluent concentrations and removal efficiencies (%) for the overall system (including results for UASB-septic tank) during the second experimental period under the imposed operational conditions. The standard deviations are presented between brackets.

Par.	Inf.†	Effluent[1] of UASB		A_1L_1		A_2L_1		A_3L_1		A_1L_2		A_2L_2		A_3L_2		A_1L_3		A_2L_3		A_3L_3	
	Avg.	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%	Avg.	%
pH	7.5	7.4 (0.14)	--	7.8 (0.1)	--	8.0 (0.1)	--	8.2 (0.1)	--	7.8 (0.1)	--	8.0 (0.1)	--	8.0 (0.1)	--	7.8 (0.1)	--	8.0 (0.1)	--	8.3 (0.0)	--
DO	nm	0	--	1.3 (0.3)	--	2.7 (0.9)	--	3.7 (1.2)	--	1.5 (0.8)	--	2.9 (1.0)	--	3.8 (0.4)	--	1.7 (0.5)	--	3.1 (0.6)	--	4.0 (0.3)	--
TS	nm	nm	--	1051 (35)	--	1030 (52)	--	1085 (40)	--	1057 (45)	--	1064 (36)	--	1029 (56)	--	930 (196)	--	1034 (117)	--	1176 (53)	--
TSS	614	117 (19)	80 (4.6)	148 (35)	76 (5.7)	162 (47)	74 (7.7)	134 (17)	78 (2.7)	128 (17)	79 (2.7)	143 (26)	77 (4.2)	89 (32)	86 (5.2)	131 (73)	79 (11.9)	136 (28)	78 (4.6)	153 (26)	75 (4.2)
TDS	nm	973 (31)	--	801 (24)	--	769 (8)	--	774 (6)	--	797 (10)	--	778 (6)	--	781 (12)	--	784 (23)	--	765 (17)	--	760 (12)	--
COD	1185	493 (95)	58 (7)	198 (43)	83 (3.6)	186 (44)	84 (3.7)	159 (38)	87 (3.2)	230 (54)	81 (4.6)	208 (65)	82 (5.5)	163 (32)	86 (2.7)	215 (53)	82 (4.4)	200 (42)	83 (3.6)	152 (32)	87 (2.7)
TP	14	14.2 (1.1)	-2 (9.8)	11.3 (0.9)	19 (6.7)	9.2 (0.7)	34 (4.9)	8.0 (1.0)	43 (6.8)	10.6 (0.5)	24 (3.4)	9.2 (0.9)	35 (6.3)	6.8 (1.2)	52 (8.7)	10 (0.9)	28 (6.3)	7.8 (0.6)	44 (4.4)	5.4 (0.7)	61 (5.3)
NH_4^+	58.9	59 (4.4)	-0.4 (8)	50.8 (7.4)	14 (12.6)	38.6 (5.3)	34 (0.9)	30.7 (5.8)	48 (9.9)	56.1 (8.3)	5 (14)	39.7 (6.9)	33 (11.7)	27.2 (6.1)	54 (10.4)	42.0 (7.3)	29 (12.4)	31.4 (5.4)	47 (9.2)	22.2 (3.5)	63 (5.9)
TKN	78	68 (6.7)	12 (10)	74.7 (3.3)	4 (4.2)	55.8 (3.6)	28 (4.7)	47.2 (2.7)	39 (3.5)	73.5 (3.3)	6 (4.2)	57.1 (6.7)	27 (8.6)	43.8 (4.4)	44 (5.6)	59.2 (7.6)	24 (12.7)	45.7 (5.5)	41 (7.0)	33.7 (6.5)	57 (8.3)

†: source for all values of parameters in the influent and effluent and removal efficiencies of UASB-septic tank: Al-shayah (2005).
All parameters in mg/L except: pH no unit.
Par. = parameter; Avg. = average; inf. = influent; nm = not measured.

4.9 Investigation of the causes behind duckweed death

As mentioned before, there are suitable conditions in which duckweed can survive and contribute in the treatment process. However, different limits were found in literature for the highest and/or lowest concentration of different pollutants in wastewater in which duckweed can survive. Therefore, a number of batch experiments were carried out in parallel along the main experiments to determine the criteria in which duckweed can survive and the specific reasons for the death of duckweed. The main tests that were carried out were related to the effect of wastewater strength and controlling the environmental conditions and algal growth.

4.9.1. Controlling the environmental conditions related to algal growth in DBPs

In the pilot plant system, during the first experimental period, algal growth was noticed in the DBPs system. Therefore, it was expected that algal growth was the reason behind the death of duckweed, due to the increase in pH and therefore, the ammonia toxicity. In order to prevent algal growth, the ponds were covered by black plastic cover (Photo 4.1). However, oxygen replenishment, and small amount of sunlight which are vital for the survive of duckweed, were maintained through making "holes" in the plastic cover, air and sunlight also entered to contact the wastewater surface in the ponds through the small area between the plastic cover and the wall of the pond. The cover was removed regularly twice a week. This method prevented the growth of algae. However, in practice other methods need to be investigated in order to be applied for large-scale systems.

The reason for discussing effect of algal growth on duckweed death is not the fact that algae are competitors for nutrients, because nutrients are available sufficiently (as discussed before). However, some algae are very harmful to duckweed as they clog and wrap themselves around the plant roots while others produce toxic compounds they may cause destruction of duckweed (Edward *et al.,* 1992). Entzeroth *et al.* (1986) reported that the extracts of some blue green algae inhibit growth of duckweed. Moreover, Gleason and Case (1986) reported that cyanobacterin which is released by the blue green algae inhibited the growth of *Lemna.* Consequently, similar reasons could be given to justify duckweed mortality in the first experimental period.

Covering the ponds was successful in algal growth prevention. This was noticed from the change of the color and odor of wastewater. Consequently, the pH of wastewater decreased slightly. The difference between pH values before and after the coverage of ponds ranged between 0.66-1.51. However, duckweed death after the coverage continued to occur. Therefore, it was concluded that there are other reason(s) behind duckweed death except the effect of algae.

(a)

(b)

Photo 4.1. a- The covered ponds (five days per week). b- Cover removal (for two days per week).

4.9.2. Investigation the effect of the wastewater culture on duckweed growth

As mentioned before, the duckweed culture was the pilot DBPs of Birzeit University. These ponds were out of operation since at least one year. The wastewater was expected to have a low concentration of nutrients, and the duckweed grown on these ponds were adapted to low strength of wastewater. The characteristics of wastewater from this culture were tested. Three samples were tested from each pond from where the duckweed was taken. The main results of this test are shown in Table 4.14.

Table 4.14. Main characteristics for the wastewater in the duckweed culture. Each parameter was tested three times the tabulated figures represent the concentration (± S.E.).

Pond #	n	T		pH		DO		NH_4^+		COD	
		range	Avg. (S.E.)	range	Avg. (S.E.)	range	Avg. (S.E.)	range	Avg. (S.E.)	range	Avg. (S.E.)
1	3	25.5-27	26.5 (1.1)	8.1-8.2	8.14 (0.06)	2.85-3.0	2.92 (0.09)	54.5-65	59.8 (5.9)	525-601	560.3 (42.6)
2	3	26-27.5	26.9 (0.9)	8.2-8.3	8.26 (0.06)	3.11-3.28	3.21 (0.1)	29-40	33 (6.9)	268-301	287.3 (19.5)
3	3	26-27.5	26.7 (0.9)	8-8.2	8.1(0.11)	1.95-2.11	2.05 (0.1)	36-45	40.9 (5.2)	348-412	380 (36.2)

Avg. = average value. All values are in (mg/L) except for pH (dimensionless)

According to Table 4.14, it is clear that the concentrations of nutrients in the original culture of duckweed are in the same range of the nutrients in our pilot plant. This was not expected because the wastewater in the culture is originally from Birzeit Campus that is classified as low to medium strength wastewater, but this could be attributed to the evaporation of the large quantities of water from the ponds that resulted in increasing the concentration of pollutants in wastewater. Therefore, the conclusion from this test suggested that there is no direct relation between the wastewater characteristics and the duckweed death in the pilot plant.

4.9.2.1 Effect of wastewater strength on duckweed growth

Referring to the results achieved in the previous two tests, and the notes taken from the ponds of the pilot plant. The dilution test for the effluent of the UASB-septic tank was carried out. Small buckets were used to perform this test. The test was performed in two stages. The first stage was aimed to test if the dilution of wastewater will result in duckweed survival. The second stage aimed to know the critical dilution that the duckweed withstands without die off.

4.9.2.2 First dilution test

The main objective of this test was to verify if the dilution of wastewater would result in duckweed survival. Two groups (three buckets per group) of buckets were used to perform the experiment (the details of apparatus are mentioned previously). The same dilution factor (1:1) was used for the entire buckets (Table 4.15). The first group of buckets exposed to the same conditions as the ponds and the second group was put in a shaded area near the pilot plant. Daily monitoring was made for the buckets. The evaporated quantity of water was made up with water of the same dilution factor. Duckweed survived in the diluted wastewater in both groups. Dark Green cover of duckweed was noticed and the duckweed cover density increased in the buckets that indicated that growth took place. No difference in duckweed mat was noticed between the two groups, this suggests that solar radiation did not affect duckweed growth.

Table 4.15. Main characteristics of the wastewater in the test buckets of the first and second group. The figures represent the range and average (± S.E.) for the concentration of the pollutant in the three buckets (one sample was tested for each bucket per week).

Group #	n	T		pH		DO		NH_4^+		COD		TP	
		range	Avg (S.E.)	range	Avg (S.E.)	range	Avg (S.E.)	range	Avg (S.E.)	range	Avg (S.E.)	range	Avg (S.E.)
1	4	21-23	22 (2)	7.18-7.24	7.21 (0.1)	5.01-5.4	5.21 (0.4)	23.6-31	27.3 (7.3)	203-262	232.5 (58)	5-8	6.5 (2.9)
2	4	21-22	21.5 (1)	7.3-7.76	7.53 (0.5)	3.95-4.7	4.33 (0.7)	23.1-28.3	25.7 (5.1)	221-261	241 (39)	3.8-7	5.4 (3.1)

Avg. = average value. All values are in (mg/L) except for pH (dimensionless).

According to the results obtained in Table 4.15, the concentration of all parameters were within the range of concentrations achieved in the third pond in each line were duckweed death was noticed. However, ammonium was slightly higher. Nevertheless, ammonium concentration in the original ponds was almost similar to the latter concentration in the DBPs of our system (Table 4.14). Therefore, it was concluded that there is another parameter or a combination of more than one parameter whose concentration was decreased to some limit by dilution caused the duckweed death in pure wastewater. Photo (4.2) shows the buckets at the beginning and end of the test.

(a) (b)

Photos 4.2. A test bucket exposed to the same conditions as the ponds. a- The bucket at the beginning of the test. b- The same bucket at the end of the test with the duckweed plant crowded because of growth.

4.9.2.3 Second Dilution Test

The aim of this test is to determine the concentration at which duckweed cannot survive. Three groups of small containers were used to perform this test (photo 4.3). Each group contained number of containers that include wastewater at different dilutions (0%, 25%, 50%, and 75%). The first group was exposed to the same conditions as the ponds, the second group was put in a shaded area and the third group was covered in a way similar to that of the ponds. The main results obtained in this test were that the duckweed survived in all the buckets and containers from the three groups except for the buckets and containers that contained wastewater without dilution (0% dilution factor).

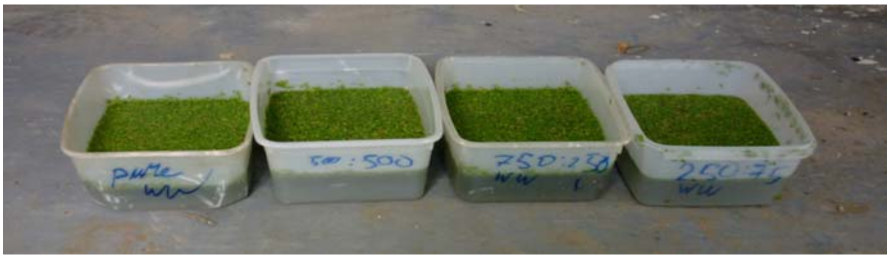


Photo 4.3. The small containers with different dilutions (the shape of the buckets in this test is the same as that in photo 4.2).

Table 4.16 depicts the main results achieved for the three groups in this batch experiment. According to literature, the range of temperature in which duckweed live without any thermal stress is (1 to 3 ^{0}C) (Wildschut, 1983) and (33 ^{0}C to 34 ^{0}C) (Oron *et al*., 1985). Then the range of temperature during the experiments was suitable for the duckweed growth. The radiation in the Ramallah district is 150-260 W/m^2 (ARIJ, 1997) is lower than the very intense radiation of the Negev desert (300-600 W/m^2) which was not found to reduce the duckweed production (Steen, 1998). Accordingly, the light intensity is not expected to be as one of the reasons for duckweed death. It was reported in literature that, *Lemna gibba* can grow in a pH range of 3 to 10 (Zirschky and Reed, 1988; Landolt, 1986; Wildschut 1983). Therefore, the range of pH in the duckweed-based ponds was optimal for the growth of duckweed.

COD concentration was also within the range in which duckweed can live. Mandi (1995) demonstrated that *Lemna gibba* could be used in primary treatment of wastewater with COD ranging from 305-530 mg /L. Similarly, Steen (1998) reported also that the initial

COD concentration range applied was 254–600 mg COD /L. Actually, COD value in the first pond of each treatment line increased occasionally above 600 mg/L. Nevertheless, COD concentration never exceeded 500 mg/L in the last pond of any line, but duckweed death was noticed also in these ponds, but "death rate" in these ponds was observed to be lower than that of the first ponds.

Table 4.16. Main characteristics for the wastewater in the test buckets of the three groups. The figures represent the average for the concentration of the pollutants (one sample was tested for each bucket per week). Note that the dilution factor X:Y means wastewater quantity : fresh water quantity.

Condition	T	pH	DO	NH_4^+	COD	PO_4	TSS	TKN
Dilution	**Pure wastewater**							
Outside	24.5	7.4	0.0	55.1	867.0	12.2	82.0	71.2
Shaded	24.0	7.2	0.0	57.0	846.7	13.0	88.0	73.1
Covered	24.5	7.4	0.0	52.8	860.0	12.8	74.0	68.0
Dilution	**3:1**							
Outside	24.0	7.4	1.3	40.8	642.5	9.3	60.7	52.8
Shaded	23.0	7.6	1.2	42.2	637.3	9.6	65.2	54.2
Covered	26.0	7.3	1.0	39.1	622.2	9.5	56.3	50.3
Dilution	**1:1**							
Outside	24.0	7.3	3.2	23.1	385.2	5.7	37.0	35.8
Shaded	23.0	7.5	2.9	24.9	368.3	5.6	33.0	31.0
Covered	24.0	7.4	2.0	23.8	370.1	5.4	35.0	33.6
Dilution	**1:3**							
Outside	24.5	7.5	4.8	16.7	262.2	3.8	24.8	21.5
Shaded	23.5	7.4	5.1	17.2	252.1	3.9	26.6	22.1
Covered	24.0	7.4	5.5	16.0	260.1	3.6	23.0	20.6

NH_4^+ in the duckweed ponds did not exceed 56.2 mg/L. Steen (1998) demonstrated that a concentration of NH_4^+ 25–100 mg-N/L did not affect the growth of duckweed. Rejmankova, (1979) stated that duckweed can tolerate N- concentration up to 375 mg/L, however, it is not specified that the nitrogen was in the form of NH_4^+ or NH_3. However, Zimmo (2003) reported that when wastewater from Al-Bireh was used which contained nitrogen concentrations of >100 mg/l resulted in complete decay of duckweed. Actually this result is important for this research, but the current system differs because total nitrogen concentration did not exceeded 100 mg/L, and significantly lower concentrations were observed in the last ponds of each line. Caicedo *et al*. (2000) reported that a negative effect on growth of *Spirodela* at higher concentration than 50 mg (NH_4^+ + NH_3)-N/L. Moreover, Ammonia toxicity could be the reason because at NH_3-N concentrations of 3-7.16 mg/L, the growth rate is inhibited (Wang, 1991; Ghosh, 1994; Zimmo, 2003; Wang, 1991; Clement and Marlin, 1995). The concentration of ammonia was calculated before, but it never exceeded 3 mg/L (according to calculations). Therefore, ammonia toxicity was not the reason for duckweed death. Therefore, in order to have concrete results, further research is required to study other probable reasons for duckweed death such as heavy metals concentration.

Chapter Five

Conclusions and Recommendations

5.1. Conclusions

- The effect of depth on the environmental conditions in the ponds system for the two experimental periods was as follows:

  1- Significant increase in DO concentration was found among the effluent of the successive ponds in each line for the two periods ($\rho \leq 0.05$). However, no significant change in DO concentrations was found as the depth of the ponds decrease in DBPs and ABPs experimental periods ($\rho \geq 0.05$).

  2- pH values in both systems decreased slightly by increasing depth. pH values at the surface of water column for the 90cm, 60cm, and 30 cm depths were (7.4, 7.5, and 7.7) and (8.0, 8.1, and 8.3) in DBPs and ABPs, respectively. No significant difference was found in average pH in the influent for the two experimental periods ($\rho \geq 0.05$). The effect of depth on pH value during the two experimental periods was as follows:

     a. Significant difference in pH values for the first experimental period was found among the three lines ($\rho \leq 0.05$), except between the first pond of each line ($\rho \geq 0.05$).

     b. No significant difference in pH values was observed for the three lines in the second experimental period ($\rho \geq 0.05$) except for the third pond of each line ($\rho \leq 0.05$).

- Effect of depth on removal efficiency of different pollutants forms was as follows:

  1- COD removal efficiency was found to increase as the depth of the pond decrease in the two experimental periods. Nevertheless, average COD removal rates were observed to be significantly higher in the line of deepest ponds compared to line of shallowest ponds. COD removal efficiencies in the two experimental periods were as follows:

a. Significant increase in COD removal efficiency was observed only in the third pond in each line for the DBPs period ($\rho \leq 0.05$). The removal efficiencies in the three lines (90, 30 and 30 cm respectively) were 62.5 ± 5.7%, 70.6 ± 11.6% and 75.4 ± 4.1%.

b. No significant increase in removal efficiency was observed among the three lines of ABPs ($\rho \geq 0.05$). The removal efficiencies in the three lines (90, 30 and 30 cm respectively) were 51.6 ± 3.2, 53.4 ± 4.3, and 54 ± 1.1%. Moreover, significant increase in COD removal efficiency was observed among the successive ponds in the first and third lines ($\rho \leq 0.05$).

2- No relation was found between TS, TDS and TSS removal and the depth. TSS removal efficiency in DBPs period ranged between 58-64%. However, negative TSS removal efficiency was observed in ABPs.

3- TP removal efficiency was found to increase as the depth of the pond decrease in the two experimental periods. Moreover, significant increase in removal efficiency was found between the first line (90 cm) and the other two lines (30 and 60 cm), ($\rho \leq 0.05$). However, no significant difference was found in TP removal at the second and third lines ($\rho \geq 0.05$). TP removal efficiency in the two periods was as follows:

a. The removal efficiencies of TP for the three DBPs lines (90, 30 and 30 cm respectively) were 38.5 ± 4, 40.8 ± 3 and 48.5 ± 9.2%.

b. The removal efficiencies of TP for the three ABPs lines (90, 60 and 30 cm, respectively) were 37.6 ± 6.4%, 46.4 ± 10.5%, and 57.6 ± 5.6%.

- The effect of depth on nitrogen removal efficiency in DBPs and ABPs ponds was as follows:

1- Higher NH_4^+ removal was observed at ponds with lower depth in the two experimental periods (DBPs and ABPs). Significant increase in removal efficiency was observed between the successive ponds in each line ($\rho \leq 0.05$). Likewise, significant increase in removal efficiency was observed among the effluent of the three mentioned lines ($\rho \leq 0.05$). NH_4^+ removal in the two systems was as follows:

a. Removal efficiencies achieved in the three DBPs lines (90, 60 and 30 cm respectively) were, 30.9 ± 1.6, 37.9 ± 1.7 and 46.6 ± 5.2% ($\rho \leq 0.05$). Nevertheless, maximum NH_4^+ removal rate was achieved in the deepest ponds this could be attributed to highest flow at which this line is operated.

b. Removal efficiencies observed in the three DBPs lines (90, 60 and 30 cm respectively) were 51.2 ± 1.9, 56.9 ± 2.9 and 64.5 ± 2.8%, respectively. Nevertheless, highest NH_4^+ removal rate was achieved also in the deepest ponds.

2- Higher TKN removal was observed at lower ponds depth in the two experimental periods. Significant increase in removal efficiency was observed between the successive ponds in each line ($\rho \leq 0.05$). TKN removal in the two systems was as follows:

c. Significant increase was observed in the removal efficiency among the three treatment lines for DBPs period ($\rho \leq 0.05$). The removal efficiencies of the three lines (90, 30 and 30 cm respectively) were 29.4 ± 6.8%, 35.7 ± 5.2% and 44.5 ± 6.3%, respectively. Nevertheless, highest TKN removal rates were achieved in the deepest ponds compared to shallower ponds.

d. No significant difference was noticed between line one (90 cm depth) and line two (60 cm depth) in TKN removal rate ($\rho \geq 0.05$). The removal efficiencies achieved in the three lines (90, 30 and 30 cm respectively) were 45.4 ± 3.1%, 49.3 ± 4.3% and 61.1 ± 4.5%, respectively. Nevertheless, maximum TKN removal rate was achieved in the deepest ponds compared to other shallower ponds ($\rho \leq 0.05$).

Main differences in removal between ABPs and DBPs (comparing similar pond depths) were as follows:

1- Significantly higher DO and pH concentrations were achieved in ABPs compared to DBPs. These results could be attributed to the photosynthetic activity of algae.

2- Lower COD removal efficiency was achieved in ABPs compared to DBPs.

3- Higher TP, NH_4^+ and TKN removal efficiencies were achieved in ABPs compared to DBPs in the reported conditions.

4- Higher sedimentation rate was achieved in ABPs compared to DBPs.

- Duckweed did not survive in the anaerobically pretreated municipal wastewater at the reported conditions.
- Ammonium concentration was found in the range 32.3 mg/L and 60.6 mg/L in the ponds system which was not toxic to duckweed.
- Covering of duckweed ponds succeeded in removal of algae but it failed to rescue duckweed. Therefore, it was concluded that algal growth in our system was not the reason for the death of duckweed.
- During the investigation for duckweed death using batch reactors that include wastewater (effluent of UASB-septic tank) with different concentrations and exposed to different environmental conditions (covering, shading and none), no effect was found for the environmental conditions. However, duckweed was not affected in diluted wastewater with the following concentrations (TKN, NH_4^+ and COD respectively) 71.2, 55.1 and 867 mg/L, respectively.
- Deeper DBPs and ABPs (90 cm) proved to be more feasible compared to shallower ponds (60 and 30 cm). 50% of land requirement could be saved when deepest ponds were applied.
- The land requirement in our system for the investigated parameters to comply with WHO guidelines for restricted irrigation (excluding land requirement for UASB-septic tank and other treatment plant facilities) were 6.9 and 3.9 m^2/capita for DBPs and ABPs, respectively. The influent quality during the ABPs period complied with the Jordanian guidelines for restricted irrigation. However, land requirement according to Jordanian guidelines for restricted irrigation for the DBPs was 0.8 m^2/capita.

5.2. Recommendations

1. On the basis of results presented in this research, deepest ponds (90 cm) are highly recommended for large scale application rather than shallower ponds (60 and 30 cm).
2. The application of UASB-septic tank-followed by ponds system is recommended especially in the Palestinian rural areas for enabling reuse of treated effluent for restricted irrigation in the increasing water-scarce territories and decrease environmental problems.
3. According to the achieved results in this research, ABPs are more promising than DBPs as a post-treatment unit after the anaerobic pretreatment unit in terms of land requirement, operation and maintenance costs.
4. DBPs are not efficient as post-treatment unit for the high strength-black wastewater in Palestine. Nevertheless, this system could achieve better results if it is used in treatment of low-strength wastewater or gray wastewater.

Recommendations for further research

- ⊙ UASB-septic tank followed by DBPs system should be investigated with domestic wastewater rather than municipal for detection the feasibility of such system in domestic applications mainly for rural areas.
- ⊙ For achieving concrete results, another range of depths has to be tested, mainly depths > 90cm. Furthermore, ammonia volatilization, nitrate and nitrite have to be measured in order to achieve complete nitrogen mass balance.
- ⊙ For using the same system in completion this research, the holding tank has to be changed in order to reduce the possibility of accumulation of sediment in its bottom. Moreover, the suction point has to be installed in the center of the holding tank.
- ⊙ The concentrations of heavy metals (mainly CU, Cr, and Hg) have to be measured in order to detect if there is a relation between duckweed death and heavy metals concentration.
- ⊙ The performance of UASB-septic tank followed by pond system should be investigated during winter period at lower ambient temperature.

References

Abdel Wahhab R., Lubberding H.J., and Alaerts G.J. (1995). Copper and Chromium (III) uptake by duckweed. *Wat. Sci. Tech*., **32** (11), 105-110.

Abeling U. and Seyfried C.F. (1992). Anaerobic-aerobic treatment of high-strength ammonia wastewater-nitrogen removal via nitrite. *Wat. Sci. Tech*., **26**, 1007-1015.

Abu Madi M., Al-Sa`ed R., Braadbart O. and Alaerts G. (2000) Selection criteria for appropriate sanitation in the Palestinian rural and semi-urban communities. Proceedings of the International Symposium on Water Sector Capacity Building and Research in Palestine, Birzeit University, Birzeit, West Bank, Palestine

Abu-Madi, M. (2004). Incentive systems for wastewater treatment and reuse in irrigated agriculture in the MENA region: Evidence from Jordan and Tunisia. PhD thesis, TU-Delft and UNESCO-IHE, Delft, the Netherlands.

Alaerts G.J., Mahbubar M.R. and Kelderman P. (1996). Performance of a full-scale duckweed-covered sewage lagoon. *Wat. Res,*. **30**, 843-852.

Al-Jabari F.M. (2003) Domestic wastewater treatment in an integrated UASB/algae and duckweed based pond system. MSc thesis at Birzeit University-Palestine.

Al-Juaidy A. (2001). Enhanced pretreatment of black wastewater from Birzeit University using UASB technology. MSc thesis at Birzeit University-Palestine.

Al-Nozaily F. (2001). Performance and process analysis of duckweed-covered sewage lagoons for high strength sewage. The case of Sana'a Yemen. Ph.D. Thesis, IHE, The Netherlands.

Al-Sa`ed R. (2000) Obstacles and chances to cut pollution load discharges from the urban Palestine. Proceedings of the 1st Scientific Meeting at Hebron Polytechnic University, Hebron, West Bank, Palestine

Al-Shayah M.N. (2005) Community on-site anaerobic sewage treatment in a UASB-septic tank system . MSc thesis at Birzeit University-Palestine.

Anon (1989). Natural systems vs. the black box. *J. Bio Cycle*, **30**, (6), 68-69.

APHA (1995) Standard methods for the examination of water and wastewater. 18th edition, New York.

Aquasan L.C. (1995). The Aquasan process: wastewater treatment, food production and water reclaimation. Aquasan L.L.C: Bethesda, USA.

ARIJ (1996) Environmental profile for the West Bank, Vol. 7, Applied Research Institute-Jerusalem, Palestine.

Arthur J. P (1983). Notes on the design and operation of waste stabilisation ponds in warm climates of developing countries, World Bank Technical Paper No. 6, IBRD, Washington, DC.

Asano T. (1998). Wastewater reclamation and reuse. Water quality management library Vol. **10**. Technomic Publications, Lancaster, Basel.

Awadallah W.M. (unpublished). Wastewater treatment by duckweeds with emphasis on seasonal variations in plant growth and nitrogen removal.

Bogte J. J., Breure A. M., Andel J. G. v. and Lettinga G. (1993). Anaerobic treatment of domestic wastewater in small scale UASB reactors. *Wat. Sci. Tech*., **27**(9), 75-82.

Boniardi N, Rota R (1999). Effect of dissolved metals on the organic load removal efficiency of Lemna gibba . Water research, **33** (2), 530-538.

Bonomo L., Pastorelli G. and Zambon N. (1996). Advantages and limitations of duckweed-based wastewater treatment systems. 5th *International Conference on Wetland System for Water Pollution Control*, V3/2, 1-8, Vienna.

Brix H. and Schierup H. H. (1989). The use of aquatic macrophytes in water pollution control. *Ambio*, **18**, 100-107.

Buddhavarapu L.R. and Hancock S.J. (1991). Advanced treatment for lagoons using duckweed. Wat. Environ. Technol. **3**: 41-44.

Caicedo Bejrano J.R. (1995). Effesct of ammonia, pH, and organic matter on growth of Spirodella pollerhiza. MSc. Thesis, E.E. 1984, International Institute of infrastructural, Hydraulic, and Environmental Engineering.

Caicedo J. R., van der Steen N.P. Arce O. and Gijzen H.J. (2000). Effect of total nitrogen concentration and pH on growth rates of duckweed (Spirodela polyrrhiza). *Wat. Res.,* **15**, 3829-3835.

Charpentier J.R., Garnier J. and Flaugnatti R. (1987). Txicity and bioaccumulation of Cadimium in experimental cultures of duckweed, Lemna polyrrhiza *L. Bull Env. Contam.Toxic.* **38** (15), 1055-1061.

Clark J.R., Van Hassel J.H., Nicholson R.B., Cherry D.S. and CCairns J.R. (1981) Accumu;ation and depuration of metals by duckweed (Lemna perpusilla). *Ecotox. Environ. Safety* **5,** 87-96.

Clement B. and Merlin G. (1995). The contribution of ammonia and alkalinity to landfill leachate toxicity to duckweed. *Sci. Total Environ.* **170**, 71-79.

Culley D.D., Gholson J.H., Chisholm T.S., Standifer L.C. and Epps E.A. (1978). Water quality renovation of animal waste lagoons utilising aquatic plants. *U.S. EPA, Ada*, Oklahoma. pp. 166.

DeBusk T.A. and Ryther J.H. (1981), Effect of seasonality and plant density on the productivity of some fresh water macrophytes. *J. Aquat. Bot*. **10**,133-142.

Doyle J.S. (1977) Effects of low levels dietary Cadmium in animals. *J. Environm. Qual.* **6**, 111-117.

Edwards P. (1980). Food potential of aquatic macrophytes. *ICLARM Studies and Reviews* **5**, 51 pp.

Edwards P. Hassan M.S. Chao C.H. and Pacharapraciti C. (1992) Cultivation of duckweed in septage-loaded earthen ponds. *J. Bioresource Technology* **40**, 109-117.

Edwards, P., Hassan, M.S., Chao, C.H. and Pacharaprakiti, C. (1992). Cultivation of duckweeds in septage-loaded earthen ponds. *Bioresource Techn.* **40**, 109-117.

Elmitwalli T.A(2000).Anaerobic treatment of domestic sewage at low temperature. Ph.D. thesis ,Wagningen University-the Netherlands.

Fabbro L. and Gregor G. (2003). Blue green algae, general information. The State of Queensland Department of Natural Resources and Mines. *Natural Resources Science* 03092.

Ferrara R.A. and Harleman D.R.F. (1980). Dynamic nutrient cycle model for waste stabilisation ponds. *J.Env. Engrg. Div., ASCE,* **106**(1), 37-53.

Ferrara R.A. and Avci C.B. (1982). Nitrogen dynamics in waste stabilisation ponds. *J. WPCF.,* **54**, 361-369.

Focht D.D. and Verstraete W. (1977). Biochemical ecology of nitrification and denitrification. Advances in Microbial Ecology. **1**, 135-198.

Ghosh Chirshree (1994). An evaluation of duckweed (Lemnaceae) as a candidate for domestic wastewater treatment in cold climate. Department of environmental technology, Agricultural University of Wageningen, The Netherlands.

Gijzen H.J. and Mulder A. (2001). The nitrogen cycle out of balance. *Water*21, August 2001, pp. 38-40.

Horne A.J. and Goldman C.R. (1994). Limnology. McGraw-Hill, New York. Second Edition, 627 pp.

Iqbal S. (1999). Duckweed aquaculture: potentials, possibilities, and limitations for the combined wastewater treatment and animal feed production in developing countries. *EAWAG and SANDEC*., Duebendorf, Switzerland.

Javed Q. (1995). Effect of depth and mixing on performance of duckweed-based system using domestix wastewater. MSc. Thesis, International Institute of infrastructural, Hydraulic, and Environmental Engineering, The Netherlands.

Journey W.K. and Scott C.M. (1996). Anaerobic enhanced treatment of wastewater and option for further treatment.

Jules B., Van Lier, Zeeman G. and Lettinga G. (2001). The role of anaerobic treatment at community-and house-on site treatment of domestic wastewater. Effect of different collection and transport systems. *International Symposium on Low Cost Wastewater Treatment and Reuse*., Cairo, Feb. 2001.

Kawabata Z., Tatsukawa R. and Sato K. (1986). Growth of duckweed and nutrient removal in a paddy field irrigated with sewage effluent. *Journal Environmental Studies.* **27**, 277-285.

King D.L. (1978). The role of ponds in land treatment of wastewater. Proc. *International Symposium on Land Treatment of Wastewater*, Hanover, N.H. 191.

Koning A., Pearson H.P. and Silva S.A. (1987). Ammonia toxicity to algal growth in waste stabilization ponds. *J. Wat. Sci. Tech.,* **19**,115-122.

Körner S., Lyatuu G. B. and Vermaat J. E. (1998). The influence of Lemna gibba L. on the degradation of organic material in duckweed-covered domestic wastewater. *Wat. Res.,* **32**(10): 3092-3098.

Kvet J., Rejmankova E. and Rejmanek M. (1979). High aquatic plants and biological waste water treatment. The outline of possibilities (In Czech). Proceedings of the *Aktiv Jihoceskych Vodohospodaru Conference.* pp 9.

Landolt E. (1986). The family of Lemnaceae - a monographic study. Veröffentlichungen des Geobotanischen Institutes der ETH Zürich, Band **71**: 566 pp.

Lettinga G., Rebac S., Parshina S., Noezhvin Kova A., Lier J., and Stams A. (1999). High rate anaerobic treatment of wastewater at low temperature. Biotechnology and Bioengineering 80 (5): 499-506.

Mahbur Rahman Md. (1994). Performance characteristics and nutrients budget of the duckweed-based stabilization pond system at Mirzapur, Bangladish. MSc. Thesis, EE 166, International Institute of infrastructural, Hydraulic, and Environmental Engineering.

Mahmoud, N. (2002). Anaerobic pre-treatment of sewage under low temperature (15 ºC) conditions in an integrated UASB-Digester system. Ph.D. thesis, Wageningen University, Wageningen, The Netherlands.

Mandi L. (1994). Marrakesh wastewater purification experiment using vascular aquatic plants Eichhornia crassipes and Lemna gibba. *Wat. Sci. Tech.,* **29**, 283-287.

Mara D.D., Alabaster G.P., Pearson H.W. and Mills S.W. (1992). Waste stabilisation ponds: a design manual for eastern Africa. Lagoon Technology International, Leeds.

Mara D. and Pearson H. (1998) Design manual for waste stabilization ponds in Mediterranean countries. Lagoon Technology International, Leeds, England.

Metcalf and Eddy (1991). Wastewater engineering. Treatment, disposal and reuse. Third edition, McGraw – Hill. New York.

Middlebrooks E. J. (1995). Upgrading pond effluents: an overview. *Wat. Sci. Tech.* **31**, 353-368.

Moorhead K.K. and Reddy K.R. (1988). Oxygen transport through selected aquatic macrophytes. *J. Environ. Qual.,* **17**, 138-142.

MOPIC (1998) Regional Plan for the West Bank Governorates; Water and Wastewater Existing situation, Ministry of Planning and International Cooperation, Gaza, Palestine.

Moreno D. (1990). A tracer study of the hydraulics of facultative stabilisation ponds. *Wat. Res*., **24**, 1025-1030.

Mustafa J. Y. (1996) Onsite Wastewater Treatment: Disposal and reuse, M. Sc. Thesis, IHE Delft, The Netherlands.

NERC (1978). A beginners Guide to fresh water algae. Natural Environmental Research Council. Cambridge.

Nielsen L.P. (1992). Denitrification in sediment determined from nitrogen isotope pairing. *FEMS Microbiol. Ecol.,* **86.** 357-362.

Oron, G., Wildschut L.R. and Porath D. (1985). Waste water recycling by duckweed for protein production and effluent renovation. *Wat. Sci. Tech.*,**17**, 803-817.

Oron, G. De-Vegt, A. and Porath, D. (1986). Wastewater Treatment and Renovation by different duckweed species. *J.ASCE*.

Oron, G. and Porath, D. (1987), Wastewater recycling by duckweed for protein production and effluent renovation. L. *Wat. Sci. Tech..*, **17** (4-5), 803-817.

Oron, G. De Vegt Andre and Porath Dan (1988). Nitrogen removal and concersion by duckweed growth on wastewater. *J. Wat. Res.*. **22**, 197-184.

Pano A. and Middlebrooks E.J. (1982). Ammonia nitrogen removal in facultative wastewater stabilization ponds. *J. WPCF*., **54**, 344-351.

Pearson H.W., Mara D.D., Mills S.W. and Smallman D.J. (1987). Physico-chemical parameters influencing faecal bacterial survival in waste stabilization ponds. Wat. *Sci. Tech.,* **19**, 145-152.

Pearson H.W., Mara D.D. and Arridge H.A. (1995). The influence of pond geometry and configuration on facultative and maturation waste stabilisation pond performance and efficiency. *Wat. Sci. Tech.,* **31**, 129-139.

PHG (1999) Design Report: small diameter collection UASB/facultative pond treatment, sludge treatment effluent irrigation scheme. Palestinian Hydrology Group: Sanitation and Environment Program.

Polprasert C., Dissanayake M.G. and Tanh N.C. (1983). Bacterial die-off kinetics in waste stabilisation ponds. *J. WPCF*., **55**(3), 285-296.

PRISM (1992), The Shobuj Shona village enterprise project-progress report. The PRISM group, Dhaka.

Reed Sherwood C., Middlebrooks E. Joe and Crites Ronald W. (1988), Natural systems for wastewater management and treatment. McGraw-Hill Book co. New York, N.Y.

Reed S.C., Crites R.W. and Middlebrooks E.J. (1995). Natural systems for management and treatment. McGraw - Hill. New York.

Samhan S. Pathogen removal efficiency of an integrated UASB-pond system treating domestic wastewater. MSc thesis at Birzeit University-Palestine.

Saqqar M. M. and Pescod M. B. (1992). Modelling coliform reduction in wastewater stabilization ponds. *Wat. Sci. Tech.* **26**, 1667-1677.

Sculthrope C.D. (1967). The biology of aquaculture vascular plants. Edward Arnold, London.

Silva S. A. (1982). On the treatment of domestic sewage in waste stabilization ponds in N.E. Brazil. PhD thesis, University of Dundee, UK.

Silva S.A., de Oliveira R., Soares J., Mara D.D. and Pearson H.W. (1995). Nitrogen removal in pond systems with different configurations and geometries. *Wat. Sci. Tech.,* **31**(12), 321-330.

Skillicorn P., Spira W. and Jouney W. (1993). Duckweed Aquaculture, a new aquatic farming system for developing countries. The World Bank, Washington, USA.

Smith and Moelyowati .M.D and Moelyowati I. (1999). Duckweed based wastewater treatment (DWWT): design guidelines for hot climates. Water and Engineering Development Centre (WEDC), Department of Civil Engineering, Loughborough University, Leicestershire LE11 3TU, UK.

Soares J., Silva S.A., de Oliveira R., Araujo A.L.C., Mara D.D. and Pearson H.W. (1996). Ammonia removal in a pilot-scale WSP complex in Northeast Brazil. *Wat. Sci. Tech.*, **33**, 165-171.

Soler A., Saez, J., Lorens, M., Martinez, I., Torella, F. and Berna, L. (1991). Changes in physicochemical parameters and photosynthetic microorganisms in a deep wastewater selfdepuration lagoong. *Wat. Res.,* **25** (6), 689-695.

Taylor T. and Bishop P. (1989). Distribution and Role of Bacterial Nitrifying Population in Nitrogen Removal in Aquatic Treatment System. *Wat. Res.*, **23,** 947-955.

Underhill S.G. and Prosser J.I. (1987). Surface attachment of nitrifying bacteria and their inhibition by potassium ethylxanthate. *Microbial Ecology*., **14**, 129-139.

USEPA. 1992. Guidelines for Water Reuse. Process Design Manual, Report EPA-625/r-92-004, USEPA, Cincinnati, Ohio.

Van der Steen P., Brenner A. and Oron G. (1998). An integrated duckweed and algae pond system for nitrogen removal and renovation. *Wat. Sci. Tech.,* **38**, 335-343.

Van Luijn F. (1997). Nitrogen removal by denitrification in the sediments of a shallow lake. Ph.D. Thesis. University of Wageningen, Wageningen, The Netherlands.

Verhagen F.J.M. and Laanbroek H.J. (1991). Competition for ammonium between nitrifying and heterotrophic bacteria in dual energy-limited chemostats. *Appl. Environ. Microbiol.* **57**, 3255-3263.

Vermaat J. E. and Hanif M.K. (1998). Performance of common duckweed species (Lemnaceae) and the Waterfern Azolla Filiculoides on different types of wastewater. *Wat. Res.,* **32**, 2569-2576.

Verstraete W. and Alexander M. (1973). Heterotrophic nitrification in samples of natural ecosystems. *Environ. Sci. Technol.* **7**, 39-42.

Wang W. (1991). Ammonia toxicity to macrophytes (Common duckweed and rice) using static and renewal methods. *Environm. Toxic. Chem.,* **10**, 1173-1178.

Wehner J.F. and Wilhelm R.H. (1956). Boundary conditions of flow reactor. *Chem. Eng. Sci.,* **6**, 89-93.

WHO (1989). Health guidelines for the use of wastewater in agriculture and aquaculture. Technical report series 778, World Health Organization, Geneva, Switzerland.

Wildschut l.r. (1983). Introductory study on the performance of Lemanaceae on fresh municipal wastewater with emphasis on the growth of the Lemnaceae, the ammonium removal from the wastewater. Wageningen : L.H., 84. Doktoraal verslagen serie. Vakgroep waterzuivering. Landbouuwhogeschool;1984, no 3.

Young J.C. and McCarty P.L. (1967). The anaerobic filter for wastewater treatment. In: Proc. 2nd Purdue Industrial Waste Conference, May 1967. Ann Arbor Science Publishing Inc., Ann Arbor MI. 559-574.

Zimmo O. R., Al-Sa'ed R., van der Steen P. and Gijzen H. J. (2000). Nitrogen transformation and treatment efficiency in algae-based and duckweed-based stabilisation ponds. Conference proceeding "*Managing Water and Waste in the New Millennium*".

Zimmo, O.R., Al-Sa`ed, R. M., van der Steen, N. P. and Gijzen, H. J. (2002). Process performance assessment of algae-based and duckweed-based wastewater treatment systems. *Wat. Sci. Tech.,* **45**, 91-101.

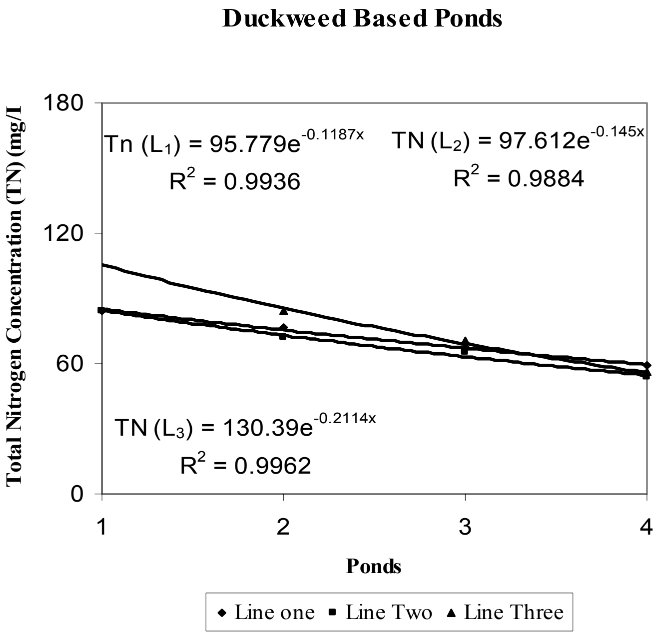
Zimmo, O.R. (2003), Nitrogen Transformations and Removal Mechanisms in Algal and Duckweed Waste Stabilization Ponds, Ph.D. Thesis. University of Wageningen, Wageningen, The Netherlands.

Zirschky J. and Reed S. C. (1988). The use of duckweed for wastewater treatment. *J. WPCF.,* **60**, 1253-1258.

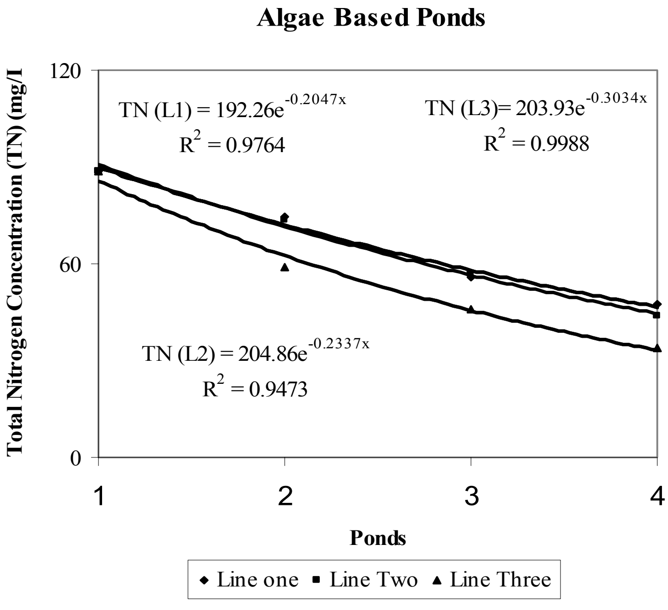
Zuberer D.A. (1982). Nitrogen fixation (acetylene reduction) associated with duckweed (Lemnaceae) mats. *Appl. Environ. Microbiol*., **43**, 823-828.

Appendices

A.1 **Removal curves for different constituents from the pilot plant system**

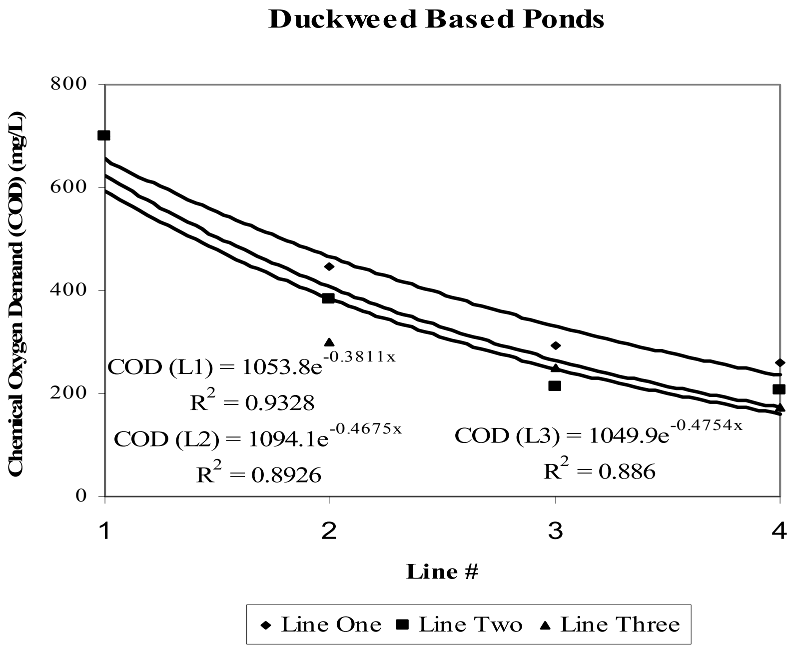


(a)

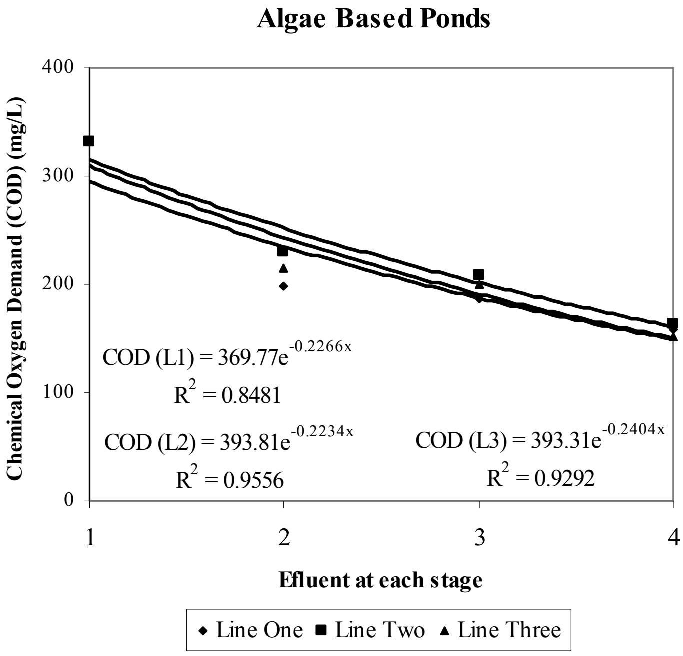


(b)

Fig. A.1. Removal curve of TN for the three lines (90, 60, and 30 cm depth) in DBPs (a), and ABPs (b). The figure shows also the best-fit exponential curve and equation for each line with the correlation coefficient between actual points and the curve. Points 1, 2, 3 and 4 (x-axis) represent influent, first, second and third ponds, respectively.

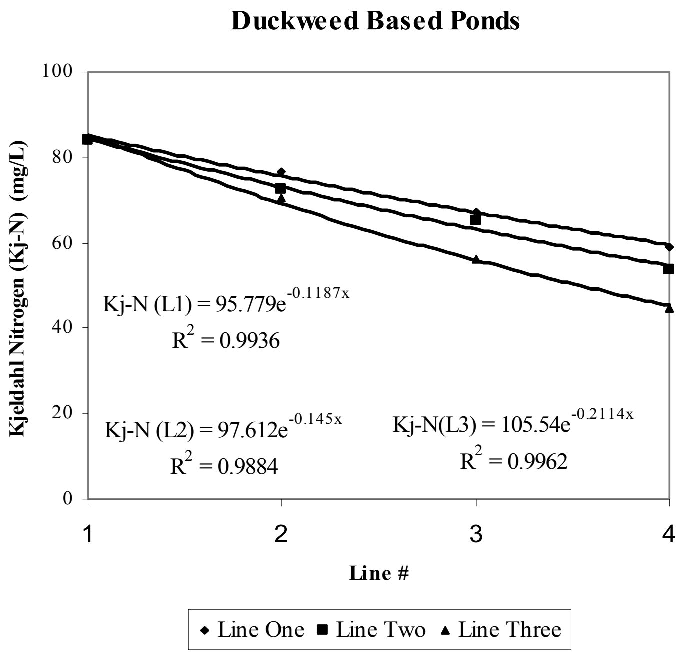


(a)

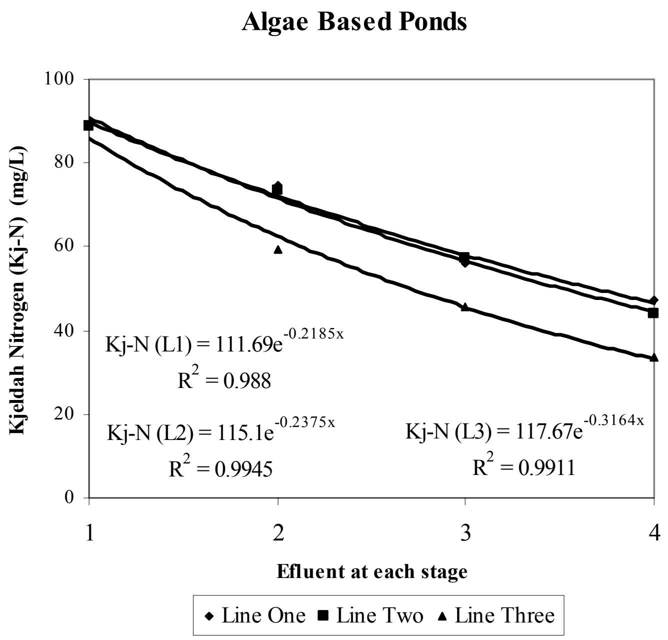


(b)

Fig. A.2 Removal curve of COD for the three lines (90, 60, and 30 cm depth ponds) in DBPs (a), and ABPs (b). The figure shows also the best-fit exponential curve and equation for each line with the correlation coefficient between actual points and the curve. Points 1, 2, 3 and 4 (x-axis) represent influent, first, second and third ponds, respectively.

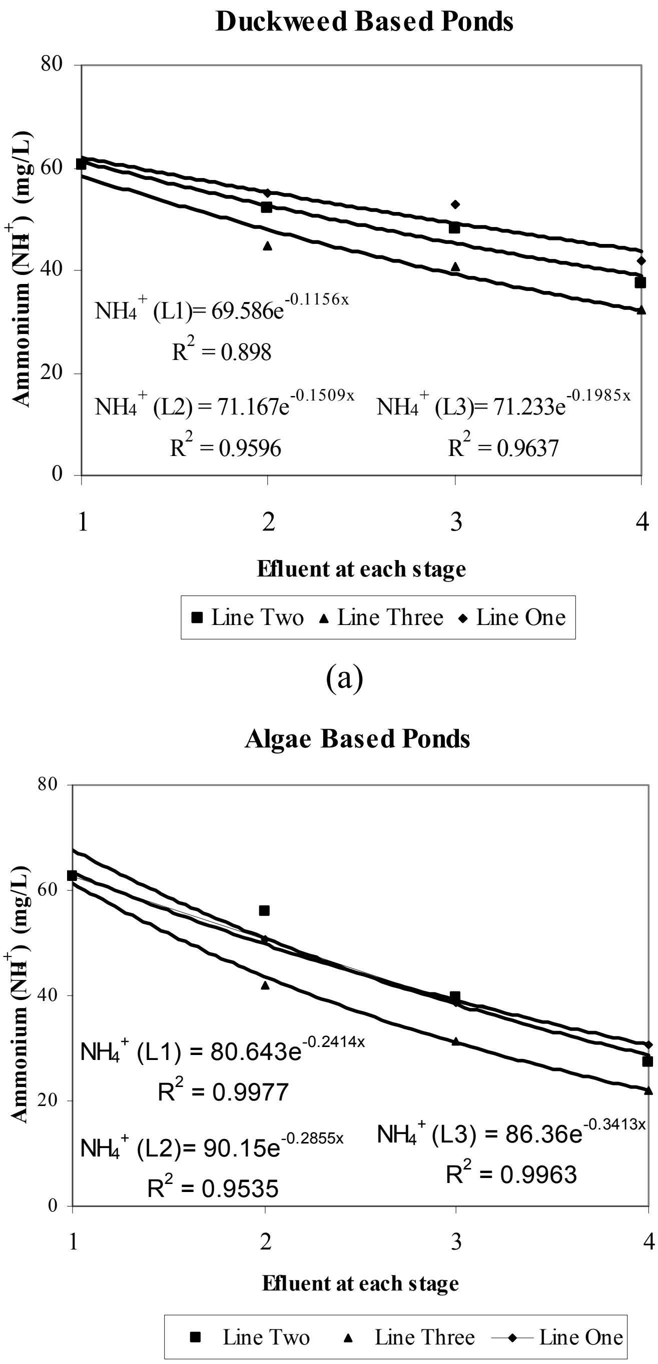


(a)



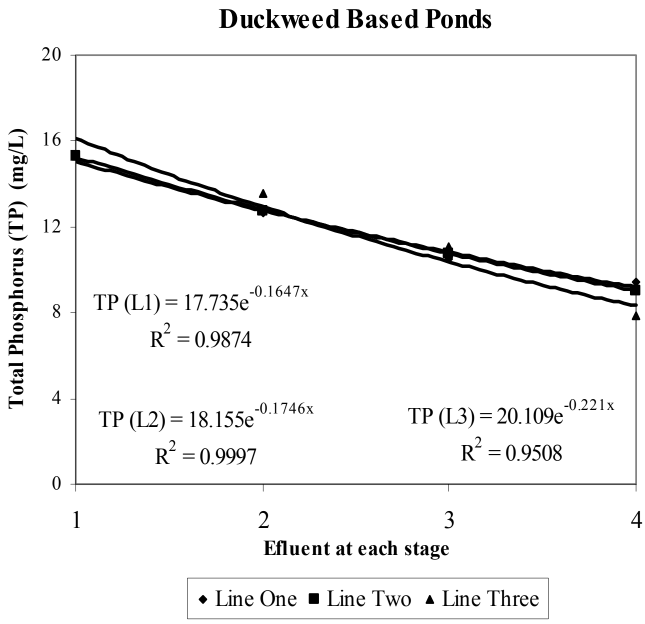
(b)

Fig. A.3. Removal curve of TKN from the three lines (90, 60, and 30 cm depth ponds) in DBPs (a), and ABPs (b). The figure shows also the best-fit exponential curve and equation for each line with the correlation coefficient between actual points and the curve. Points 1, 2, 3 and 4 (x-axis) represent influent, first, second and third ponds, respectively.

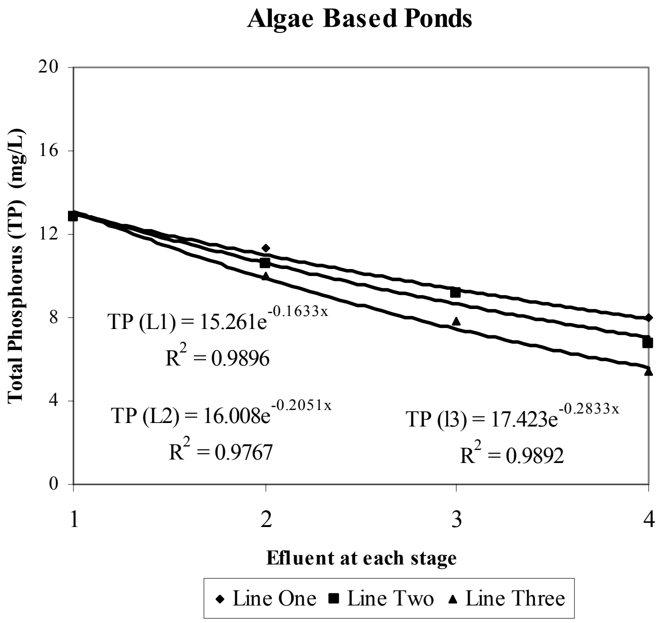


(b)

Fig. A.4. Removal curve of NH_4^+ from the three lines (90, 60, and 30 cm depth) in DBPs (a), and ABPs (b). The figure shows also the best-fit exponential curve and equation for each line with the correlation coefficient between actual points and the curve. Points 1, 2, 3 and 4 (x-axis) represent influent, first, second and third ponds, respectively.



(a)



(b)

Fig. A5. Removal curve of TP for the three lines (90, 60, and 30 cm depth) in DBPs (a), and ABPs (b). The figure shows also the best-fit exponential curve and equation for each line with the correlation coefficient between actual points and the curve. Points 1, 2, 3 and 4 (x-axis) represent influent, first, second and third ponds, respectively.

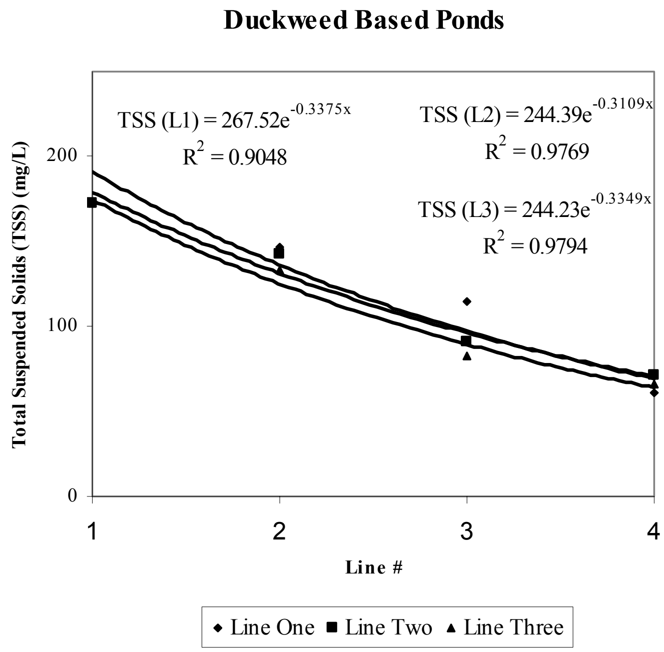


Fig. A.6. Removal curve of TS from the three lines (90, 60, and 30 cm depth) in ABPs. The figure shows also the best-fit exponential curve and equation for each line with the correlation coefficient between actual points and the curve. Points 1, 2, 3 and 4 (x-axis) represent influent, first, second and third ponds, respectively.

Table A.1. Research results for the concentration, number of samples (n), range and removal efficiencies for different parameters in the influent and effluent of the DBPs.

		pH	DO	TSS	TDS	COD	TP	NH_4^+	TKN
Infl.	**n**	14	14	12	4	14	12	14	11
	Range	7.2-7.6	0.0-0.01	104-220	795-991	513-866	12.8-16.6	48.4-87.8	50.3-101.0
	Avg.	7.3 (0.1)	0.0 (0.0)	172 (5.9)	973 (43)	701 (241)	15.3 (0.5)	60.6 (1.9)	84.1 (5.7)
D_1L_1	**n**	14	14	12	4	14	12	14	11
	Range	7.2-7.5	0.02-0.7	66.2-153	811-971	244-555	11-16.5	32.4-82.4	48.4-116.1
	Avg.	7.4 (0.1)	0.06 (0.02)	146 (13)	913 (35)	447 (124)	12.6 (0.3)	55.2 (2.2)	76.8 (6.6)
	%	--	--	14.5 (9.4)	13.6 (2.2)	41.2 (15.9)	17.6 (2.0)	8.8 (1.7)	8.6 (2.6)
D_2L_1	**n**	14	14	12	4	14	12	14	11
	Range	7.2-7.6	0.4-1.81	72-135	888-967	229-410	6.5-14.7	40.6-76.3	37.9-92.4
	Avg.	7.4 (0.2)	0.5 (0.19)	114 (30)	943 (8)	295 (30)	10.5 (0.4)	52.8 (2.7)	67.3 (6.6)
	%	--	--	33.1 (18.9)	6.2 (0.8)	60.8 (8.4)	31.1 (3.3)	12.9 (2.8)	19.7 (5.6)
D_3L_1	**n**	14	14	12	4	14	12	14	11
	Range	7.2-7.6	**0.7-2.34**	40-80	737-848	195-441	5.4-14.0	37.4-61.6	30.5-80.0
	Avg.	7.4 (0.1)	1.19 (0.44)	61 (19)	819 (2)	258 (45)	9.4 (0.5)	41.8 (0.9)	59.1 (8.2)
	%	--	--	64.4 (11.8)	3.1 (3.2)	66.0 (6.9)	38.5 (4.0)	30.9 (1.6)	29.4 (6.8)
D_1L_2	**n**	14	14	12	4	14	12	14	11
	Range	7.1-7.7	0.11-0.96	79.8--156	817-840	251-458	10.7-16.9	48.1-84.6	47.8-81.2
	Avg.	7.5 (0.1)	0.15 (0.07)	143 (13)	826 (1)	382 (76)	12.7 (0.3)	51.9 (3.0)	72.5 (6.8)
	%	--	--	16.7 (10)	15.8 (3.4)	49.3 (13.1)	16.7 (3.4)	14.3 (3.3)	13.7 (3.7)
D_2L_2	**n**	14	14	12	4	14	12	14	11
	Range	7.4-7.7	0.07-1.36	56-122	825-872	179-389	8.5-15.0	38.6-66.4	47.6-73.7
	Avg.	7.5 (0.3)	0.16 (0.03)	91 (16)	852 (27)	213 (44)	10.7 (0.4)	47.9 (4.0)	65.1 (3.7)
	%	--	--	46.6 (10.9)	15.1 (3.6)	70.5 (10.6)	29.7 (3.7)	20.9 (5.4)	22.3 (3.0)
D_3L_2	**n**	14	14	12	4	14	12	14	11
	Range	7.4-7.8	**0.12-2.93**	54-85	813-871	152-302	10.0-14.2	30.1-64.8	36.9-64.4
	Avg.	7.6 (0.1)	1.36 (0.48)	71 (18)	829 (15)	205 (83)	9.0 (0.2)	37.6 (1.7)	53.7 (4.5)
	%	--	--	58.6 (11.2)	12.4 (0.1)	72.4 (11.6)	40.8 (3)	37.9 (1.7)	35.7 (5.2)
D_1L_3	**n**	14	14	12	4	14	12	14	11
	Range	7.4-7.8	0.08-0.57	88-165	718-839	270-410	12.2-15.2	30.7-73.8	49.3-79.6
	Avg.	7.7 (0.1)	0.15 (0.06)	133 (22)	795 (33)	300 (24)	13.5 (0.6)	44.6 (1.0)	70.6 (4.9)
	%	--	--	30.1 (4.2)	14.8 (1.7)	59.0 (12.9)	11.7 (1.5)	26.3 (1.1)	15.7 (4.1)
D_2L_3	**n**	14	14	12	4	14	12	14	11
	Range	7.5-7.9	0.14-2.13	69-114	771-918	178-382	10.5-15.1	36.6-65.1	34.1-68.1
	Avg.	7.7 (0.1)	0.22 (0.1)	83 (31)	851(29)	249 (58)	11.1 (0.7)	40.6 (0.5)	56.3 (6.3)
	%	--	--	51.6 (18.9)	18.2 (8.4)	67.8 (5.4)	27.5 (4.3)	33 (2.7)	32.6 (6.7)
D_3L_3	**n**	14	14	12	4	14	12	14	11
	Range	7.6-7.9	**0.54-3.16**	49-94	830-871	113-355	5.2-13.9	26.4-55.3	22.7-61.6
	Avg.	7.8 (0.1)	1.69 (0.78)	66 (23)	841 (42)	175 (56)	7.8 (1.5)	32.3 (2.5)	44.8 (6.2)
	%	--	--	61.2 (13.8)	12.6 (0.3)	78.0 (2.8)	48.5 (9.2)	46.6 (5.2)	44.5 (6.3)

Inf. = influent; n = number of samples; avg. = average.

All parameters are measured in (mg/L) except for pH (dimensionless).

Table A.1. Research results for the concentration, number of samples (n), range and removal efficiencies for different parameters in the influent and effluent of the ABPs.

		pH	DO	TDS	TSS	TDS	COD	TP	NH_4^+	TKN
influent	**n**	10	10	10	10	10	10	10	10	10
	Range	7.1-7.5	0.0-0.01	998-1516	82-115	818-866	276-608	8.5-14.7	48.6-66.5	80.1-109.1
	Avg.	7.3 (0.1)	0 (0.0)	1143 (106)	95 (5.8)	849 (8.8)	331 (69)	12.8 (0.7)	62.7 (5.7)	88.8 (1.8)
A_1L_1	**n**	10	10	10	10	10	10	10	10	10
	Range	7.5-7.9	0.4-2.64	1010-1300	76-208	751-888	159-434	6.9-11.9	38.2-58.9	62.1-88.5
	Avg.	7.8 (0.1)	1.3 (0.3)	1051 (35)	148 (35)	801 (24)	198 (43)	11.3 (0.9)	50.8 (7.4)	74.7 (3.3)
	%	--	--	1.5 (4.2)	-59.4 (29)	5.7 (1.7)	39.7 (2.2)	11.5 (3.1)	19 (1.8)	13.7 (4.8)
A_2L_1	**n**	10	10	10	10	10	10	10	10	8
	Range	7.7-8.3	1.68-4.11	979-1258	80-174	762-846	145-414	6.0-10.9	29.9-47.9	43.2-67.5
	Avg.	8.0 (0.1)	2.7 (0.9)	1030 (52)	162 (47)	769 (8)	186 (44)	9.2 (0.7)	38.6 (5.3)	55.8 (3.6)
	%	--	--	4.3 (4.4)	-68.8 (23.2)	9.4 (1.3)	43.6 (3.3)	28.3 (1.7)	38.2 (3)	35.7 (1.4)
A_3L_1	**n**	10	10	10	10	10	10	10	10	10
	Range	7.7-8.35	2.1-4.97	1040-1168	108-244	495-935	123-328	6.9-10.2	23.0-45.7	40.3-77.7
	Avg.	8.2 (0.1)	3.7 (1.2)	1085 (40)	134 (17)	774 (6)	159 (38)	8.0 (1.0)	30.7 (5.8)	47.2 (2.7)
	%	--	--	2.6 (7.1)	-41.5 (8.9)	8.9 (1)	51.6 (3.2)	37.6 (6.4)	51.2 (1.9)	45.4 (3.1)
A_1L_2	**n**	10	10	10	10	10	10	10	10	10
	Range	7.6-8.0	0.33-1.74	1018-1444	114-242	785-829	180-534	6.8-12.75	42.7-65.7	57.9-93.5
	Avg.	7.8 (0.1)	1.5 (0.8)	1057 (45)	128 (17)	797 (10)	230 (54)	10.6 (0.5)	56.1(8.3)	73.5 (3.3)
	%	--	--	0.7 (3.9)	-35.1 (8.8)	6.2 (0.7)	30.6 (3.7)	16.9 (1.5)	10.6 (0.6)	15.3 (1.3)
A_2L_2	**n**	10	10	10	10	10	10	10	10	8
	Range	7.8-8.3	1.2-4.76	1020-1336	118-180	752-827	146-472	6.6-12.1	37.1-52.9	47.1-56
	Avg.	8.0 (0.1)	2.9 (1.0)	1064 (36)	143 (26)	778 (6)	208 (65)	9.2 (0.9)	39.7 (6.9)	57.1 (6.7)
	%	--	--	1.3 (5.3)	-54.2 (27.2)	8.3 (1.1)	38 (8.3)	28.3 (6.2)	36.8 (2.5)	34 (6.3)
A_3L_2	**n**	10	10	10	10	10	10	10	10	10
	Range	7.7-8.1	1.47-4.96	972-1168	66-112	580-836	134-462	5.1-9.8	25.8-48.3	29.5-67.2
	Avg.	8.0 (0.1)	3.8 (0.4)	1029 (56)	89 (32)	781 (12)	163 (32)	6.8 (1.2)	27.2 (6.1)	43.8 (4.4)
	%	--	--	5.4 (5.6)	7.2 (17.9)	8.1 (1.1)	53.4 (4.3)	46.4 (10.5)	56.9 (2.9)	49.3 (4.3)

A_1L_3	**n**	10	10	10	10	10	10	10	10	10
	Range	7.7-7.9	1.03-2.43	598-1426	54-232	756-839	166-456	6.9-13.0	34.5-53.9	50.4-74.5
	Avg.	7.8 (0.1)	1.7 (0.5)	930 (196)	131 (73)	784 (23)	215 (53)	10 (0.9)	42.0 (7.3)	59.2 (7.6)
	%	--	--	8 (9)	-34.3 (42.1)	7.7 (1.6)	35.2 (4.1)	21.7 (3.4)	33 (4.4)	31.8 (5.3)
A_2L_3	**n**	10	10	10	10	10	10	10	10	10
	Range	7.8-8.2	1.45-3.91	980-1224	118-248	722-854	161-431	5.4-12.6	23.2-44.7	30.5-54.5
	Avg.	8.0 (0.1)	3.1 (0.6)	1034 (117)	136 (28)	765 (17)	200 (42)	7.8 (0.6)	31.4 (5.4)	45.7 (5.5)
	%	--	--	-64.5 (130)	-43.7 (15.4)	9.9 (1)	39.2 (2.0)	39 (1.8)	49.9 (3.6)	47.4 (3.5)
A_3L_3	**n**	10	10	10	10	10	10	10	10	10
	Range	7.8-8.4	2.52-4.98	1080-1266	84-236	714-854	123-329	3.8-8.7	12.0-43.9	21.4-53.8
	Avg.	8.3 (0.0)	4.0 (0.3)	1176 (53)	153 (26)	760 (12)	152 (32)	5.4 (0.7)	22.2 (3.5)	33.7 (6.5)
	%	--	--	-2.5 (8.9)	-64.2 (24.8)	10.4 (1.5)	55.9 (3.7)	57.6 (5.6)	64.5 (2.8)	61.1 (4.5)

Inf. = influent; n = number of samples; avg. = average.
All parameters are measured in (mg/L) except for pH (dimensionless).

Appendix C. Photos of the experimental setup

Photo A.1. Side view for the experimental pilot plant; the three lines of stabilization ponds and the UASB-septic tanks in AWWTP.

Photo A.2. Front view for the experimental pilot plant; the three lines of stabilization ponds and the UASB-septic tanks in AWWTP.

Photo A.3. Connecting container which collect the UASB-septic tank effluent. The photo shows also the hose which convey the effluent to the holding tank (see photo A.4).

Photo A.4. Holding tank, (collection reservoir for the UASBs' effluent). The photo shows also the overflow line from which excess pretreated sewage is conveyed back to the grit chamber.

Photo A.5. Suction point from the holding tank by the pumps to the stabilization ponds. The MASTER FLEX hoses are also shown. The photo shows also the pump feeding the deepest stabilization ponds.

Photo A.6. The baffle which was installed for preventing short circuiting of wastewater and to prevent floating materials from transporting to successive stabilization ponds. The 4" PVC pipe connecting stabilization ponds is also shown

Photo A.7. Galvanized steel holder for the sedimentation cup, two rods were used for the purpose of holding; the first rod (vertical) was used to hold the cup and keep it from turn over and the second one (horizontal) was used connect the vertical holder on the pond body.

Photo A.8. The holder mentioned in A.7, this photo shows particularly the ponds after scavenging the mortared duckweed mat; the whole mat quantity was absolutely removed to prevent decayed duckweed from affecting the readings of different parameters.

Photo A.9. One of duckweed culture, it is a big bucket of dimensions (dia. = 34 cm, L = 40cm). As mentioned elsewhere, this culture was used as a backup source of duckweed and for testing the duckweed survival in diluted effluent of the UASB-septic tank.

Photo A.10. Healthy duckweed mat in the stabilization ponds at the moment of seeding. It is clear from the figure that this cover was sufficiently dense to prevent algal growth which contributes to duckweed death.

Photo A.11. Duckweed mat in the stabilization ponds. It is shown in this photo that this duckweed contains considerable amount of duckweed grains with brownish color that indicates that the duckweed faces stress resulted mainly from wastewater characteristics (strength).

Photo A.12. Mortared duckweed cover; it is clear that the duckweed mat has a brownish color which indicates that the duckweed is no longer active. For this case the whole duckweed mat was absolutely removed and another new duckweed mat was brought from the original cultures (of BZU) in the same day.

المعالجة أوليا، تمت دراسة أثر العمق في البرك المعتمدة على الطحالب. بدأت المرحلة المرحلة الأولى في 2004/5/2 وانتهت في 2004/8/18. كما وتم البدء في دراسة المرحلة الثانية مباشرةأي في 2004/8/18 وتم الإنتهاء منها في 2004/11/1.

جائت معظم فترة الدراسة في فصل الصيف حيث كان معدل درجات الحرارة في فترة الدراسة 24.5 درجة مؤوية وكان معدل درجات حرارة المياه العادمة في الفترة الأولى 23.6 درحة مئوية، بينما كان معدل درجات الحرارة للمياه العادمة في المرحلة الثانية 22.9 درحة مئوية. وكان تركيز الأكسجين الكلي المستهلك كيميائيا 1275 ± 84 مغم\لتر. بينت نتائج الدراسة بأن كفاءة البرك في إزالة كل من الأكسجين الكلي المستهلك كيميائيا و الفوسفور والنيتروجين تناسبت عكسيا مع العمق لهذه البرك. لقد كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 90 سم 62.5 ± 5.7% كما و كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 30 سم 75.4 ± 4.1. أما في البرك المعدة على الطحالب فقد كانت كفاءة البرك المعتمدة على الطحالب ذات العمق 90 سم 51.6 ± 3.2% كما و كانت كفاءة البرك المعتمدة على الطحالب ذات العمق 30 سم 54 ± 1.1%. وكانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 30 سم في أزالة الفوسفور 48.5 ± 9.2% كما و كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 90 سم 38.5 ± 4% أما في البرك المعدة على الطحالب فقد كانت كفاءة البرك المعتمدة على الطحالب ذات العمق 30سم 57.6 ± 5.6% كما و كانت كفاءة البرك المعتمدة على الطحالب ذات العمق 90 سم 37.6 ± 6.4%. أما بالنسبة للمواد العالقة الكلية فقد كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 30 سم في إزالة المواد العلقة الكلية64.4 ±11.8%، كما و كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 30سم 58 ±11.2%. أما في البرك المعدة على الطحالب فقد لوحظ زيادة في تركيز المواد العالقة الكلية وذلك لنمو الطحالب في هذه البرك.

أما بالنسبة لمواد النيتروجين فقد لوحظ أيضا زيادة في الكفاءة في البرك الضحلة مقارنة بالبرك العميقة. لقد لقد كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 90سم في إزالة الأمونيوم 30.9± 1.6% كما و كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 30 سم 46.6± 5.2%. أما في البرك المعدة على الطحالب فقد كانت كفاءة البرك المعتمدة على الطحالب ذات العمق 90سم في إزالة نيتروجين الكلدال 51.2 ± 1.9% كما و كانت كفاءة البرك المعتمدة على الطحالب ذات العمق 30 سم 64.5 ± 2.8%. أمابالنسبة لنيتروجين الكلدال فقد كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 90سم في إزالة الأمونيوم 29.4 ± 6.8% كما و كانت كفاءة البرك المعتمدة على عدس الماء ذات العمق 30 سم 44.5 ± 6.3%. أما في البرك المعدة على الطحالب فقد كانت كفاءة البرك المعتمدة على الطحالب ذات العمق 90 سم في إزالة نيتروجين الكلدال 45.4 ±3.1%، كما و كانت كفاءة البرك المعتمدة على الطحالب ذات العمق 30 سم في إزالة نيتروجين الكلدال 61.1 ±4.5%. رغم أن البرك ذات الإرتفاع المنخفض قد حققت كفاءة أفضل من البرك الأكثر عمقا إلا أن الأخيرة تحتاج إلى مساحة أقل من الأراضي للوصول إلى التراكيز المطلوبة لتكون المياه المعالجة صالحة للري ويعود ذلك لاستيعابها الأكبر للمياه وبالتالي فإن القص في الكفاءة يعوض بالزيادة في السعة فيمكن زيادة الكفاءة لهذه البرك بزيادة المساحة لهذه البرك. وقد بينت النتائج أن البرك العميقة المعتمدة على عدس المياه تحتاج إلى 0.8متر مربع\نسمة لتوافق المواصفات الأردنية للري بالمياه المعالجة.

الخلاصة

تعتبر أنظمة معالجة المياه العادمة المستعملة مثل نظام الحمأة المنشطة مكلفة من حيث رأس المال و التشغيل والصيانة. مما يحول دون استخدام هذه التقنيات في الدول الفقيرة ذات الدخل المحدود، الأمر الذي يزيد من حدة المشاكل البيئية في هذه الدول. وعلى الرغم من ذلك فهناك أنظمة معالجة بديلة تعتبر بديلا مناسبا للأنظمة المذكورة أعلاه من حيث التكاليف والكفاءة، ومن الأمثلة على هذه الأنظمة، نظام غطاء الحمأة اللاهوائي الصاعد إذ يعتبر هذا النظام الأكثر شيوعا بين الأنظمة اللاهوائية وذلك لانخفاض تكاليف البناء والتشغيل والصيانة لهذا النظام وبالإضافة إلى ذلك فإن هذا النظام هو من أقل الأنظمة المخلفة للحمأة التي تحتاج بدورها إلى معالجة قبل طرحها للخارج. وعلى الرغم من هذه المميزات لهذا النظام إلا أنه غير كفؤ في معالجة مركبات النيتروجين والفوسفور، علما بأن مركبات النيتروجين والفوسفور تعتبر من المواد الملوثة للبيئة بشكل خطير إذا تم طرحها دون معالجة، حيث تسبب هذه المواد مشاكل صحية مثل حالات "الطفل الأزرق" كما وتسبب هذه المواد تلوثا شديدا في المياه السطحية مثل النمو الكثيف للطحالب مما يؤدي إلى مضاعفات خطيرة على الحياة المائية في هذه الأحواض المائية، كما وتؤدي إلى تردي نوعية المياه في هذه التجمعات المائية مما يجعلها غير صالحة للشرب أو الري أو لاستعمالات أخرى. لذا فيجب إضافة وحدة معالجة ثانية للمياه الخارجة من النظام اللاهوائي المذكور وذلك لجعل هذه المياه صالحة لاستعمالات مختلفة أهمها الزراعة، ومن الأنظمة قليلة التكلفة التي تصلح لهذه المهمة هو نظام البرك المعتمدة على الطحالب والتي بدورها قادرة على تنقية هذه المياه لتجعلها صالحة للزراعة. إلا أن هذه الأنظمة تحتاج الى مساحة كبيرة للقيام بهذا الدور.

تعتبر البرك المعتمدة على عدس الماء في جوهرها هي نظام منبثق ومحسن عن نظام البرك المعتمدة على الطحالب، وبالإضافة الى ذلك فإن هذا النظام يعتبر من الأنظمة المستدامة وذلك لأن نبات عدس الماء غني بالبروتينات المفيدة فيستخدم هذا النبات كعلف للدواجن كما ويستخدم كسماد وذلك لغناه بالنيتروجين إذ ينمو عدس الماء بشكل كثيف في هذه البرك متغذيا على المواد العضوية ومركبات النيتروجين والفوسفر الموجودة في الماء، وبالتالي فإن بيع هذا النبات هو مصدر دخل يمكن أن يغطي جزءا كبيرا من تكاليف التشغيل والصيانة، إلا أن هذا النظام أيضا يحتاج إلى مساحات كبيرة لكي يعمل بشكل جيد حيث يحتاج إلى 2-4م^2\نسمة. وبالتالي فإن تقليل المساحة المطلوبة لهذين النظامين بجعلهما من الأنظمة الواعدة. إن المساحة المطلوبة في هذه الأنظمة يمكن تقليلها إذا تم زيادة عمق هذه البرك على أن يبقى الحجم ثابتا، ولكن لا يوجد دراسات كافية تشرح أثر زيادة العمق على كفاءة النظام في أزالة الملوثات عامة والنيتروجين بشكل خاص.

يهدف بحث هذه الرسالة إلى دراسة أثر زيادة العمق على كفاءة البرك المعتمدة علىعدس الماء. وبهدف هذه الدراسة تم أنشاء محطة دراسية في محطة البيرة لمعالجة المياه العادمة التي تقع شمال شرق مدينة القدس. تكونت المحطة الدراسية من نظام غطاء الحمأة اللاهوائي الصاعد يتلوه ثلاثة صفوف من البرك في كل صف ثلاثة برك حيث كان عمق البرك في الصف الأول 90 سم وفي الصف الثاني 60 سم أما في الصف الثالث فكان العمق 30 سم. تم التحكم بمعدل ضخ المياه الخارجة من نظام المعالجة اللاهوائي بحيث يكون إجمالي زمن المكوث للمياه المعالجة أوليا في الصفوف الثلاثة المذكورة 28 يوماً. تكونت هذه الدراسة من مرحلتين، المرحلة الأولى هدفت إلى دراسة اثر العمق في البرك المعتمدة على عدس الماء، ولكن عندما تعذر نمو هذا النبات في المياه العادمة

كلية الدراسات العليا

معهد علوم وتكنولوجيا المياه

أثر العمق على إزالة النيتروجين في البرك المعتمدة على عدس الماء والطحالب للمعالجة الثانوية للمياه الخارجة من غطاء الحمأة اللاهوائي الصاعد

رسالة ماجستير مقدمة من:

أشرف عبدالله إسعيد

الرقم الجامعي: 1025081

إشراف

د. عمر زمو

قدمت هذه الرسالة استكمالا لمتطلبات درجة الماجستير في هندسة المياه من كلية الدراسات العليا في جامعة بيرزيت - فلسطين

نيسان - 2005