Efficient Management of Wastewater

Its Treatment and Reuse in Water-Scarce Countries

Ismail Al Baz • Ralf Otterpohl • Claudia Wendland
Editors

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Preface

Water in the MEDA region is a crucial issue, with regard to the availability of renewable water resources in the MEDA countries most will face even more serious problems in the management of their limited water resources in the near future. This will require a lot of efforts to be made for more efficient management of water, in order to secure the economic and social development of the coming generations.

According to the FAO (2006) the average of renewable water resources in the MENA region is below the limit of 1000 CM per Capita and Year, for Egypt for example is this 794 CM, for Algeria und Tunisia 481 CM, for Jordan 180, Yemen 234, and Palestine 100 which are far below the limit of 500 CM that classify these countries as the most water stressed countries worldwide.

The alarming aspect is the fact that the limited renewable available water resources development have been decreasing in the last thirty years, between 1974 – 2000 we had 66% decrease for Jordan and 64 % for Yemen, due to the increasing population growth and the increase of water demands for agriculture, industrial and domestic use. These figures underline the importance of the topics of this book that shall give help to experts and decision makers to over come the future water resources problems in the region.

Water reuse plays an important role in water management as the domestic wastewater streams are mostly independent from dry and wet seasons and can be considered as a relatively fixed amount of available resource throughout the year.

If we look at the example of wastewater treatment in the western European countries, where water scarcity is mostly not a major issue, the centralized water borne sanitation systems have contributed to ensuring public health without doubt. Their implementation were a major step in improving living conditions in growing cities about 100 years ago.

However, today we realize that this system is not the best available principle for the next decades or even centuries especially not for water scarce regions. Reasons are:

- Drinking water is wasted for transportation of faeces and urine, a relatively high flow is needed only to keep sewers functional
- Central wastewater treatment is more complex and costly than necessary, esp. high energy costs and is often not where reuse is possible
- Sewerage of centralized systems is very costly in construction, rehabilitation and maintenance

- Nutrients are only partly removed and lost as fertilizer
- Problems of sludge disposal

Within the last decades, many technologies in the field of wastewater management such as systems like different sewerage for wastewater collection and hybrid system, UASB or products by many companies for treatment were developed for special applications and frame conditions that the variety is difficult to overview. However, wastewater treatment is not just a black box that can be easily copied and implemented in any case. As every area, town, village has its own properties and conditions, there is no unique solution possible. Each region, area, town or village needs a tailor made solution for wastewater management. The systems and technologies are developed but must be adapted to the local conditions. Wastewater treatment touches all parts of society that it is very important to consider it from the beginning in regional planning.

Within the regional planning, the following priorities should be considered from the beginning:

1. Wastewater flows can be reduced by demand side management. Efficient usage of water can be achieved such as public awareness for water efficient house installations and water saving toilets and efficient drip irrigation.
2. Rainwater should be harvested where appropriate and possible.
3. To avoid the dilution of pollutants in the wastewater, heavily polluted industrial effluents should be treated and reused separately.
4. When selecting the wastewater system, central, communal or decentral solutions must be considered and compared with dynamic cost comparison.

Based on these considerations, the best option for wastewater collection, treatment and reuse can be evaluated.

The decision behind the publication of this book came after a successfully conducted conference within the activities of the EU funded EMWater project in Jordan in autumn 2006..

The EMWater regional conference has had very good response, with more than 150 participants and 60 papers from 17 different countries from the EU – MEDA region, from Australia and the USA. The best papers presented in the EMWater regional conference are being published in this book.

The EU funded EMWater Project "Efficient Management of Wastewater, its Treatment and Reuse in the MEDA Countries" started in May 2003 with 9 different partners from MEDA and the EU. The EMWater project consortium consists of four EU partners – InWEnt Capacity Building International, Germany, Hamburg University of Technology (TUHH), Germany, Adelphi Research, Germany and the National Agency for New Technology, Energy & Environment (ENEA), Italy – and five Mediterranean partners – YILDIZ Technical University, Turkey, University of Balamand, Lebanon, Lebanese American University, Lebanon, Al al Bayt University, Jordan and Birzeit University, Palestine

The EMWater project is focusing through its different activities on the aspects of wastewater treatment and water reuse.

During the last 4-year EMWater project program that conducted in the region, the following activities have been implemented:-

1. ***Data collection and evaluation of wastewater situation in the target countries: Lebanon, Palestine, Jordan and Turkey.***
2. ***Production of public awareness programs in Arabic, Turkish and English.***
3. ***Conduction of Capacity building programs through local, regional and web based training courses; more than 1300 participants from the region have been trained.***
4. ***Design and construction of 5 pilot plants for demonstration and research purposes***
5. ***Elaboration of EMWater guide for wastewater treatment and water reuse***
6. ***Publication of trainer tool kits for experts and trainer who work in the field of wastewater treatment and water reuse***

Some of the authors of this book are covering the aspects of water and sludge reuse and the positive impact on soil and plants production. The benefit of integrated anaerobic and aerobic wastewater treatment which reduce energy consumption, operation costs and increase treatment efficiency, is recommended by other authors as a sustainable treatment option for the Middle East countries.

Case studies for sustainable sanitation by using constructed wetlands, cost benefit analyses for centralized and decentralized systems to support decision makers, the benefit of using bio membrane reactors technology to get high performance for variable wastewater treatment characteristics, were also included in this book.

Other important topics like social and economic aspects of water reuse, the community participation, culture relation, water value, global climate change and water scarcity, were included in the book too.

The new recently published WHO Guidelines for safe wastewater use in agriculture were evaluated by other authors who give decision makers practical guidance how to apply these new guide lines. The pond system as an efficient, natural and adequate wasterwater technique was also recommended, and finally the aspects of improved wastewater treatment by using constructed wetlands by applying earth worms and alternative plants were highlighted by some other authors.

The authors in this book are well known experts in the field of wastewater treatment and water reuse. Through their papers published in this book, we hope that an essential contribution will be made towards solving the current and future water stress problems in the MEDA region.

As editors, we want the readers to have a look at the various aspects of wastewater management and water reuse, to consider innovative technologies as well as innovative low-tech based on traditional systems of the region and to aim at their tailor-made solution appropriate for the local situation.

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Chapter 1
The 2006 WHO Guidelines for Wastewater and Greywater Use in Agriculture: A Practical Interpretation

Duncan Mara(✉) and Annika Kramer

Abstract The World Health Organization (WHO) published the third edition of its guidelines for the safe use of wastewater, excreta and greywater in agriculture in September 2006. These new guidelines are intended to support the establishment of national standards and regulations. However, it is not straightforward for policymakers or practicing engineers to translate them into numerical values that are easy to implement. This chapter presents a practical interpretation of the main concepts of the new WHO guidelines and provides guidance on how to apply them in national settings.

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1.1. Introduction

The 1989 World Health Organization (WHO) guidelines for the safe use of wastewater in agriculture have long been the standard reference for regulating wastewater reuse. However, subsequent research and expert opinion has stressed the fact that the 1989 guidelines needed to be more easily adaptable to local conditions and should be co-implemented with such other health interventions as hygiene promotion, provision of adequate drinking water and sanitation, and other healthcare measures. The 1989 guidelines have therefore been revised based on new data from epidemiological studies, quantitative microbial risk assessments and other relevant information.

The revised WHO guidelines published in 2006 (WHO, 2006a, 2006b) are essentially a code of good management practices to ensure that, when wastewater is used in agriculture (mainly for irrigating crops, including food crops that are or may be eaten uncooked), it is used safely and with minimal risks to health. To reduce the health risks resulting from human exposure to pathogens in the wastewater, the new guidelines focus on health-based targets, instead of water quality standards, and offer various combinations of risk management options for meeting them.

This is a logical approach since the real question is not how many pathogens (or *E. coli*, fecal coliforms) are permissible in the treated wastewater (this was the approach adopted in the 1989 guidelines), but rather how many pathogens can be ingested, in the case of restricted irrigation (Section 1.2), with wastewater-contaminated soil or, in the case of unrestricted irrigation (Section 1.3), with wastewater-irrigated food, without the resulting infection and disease risks being unacceptably high.

The following sections elaborate on the methodology used in the 2006 WHO guidelines to determine the actual disease risk linked to wastewater irrigation. Moreover, they give numerical values of infection risk related to different wastewater qualities determined through risk simulations. The final section explains how the health-based targets can be adapted to existing public health, socio-economic and environmental circumstances when setting national standards.

1.2. Health-Based Targets in the 2006 WHO Guidelines

The sequence of the approach to human health protection in the 2006 guidelines is as follows:

1. establish the maximum additional disease burden resulting from the use of wastewater for crop irrigation;
2. determine the maximum number of pathogens that could be ingested without exceeding this tolerable disease burden;
3. determine, through realistic human exposure scenarios, the number of pathogens that could be ingested under different irrigation regimes for different crop types;
4. calculate the required reduction of pathogen numbers that needs to be achieved, depending on the initial wastewater quality and the crop type; and
5. select a combination of health-based control measures to achieve this required pathogen reduction.

This approach has been used to develop microbial reduction targets for viral, bacterial and protozoan pathogens. The steps are pursued through a combination of the analytical methods detailed later.

For helminth eggs, this approach cannot be used as data on the resulting health risks are not available. Instead, limit values were determined from epidemiological studies. The recommendation in the guidelines is that wastewater used in agriculture should contain ≤1 human intestinal nematode egg per liter. The helminths referred to here are the human intestinal nematodes: *Ascaris lumbricoides* (the human roundworm), *Trichuris trichiura* (the human whipworm) and *Ancylostoma duodenale* and *Necator americanus* (the human hookworms). (Details of the diseases they cause and their life cycles are given in Feachem et al., 1983.)

This is the same as was recommended in the 1989 guidelines (WHO, 1989), but with two important differences: (i) when children under the age of 15 are exposed (by working or playing in wastewater-irrigated fields) additional measures are needed, such as regular deworming (by their parents or at school); and (ii) the ≤1 egg per liter recommendation does not apply in the case of drip irrigation of high-growing crops (such as tomatoes); in this case, no recommendation is necessary.

1.2.1. Tolerable Additional Disease Burden and Disease and Infection Risks

The basis of human health protection in the 2006 guidelines is that the additional disease burden arising from working in wastewater-irrigated fields or consuming wastewater-irrigated crops should not exceed 10^{-6} disability-adjusted life year (DALY) loss per person per year (pppy; see Box 1.1 for a brief description of DALYs). This level of health protection was used by WHO in its 2004 guidelines on drinking water quality (WHO, 2004). Thus, the health risks resulting from wastewater use in agriculture are the same as those from drinking fully treated drinking water, and this is basically what consumers want as they expect the food they eat to be as safe as the water they drink.

Three “index” pathogens were selected: rotavirus (the most common viral cause of diarrheal disease worldwide), *Campylobacter* (the most common bacterial cause of diarrheal disease worldwide) and *Cryptosporidium* (one of the three most common protozoan causes of diarrheal disease worldwide, the other two being *Giardia* and *Entamoeba*).

To determine the maximum tolerable pathogen exposure resulting from working in wastewater-irrigated fields or consuming wastewater-irrigated crops, the tolerable additional disease burden of 10^{-6} DALY loss pppy is first “translated” into tolerable disease and infection risks as follows:

$$\text{Tolerable disease risk pppy} = \frac{\text{Tolerable DALY loss pppy}}{\text{DALY loss per case of disease}}$$

$$\text{Tolerable infection risk pppy} = \frac{\text{Tolerable disease risk pppy}}{\text{Disease/infection ratio}}$$

Box 1.1 Disability-Adjusted Life Years (DALYs)

DALYs are a measure of the health of a population or burden of disease due to a specific disease or risk factor. DALYs attempt to measure the time lost because of disability or death from a disease compared with a long life free of disability in the absence of the disease. DALYs are calculated by adding the years of life lost to premature death (YLL) to the years lived with a disability (YLD). YLL are calculated from age-specific mortality rates and the standard life expectancies of a given population. YLD are calculated from the number of cases multiplied by the average duration of the disease and a severity factor ranging from 1 (death) to 0 (perfect health) based on the disease (e.g., watery diarrhea has a severity factor from 0.09 to 0.12 depending on the age group; Murray and Lopez, 1996; Prüss and Havelaar, 2001). Thus, 1 DALY loss is equivalent to 1 year of illness or 1 YLL.

DALYs are an important tool for comparing health outcomes because they account for not only acute health effects but also for delayed and chronic effects, including morbidity and mortality (Bartram et al., 2001). Thus, when risk is described in DALYs, different health outcomes (e.g., cancer vs. giardiasis) can be compared and risk management decisions prioritized. Thus, the DALY loss per case of campylobacteriosis in Table 1.1 includes the appropriate allowance for the occurrence of Guillain-Barré syndrome (an inflammatory disorder of the peripheral nerves that may lead to paralysis and that occurs in around 1 in 1,000 cases of campylobacteriosis).

What does 10^{-6} DALY loss pppy mean?

The tolerable additional disease burden of 10^{-6} DALY loss pppy adopted in the guidelines means that a city of 1 million people collectively suffers the loss of 1 DALY per year. The highest DALY loss per case of diarrheal disease in Table 1.1 is 2.6×10^{-2}, for rotavirus disease in developing countries. Assuming that the recommendations in the guidelines are completely followed, this means that the tolerable number of cases of rotavirus disease, caused by the consumption of wastewater-irrigated food, in this developing-country city of 1 million people is:

$$\frac{1\ \text{DALY loss per year}}{2.6 \times 10^{-2}\ \text{DALY loss per case}} = 38\ \text{cases per year}$$

The chance of an individual living in this developing-country city of 1 million becoming ill with rotavirus diarrhea in any one year is (38×10^{-6}) – i.e., 3.8×10^{-5}, which is the tolerable rotavirus disease risk per person per year in developing countries determined in Table 1.1.

Source (first two paragraphs): WHO (2006a).

Table 1.1 DALY losses, disease risks, disease/infection ratios and tolerable infection risks for rotavirus, *Campylobacter* and *Cryptosporidium*

Pathogen	DALY loss per case of disease[a]	Tolerable disease risk pppy equivalent to 10^{-6} DALY loss pppy[b]	Disease/ infection ratio	Tolerable infection risk pppy[c]
Rotavirus: (1) IC[d]	1.4×10^{-2}	7.1×10^{-5}	0.05[e]	1.4×10^{-3}
(2) DC[d]	2.6×10^{-2} [d]	3.8×10^{-5}	0.05[e]	7.7×10^{-4}
Campylobacter	4.6×10^{-3}	2.2×10^{-4}	0.7	3.1×10^{-4}
Cryptosporidium	1.5×10^{-3}	6.7×10^{-4}	0.3	2.2×10^{-3}

[a]Values from Havelaar and Melse, 2003.
[b]Tolerable disease risk = 10^{-6} DALY loss pppy ÷ DALY loss per case of disease.
[c]Tolerable infection risk = disease risk ÷ disease/infection ratio.
[d]IC, industrialized countries; DC, developing countries (there are no IC-DC differences for *Campylobacter* and *Cryptosporidium*).
[e]For developing counties, the DALY loss per rotavirus death has been reduced by 95% as ~95% of these deaths occur in children under the age of 2 who are not exposed to wastewater-irrigated foods. The disease/infection ratio for rotavirus is low as immunity is mostly developed by the age of 3.

Table 1.2 Diarrheal disease (DD) incidence pppy in 2000 by region and age[a]

Region	DD incidence in all ages	DD incidence in 0–4 year olds	DD incidence in 5–80+ year olds
Industrialized countries	0.2	0.2–1.7	0.1–0.2
Developing countries	0.8–1.3	2.4–5.2	0.4–0.6
Global average	0.7	3.7	0.4

[a]Source: Mathers et al., 2002.

Table 1.1 gives the DALY losses per case of rotavirus diarrhoea, campylobacteriosis and cryptosporidiosis and the corresponding disease/infection ratios. From the data in Table 1.1, a value of 10^{-3} pppy was selected as the tolerable rotavirus *infection* risk to be used in the risk analyses in Sections 1.2.3 and 1.2.4; rotavirus was chosen as the overall index pathogen as its associated risks are higher than those for both *Campylobacter* and *Cryptosporidium*. The corresponding tolerable rotavirus *disease* risk is ~10^{-4} pppy, which is extremely safe as it is three orders-of-magnitude lower than the actual incidence of diarrheal disease in the world (Table 1.2), and thus there is a good level of inherent protection against disease outbreaks.

1.2.2. Quantitative Microbial Risk Analyses

The Guidelines adopt a standard QMRA approach (Haas et al., 1999) to risk analysis combined with 10,000-trial Monte Carlo simulations (Mara et al., 2007) to determine required pathogen removals. The basic equations are:

(a) exponential dose-response model (for *Cryptosporidium*):

$$P_1(d) = 1 - \exp(-rd) \tag{1}$$

(b) β-Poisson dose–response model (for rotavirus and *Campylobacter*):

$$P_1(d) = 1 - [1 + (d/N_{50})(2^{1/\alpha} - 1)]^{-\alpha} \quad (2)$$

(c) annual risk of infection:

$$P_{1(A)}(d) = 1 - [1 - P_1(d)]^n \quad (3)$$

where $P_I(d)$ is the risk of infection in an individual exposed to (here, following ingestion of) a single pathogen dose *d* (this "single pathogen dose *d*" is the number of pathogens ingested on one occasion); $P_{I(A)}(d)$ is the annual risk of infection in an individual from *n* exposures per year to the single pathogen dose *d*; N_{50} is the median infective dose (i.e., the dose that causes infection in half the number of people exposed to it); and α and *r* are pathogen "infectivity constants:" for rotavirus $N_{50} = 6.17$ and $\alpha = 0.253$; for *Campylobacter* $N_{50} = 896$ and $\alpha = 0.145$; and for *Cryptosporidium* $r = 0.0042$ (Haas et al., 1999; N_{50}, α and *r* are determined experimentally from human exposure trials).

Box 1.2 gives an example of how these equations are used. As shown in Box 1.2, the end result of the application of equations 1 to 3 is the required log unit reduction of pathogens that corresponds to the targeted rotavirus infection risk of 10^{-3} pppy and hence to the tolerable additional disease burden of 10^{-6} DALY loss pppy.

In combination with Monte Carlo risk simulations, quantitative microbial risk analyses (QMRA) can be used to generate numerical values of the median infection

Box 1.2 Use of the Quantitative Microbial Risk Analysis (QMRA) Equations for Unrestricted Irrigation

This example illustrates how the QMRA equations (equations 1–3) are used to determine the pathogen reduction (in log units[a]) required to protect human health in the case of unrestricted irrigation. The exposure scenario is the consumption of wastewater-irrigated lettuce.

1. **Tolerable risk of infection:** the "design" risk of rotavirus infection is taken as 10^{-3} pppy.
2. **Quantitative microbial risk analysis:** consumer exposure to pathogens is calculated by using the following illustrative parameter values in the QMRA equations:

5000 rotaviruses per liter of untreated wastewater
10 mL of treated wastewater remaining on 100 g lettuce after irrigation
100 g lettuce consumed per person every second day throughout the year
The rotavirus dose per exposure (*d*) is the number of rotaviruses on 100 g lettuce at the time of consumption. The dose is determined by QMRA as follows:
(a) Conversion of the tolerable rotavirus infection risk of 10^{-3} pppy ($P_{I(A)}(d)$ in equation 3) to the risk of infection per person per exposure event ($P_I(d)$ in

(continued)

Box 1.2 (continued)

equations 1 and 2; i.e., per consumption of 100 g lettuce, which takes place every two days throughout the year, so *n* in equation 3 is 365/2). Thus:

$$P_I(d) = 1-(1-10^{-3})^{[(1/365/2)]} = 5.5\times10^{-6}$$

(b) Calculation of the dose per exposure event from equation 2 (the β-Poisson dose–response equation, which is used for rotavirus):

$$P_1(d) = 1 - [1+(d/N_{50})(2^{1/\alpha}-1)]^{-\alpha}$$

$$\text{i.e.,}\ d = \{[1-P_1(d)]^{-1/\alpha}-1\}/\{N_{50}/(2^{1/\alpha}-1)\}$$

The values of the "infectivity constants" for rotavirus are N_{50} = 6.17 and α = 0.253. Thus:

$$d = \{[1-(5.5\times10^{-6})]^{-1/0.253}-1\}/\{6.17/(2^{1/0.253}-1)\} = 5\times10^{-5}\ \text{per exposure event}$$

3. **Required pathogen reduction:** this dose *d* of 5×10^{-5} rotavirus, the maximum dose to keep within the maximum tolerable infection risk, is contained in the 10 mL of treated wastewater remaining on the lettuce at the time of consumption, so the rotavirus concentration is 5×10^{-5} per 10 mL or 5×10^{-3} per liter. The number of rotaviruses in the raw wastewater is 5000 per liter and therefore the required pathogen reduction in log units[a] is:

$$\log(5000) - \log(5 \times 10^{-3}) = 3.7 - (-2.3) = 6$$

[a]A 1-log unit reduction is a reduction of 90%, 2 log units a reduction of 99%, 3 log units a reduction of 99.9%, and so on (thus a "log unit" is strictly a "log10 unit"). Here, the required 6-log unit reduction is a reduction of 99.9999%, where each 9 is a significant figure.

risks related with wastewater irrigation for selected human exposure scenarios. Box 1.3 details how Monte Carlo simulations are made.

1.2.3. Assessing Median Infection Risks in Restricted Irrigation

Restricted irrigation refers to the irrigation of all crops except those eaten uncooked. The model scenario developed for assessing infection risks linked to restricted irrigation is the involuntary ingestion of soil particles by those working, or by young children playing, in wastewater-irrigated fields. This is a likely sce-

Box 1.3 Monte Carlo risk Simulations

The specimen calculations in Box 1.2 use "fixed" values for each parameter (e.g., 10 mL of wastewater remaining on 100 g of lettuce after irrigation; Shuval et al. [1997] measured a mean volume of 10.8 mL). However, there is usually some degree of uncertainty about the precise values of the parameters used in these QMRA equations. This uncertainty is taken into account by assigning to each parameter a range of values (e.g., 10–15 mL of wastewater remaining on 100 g of lettuce after irrigation), although a fixed value can be assigned to any parameter if so wished. A computer program then selects at random a value for each parameter from the range of values specified for it and then determines the resulting risk. The program repeats this process many times (a total of 10,000 times for the simulations reported herein) and then determines the median risk. This large number of repetitions removes some of the uncertainty associated with the parameter values and makes the results generated by multi-trial Monte Carlo simulations much more robust, although of course only as good as the assumptions made.

nario as wastewater-saturated soil would contaminate the workers' or children's fingers and so some pathogens could be transmitted to their mouths and hence ingested. The quantity of soil involuntarily ingested in this way has been reported (but not specifically for this restricted-irrigation scenario) as up to ~100 mg per person per day of exposure (Haas et al., 1999; WHO, 2001). Two sub-scenarios were investigated: (a) highly mechanized agriculture and (b) labor-intensive agriculture. The former represents exposure in industrialized countries where farm workers typically plough, sow and harvest using tractors and associated equipment and can be expected to wear gloves and be generally hygiene-conscious when working in wastewater-irrigated fields. The latter represents farming practices in developing countries in situations where tractors are not used and gloves (and often footwear) are not worn, and where hygiene is commonly not promoted.

Risk simulation for labor-intensive agriculture: The results of the Monte Carlo-QMRA risk simulations are given in Table 1.3 for various wastewater qualities (expressed as single log ranges of *E. coli* numbers per 100 mL, with 10^7–10^8 *E. coli* per 100 mL taken as the quality of untreated wastewater) and for 300 days exposure per year (the footnote to the table gives the range of values assigned to each parameter). From Table 1.3 it can be seen that the median rotavirus infection risk is ~10^{-3} pppy for a wastewater quality of 10^3–10^4 *E. coli* per 100 mL.

Thus, the tolerable rotavirus infection risk of 10^{-3} pppy can be achieved by a 4-log unit reduction (i.e., from 10^7–10^8 to 10^3–10^4 *E. coli* per 100 mL), so that the required wastewater quality is ≤10^4 *E. coli* per 100 mL (at this level the risk given in Table 1.3 is 4.4×10^{-3} pppy, which is slightly high; however, the risk is proportional to the number of days of exposure per year, here taken as 300; in practice the risk will be closer to 10^{-3} pppy).

Table 1.3 Restricted irrigation: labor-intensive agriculture with exposure for 300 days per year: median infection risks from ingestion of wastewater-contaminated soil estimated by 10,000-trial Monte Carlo simulations[a]

Soil quality (*E. coli* per 100 g)[b]	Median infection risk per person per year		
	Rotavirus	*Campylobacter*	*Cryptosporidium*
10^7–10^8	0.99	0.50	1.4×10^{-2}
10^6–10^7	0.88	6.7×10^{-2}	1.4×10^{-3}
10^5–10^6	0.19	7.3×10^{-3}	1.4×10^{-4}
10^4–10^5	2.0×10^{-2}	7.0×10^{-4}	1.3×10^{-5}
10^4	4.4×10^{-3}	1.4×10^{-4}	3.0×10^{-6}
10^3–10^4	1.8×10^{-3}	6.1×10^{-5}	1.4×10^{-6}
100–1000	1.9×10^{-4}	5.6×10^{-6}	1.4×10^{-7}

[a]10–100 mg soil ingested per person per day for 300 days per year; 0.1–1 rotavirus and *Campylobacter*, and 0.01–0.1 *Cryptosporidium* oocyst, per 10^5 *E. coli;* $N_{50} = 6.7 \pm 25\%$ and $\alpha = 0.253 \pm 25\%$ for rotavirus; $N_{50} = 896 \pm 25\%$ and $\alpha = 0.145 \pm 25\%$ for *Campylobacter;* $r = 0.0042 \pm 25\%$ for *Cryptosporidium*. No pathogen die-off (taken as a worst case scenario).
[b]The wastewater quality is taken to be the same as the soil quality (i.e., the soil is assumed, as a worst case scenario, to be saturated with the wastewater).
Note: the median risks for *Campylobacter* and *Cryptosporidium* are all lower than those for rotavirus
Source: WHO, 2006a and Mara et al., 2007

Table 1.4 Restricted irrigation: highly mechanized agriculture with exposure for 100 days per year: median infection risks from ingestion of wastewater-contaminated soil estimated by 10,000-trial Monte Carlo simulations[a]

Soil quality (*E. coli* per 100 g)[b]	Median infection risk per person per year		
	Rotavirus	Campylobacter	*Cryptosporidium*
10^7–10^8	0.50	2.1×10^{-2}	4.7×10^{-4}
10^6–10^7	6.8×10^{-2}	1.9×10^{-3}	4.7×10^{-5}
10^5–10^6	6.7×10^{-3}	1.9×10^{-4}	4.6×10^{-6}
10^5	1.5×10^{-3}	4.5×10^{-5}	1.0×10^{-6}
10^4–10^5	6.5×10^{-4}	2.3×10^{-5}	4.6×10^{-7}
103–10^4	6.8×10^{-5}	2.4×10^{-6}	5.0×10^{-8}
100–1000	6.3×10^{-6}	2.2×10^{-7}	$\leq 1 \times 10^{-8}$

[a]1–10 mg soil ingested per person per day for 100 days per year; 0.1–1 rotavirus and *Campylobacter*, and 0.01–0.1 *Cryptosporidium* oocyst, per 10^5 *E. coli;* $N_{50} = 6.7 \pm 25\%$ and $\alpha = 0.253 \pm 25\%$ for rotavirus; $N_{50} = 896 \pm 25\%$ and $\alpha = 0.145 \pm 25\%$ for *Campylobacter;* $r = 0.0042 \pm 25\%$ for *Cryptosporidium*. No pathogen die-off (taken as a worst case scenario).
[b]The wastewater quality is taken to be the same as the soil quality (i.e., the soil is assumed, as a worst case scenario, to be saturated with the wastewater).
Source: WHO, 2006a and Mara et al., 2007

Risk simulation for highly mechanized agriculture: The simulated risks for various wastewater qualities and for 100 days exposure per year are given in Table 1.4, which shows that the median rotavirus infection risk is ~10^{-3} pppy for a wastewater quality of 10^5 *E. coli* per 100 mL. Thus, a 3-log unit reduction, from 10^7–10^8 to 10^4–10^5 *E. coli* per 100 mL, is required to achieve the tolerable rotavirus infection risk of 10^{-3} pppy, and the required wastewater quality is $\leq 10^5$ *E. coli* per 100 mL.

1.2.4. Assessing the Median Infection Risks in Unrestricted Irrigation

Unrestricted irrigation refers to the irrigation of all crops, including those eaten uncooked. The exposure scenarios used for unrestricted irrigation are the consumption of wastewater-irrigated lettuce (Shuval et al., 1997) and the consumption of wastewater-irrigated onions; these crops were chosen as typical leaf and root vegetables commonly eaten uncooked, although it has not been determined whether the resulting health risks are actually typical for other leaf and root crops. The scenario also includes allowance for pathogen die-off between the last irrigation and consumption.

The results of the Monte Carlo-QMRA risk simulations are given in Table 1.5 for various wastewater qualities (expressed as single log ranges of *E. coli* numbers per 100 mL; the footnote to the table gives the range of values assigned to each parameter). From Table 1.5 it can be seen that the median rotavirus infection risk is 10^{-3} pppy for a wastewater quality of 10^3–10^4 *E. coli* per 100 mL, so the tolerable rotavirus infection risk of 10^{-3} pppy is achieved by a 4-log unit reduction, from 10^7–10^8 to 10^3–10^4 *E. coli* per 100 mL. Hence, the tolerable infection risk could be achieved by treatment to a wastewater quality of $\leq 10^4$ *E. coli* per 100 mL (at 10^4 per 100 mL the risk in Table 1.5 is 2.2×10^{-3} pppy, which is close enough to 10^{-3} pppy). This 4-log unit reduction by treatment would be supplemented by the 2–3 log unit reduction due to rotavirus die-off assumed in these risk simulations (see footnote to Table 1.5; this die-off would occur in warm climates in ~2 days; cf. Table 7), so giving a total pathogen reduction of 6–7 log units (cf. the specimen calculations in Box 1.2).

A 4-log unit reduction by treatment for unrestricted irrigation is also protective of the fieldworkers (see "Labor-intensive agriculture" in Section 1.2.3).

Table 1.5 Unrestricted irrigation: median infection risks from the consumption of wastewater-irrigated lettuce estimated by 10,000-trial Monte Carlo simulations[a]

Wastewater quality (*E. coli* per 100 mL)	Median infection risk per person per year		
	Rotavirus	*Campylobacter*	*Cryptosporidium*
10^7–10^8	0.99	0.28	0.50
10^6–10^7	0.65	6.3×10^{-2}	6.3×10^{-2}
10^5–10^6	9.7×10^{-2}	2.4×10^{-3}	6.3×10^{-3}
10^4–10^5	9.6×10^{-3}	2.6×10^{-4}	6.8×10^{-4}
10^4	2.2×10^{-3}	1.3×10^{-4}	4.5×10^{-4}
10^3–10^4	1.0×10^{-3}	2.6×10^{-5}	3.1×10^{-5}
100–1000	8.6×10^{-5}	3.1×10^{-6}	6.4×10^{-6}
10–100	8.0×10^{-6}	3.1×10^{-7}	6.7×10^{-7}

[a] 100 g lettuce eaten per person per 2 days; 10–15 mL wastewater remaining on 100 g lettuce after irrigation; 0.1–1 rotavirus and *Campylobacter*, and 0.01–0.1 oocyst, per 10^5 *E. coli*; 10^{-2}–10^{-3} rotavirus and *Campylobacter* die-off, and 0–0.1 oocyst die-off, between last irrigation and consumption; $N_{50} = 6.7 \pm 25\%$ and $\alpha = 0.253 \pm 25\%$ for rotavirus; $N_{50} = 896 \pm 25\%$ and $\alpha = 0.145 \pm 25\%$ for *Campylobacter;* $r = 0.0042 \pm 25\%$ for *Cryptosporidium*.

Note: the median risks for *Campylobacter* and *Cryptosporidium* are all lower than those for rotavirus

Source: WHO, 2006a and Mara et al., 2007.

Table 1.6 Unrestricted irrigation: required pathogen reductions for various levels of tolerable risk of infection from the consumption of wastewater-irrigated lettuce and onions estimated by 10,000-trial Monte Carlo simulations[a]

Tolerable level of rotavirus infection risk (pppy)	Corresponding required level of rotavirus reduction (log units)[b]	
	Lettuce	Onions
10^{-2}	5	6
10^{-3}	6	7
10^{-4}	7	8

[a] 100 g lettuce and onions eaten per person per 2 days; 10–15 mL and 1–5 mL wastewater remaining after irrigation on 100 g lettuce and 100 g onions, respectively; 0.1–1 and 1–5 rotavirus per 10^5 *E. coli* for lettuce and onions, respectively; $N_{50} = 6.17 \pm 25\%$ and $\alpha = 0.253 \pm 25\%$.
[b] Assuming the raw wastewater quality to be 10^7–10^8 *E. coli* per 100 mL.
Source: WHO, 2006a.

Table 1.6 gives the required total log unit reductions for unrestricted irrigation of lettuce and onions for various levels of tolerable rotavirus infection risk: 10^{-2}, 10^{-3} and 10^{-4} pppy (these Monte Carlo simulations are the reverse of those in Tables 1.3 to 1.5 as they first set the risk and then determine the required total pathogen reduction). Table 1.6 shows that (a) the consumption of root crops requires a 1-log unit pathogen reduction greater than the consumption of non-root crops, and (b) the required pathogen reductions change by an order of magnitude with each order-of-magnitude change in tolerable risk.

In England, the guidelines for the microbiological quality of "ready-to-eat" foods (such as prepared sandwiches and salads on sale in local shops and supermarkets) state that up to 10,000 fecal coliforms per 100 g is "acceptable" (Gilbert et al., 2000). Lettuce is a common component of many ready-to-eat foods, so it makes little sense to irrigate lettuces with wastewater treated to a higher quality than that required of the lettuces themselves.

1.3. Achieving the Required Pathogen Reduction

The 2006 WHO guidelines allow health risks to be managed not only by wastewater treatment, crop restriction, irrigation techniques and human exposure control (as in the 1989 guidelines), but also by pathogen die-off before consumption and food preparation measures.

1.3.1. Wastewater Treatment

Probably the most obvious approach to reduce risk of infection from wastewater is the removal or inactivation of pathogens through wastewater treatment. Conventional treatment technologies, however, focus mainly on the removal of suspended solids, organic matter and nutrients such as nitrogen and phosphorus, and not on the removal of pathogens. Water reclaimed through conventional treatment may therefore require further treatment such as filtration or disinfection to reduce the concentration of pathogens to an acceptable level. On the other hand, some unconventional wastewater treatment technologies have been shown to be more effective in removing pathogens.

In most situations in most developing countries, waste stabilization ponds are the most appropriate option for wastewater treatment (Mara, 2004; von Sperling and de Lemos Chernicharo, 2005). In warm climates, a series of ponds comprising an anaerobic pond, a secondary facultative pond and a single maturation pond can produce an effluent with $\leq 10^4$ *E. coli* per 100 mL (and also with ≤1 helminth egg per liter). (The anaerobic ponds can be covered and the biogas collected and used for such purposes as cooking or electricity generation [DeGarie et al., 2000], another form of wastewater use.)

1.3.2. Post-Treatment Health Protection Control Measures

There are various ways by which pathogen numbers are or can be reduced after treatment. The main post-treatment health protection control measures and the log unit pathogen reductions they achieve are listed in Table 1.7. These log unit reductions are extremely reliable: in essence they always occur. Hygiene education may be required in some societies to ensure that salad crops and vegetables when eaten raw are always washed in clean water prior to consumption, but this is not (at least in hygiene education terms) an arduous task. On the other hand, root crops (such as onions) are peeled before they are eaten. Post-treatment health protection control measures are only relevant for unrestricted irrigation, since in restricted irrigation the crops are cooked before consumption, leading to total pathogen inactivation.

In unrestricted irrigation, for a tolerable rotavirus infection risk of 10^{-3} pppy, the 4-log unit reduction by treatment must be supplemented by post-treatment control

Table 1.7 Post-treatment health protection control measures and corresponding pathogen reductions achieved

Control measure	Pathogen reduction (log units)	Notes
Drip irrigation	2–4	2-log unit reduction for low-growing crops, 4-log unit reduction for high-growing crops.
Pathogen die-off	0.5–2 per day	Die-off after last irrigation before harvest (value depends on climate, crop type, etc.).
Produce washing	1	Washing salad crops, vegetables and fruit with clean water.
Produce disinfection	2	Washing salad crops, vegetables and fruit with a weak disinfectant solution and rinsing with clean water.
Produce peeling	2	Fruits, root crops.

Source: WHO, 2006a.

measures totaling 2 log units for non-root crops and 3 log units for root crops (see Table 1.6). This could be achieved, for example, by a 1-log unit reduction due to die-off and a 1-log unit reduction by produce washing (or a 2-log unit reduction due to die-off) for non-root crops; and a 1-log unit reduction due to die-off and a 2-log unit reduction by produce peeling for root crops. This then gives the required total log unit reduction of 6 for non-root crops and 7 for root crops. However, it is likely that there will always be at least a 2-log unit reduction due to die-off in warm-climate countries (rather than the 1-log unit reduction assumed earlier), so that there will always be a factor of safety of at least one order-of-magnitude.

1.4. Reuse of Greywater

Unrestricted irrigation with greywater is beneficial as it increases crop yields and pathogen levels are low (Jackson et al., 2006; WHO, 2006b). The health risks are lower than those from domestic wastewater (i.e., grey and black waters combined), as pathogen numbers are much lower due to the much smaller fecal load in greywater (a cross-contamination load of ~0.04 g feces per person per day enters greywater, giving a greywater quality of ~10^4–10^5 *E. coli* per 100 mL, compared with 10^7–10^8 per 100 mL for domestic wastewater; WHO, 2006b). The QMRA studies reported in WHO (2006b) indicate that a 1.6- to 2.9-log unit reduction is required for protozoan pathogens (*Cryptosporidium, Giardia*) and a 2.3- to 3.3-log unit reduction for viral pathogens (rotavirus) so that the tolerable additional disease burden of 10^{-6} DALY loss pppy is not exceeded. These reductions can be achieved, as in the case of wastewater, by a combination of treatment and post-treatment health protection control measures (Table 1.7). But it will be apparent that little, if any, treatment is necessary as these pathogen reductions are achievable solely through pathogen die-off and produce washing/disinfection/peeling. Even so, retention in a tank for a few hours would be beneficial to remove scum and readily settleable solids.

1.5. Transposition of the Guidelines Into National Practice

The WHO 2006 guidelines are recommendations of good practice. In themselves they have no legal status in any jurisdiction. Governments can choose to adopt or adapt and adopt (or, of course, even ignore) the guidelines, and they can decide whether to transpose them into legally enforceable national standards or to keep them only as recommendations of good practice. The government departments normally involved in this decision-making process are Ministries or Departments of Health, Water, Environment and Finance, including the part of government responsible for food safety.

There are two basic decisions to be made, as follows:

1. Decision 1: are the Guidelines to be transposed into national standards or only endorsed as recommendations for good national practice?

2. Decision 2: Is the tolerable additional burden of disease of 10^{-6} DALY loss pppy appropriate for local conditions? This is an important decision as the value used for this controls the tolerable disease and infection risks pppy (Table 1.1) and thus the degree (and hence cost) of wastewater treatment needed to ensure that these risks are not exceeded. Is a value of 10^{-5} DALY loss pppy locally more appropriate?

The following points should be taken into consideration in making the second decision:

1. A stricter requirement would not normally be needed since, as noted earlier, a DALY loss of 10^{-6} pppy is the value used by WHO (2004) in its drinking water quality guidelines. Thus the consumption of wastewater-irrigated food is as safe as drinking fully treated drinking water if the recommendations in the 2006 guidelines are followed.
2. A less stringent requirement results in higher tolerable disease and infection risks pppy. For example, a tolerable additional disease burden of 10^{-5} DALY loss pppy would increase the disease and infection risks in Table 1.1 by a factor of 10, resulting in a tolerable rotavirus disease risk of 10^{-3} pppy, which is still two orders of magnitude lower than the current global incidence of diarrheal disease of 0.1 to 1 pppy (Table 1.2). The corresponding tolerable rotavirus infection risk is 10^{-2} pppy and therefore the required effluent qualities discussed earlier become one order-of-magnitude less stringent (for example, for restricted irrigation with labor-intensive agriculture, the required wastewater quality is $\leq 10^5$ *E. coli* per 100 mL, rather than $\leq 10^4$ per 100 mL). Governments may decide that this level of health protection (i.e., 10^{-5} DALY loss pppy) is sufficient if the local incidence of diarrhoeal disease is high (i.e., closer to 1 pppy than to 0.1 pppy). (Countries with a high diarrheal disease incidence include, of course, many developing countries, but also Australia [~0.9 pppy; Hall et al., 2006] and the United States [~0.8 pppy; Mead et al., 1999]).
3. An alternative basis for choosing 10^{-5} (rather than 10^{-6}) DALY loss pppy might be that the additional cost of wastewater treatment to meet the 10^{-6} DALY loss pppy is not affordable (or the extra money would be better spent on something else). This could be a decision for the medium-to-long term (especially if the local incidence of diarrhoeal disease is high), or for the short-to-medium term (unaffordable now, but the intention would be to upgrade treatment to meet the 10^{-6} DALY loss pppy in the not-too-distant future).
4. As treatment is required more to protect the fieldworkers (it is the only health protection measure available for restricted irrigation), a decision could be taken to adopt a 10^{-5} DALY loss pppy for the fieldworkers (for whom additional measures should be required, such as the provision by their employers of oral rehydration salts and access to medical assistance), whilst maintaining a 10^{-6} DALY loss pppy for unrestricted irrigation (i.e., adopting this level of health protection for consumers) by ensuring that an additional 1-log unit pathogen reduction is provided by the post-treatment health protection control measures listed in Table 1.7.

Thus there are three options and these are summarized in Table 1.8, together with their requirements for treatment and post-treatment health protection control

Table 1.8 Summary of requirements for wastewater treatment and post-treatment health protection control measures for restricted and unrestricted irrigation for health protection levels of 10^{-6} and 10^{-5} DALY loss per person per year[a]

Heath protection level	Irrigation and farming system	Wastewater treatment requirements	Post-treatment health protection control measures (Table 7)
1. 10^{-6} DALY loss pppy	(a) Restricted irrigation		
	(i) Labor-intensive agriculture	4-log unit pathogen reduction (i.e., to $\leq 10^4$ *E. coli* per 100 mL)	Not applicable
	(ii) Highly mechanized agriculture	3-log unit pathogen reduction (i.e., to $\leq 10^5$ *E. coli* per 100 mL)	Not applicable
	(b) Unrestricted irrigation		
	(i) Labor-intensive agriculture	4-log unit pathogen reduction (i.e., to $\leq 10^4$ *E. coli* per 100 mL)	Provision of additional 2-log unit pathogen reduction for non-root crops and 3-log unit reduction for root crops
	(ii) Highly mechanized agriculture	3-log unit pathogen reduction (i.e., to $\leq 10^5$ *E. coli* per 100 mL)	Provision of additional 3-log unit pathogen reduction for non-root crops and 4-log unit reduction for root crops
2. 10^{-5} DALY loss pppy	(a) Restricted irrigation		
	(i) Labor-intensive agriculture	3-log unit pathogen reduction (i.e., to $\leq 10^5$ *E. coli* per 100 mL)	Not applicable
	(ii) Highly mechanized agriculture	2-log unit pathogen reduction (i.e., to $\leq 10^6$ *E. coli* per 100 mL)	Not applicable
	(b) Unrestricted irrigation		
	(i) Labor-intensive agriculture	3-log unit pathogen reduction (i.e., to $\leq 10^5$ *E. coli* per 100 mL)	Provision of additional 2-log unit pathogen reduction for non-root crops and 3-log unit reduction for root crops
	(ii) Highly mechanized agriculture	2-log unit pathogen reduction (i.e., to $\leq 10^6$ *E. coli* per 100 mL)	Provision of additional 3-log unit pathogen reduction for non-root crops and 4-log unit reduction for root crops
3. 10^{-6} DALY loss pppy for consumers, and 10^{-5} DALY loss pppy for fieldworkers	Unrestricted irrigation: labor-intensive agriculture	3-log unit pathogen reduction (i.e., to $\leq 10^5$ *E. coli* per 100 mL)	Provision of additional 3-log unit pathogen reduction for non-root crops and 4-log unit reduction for root crops

[a]Based on the risk analyses summarized in Tables 3–6 and the values in Table 7.

measures. This table can easily be modified if the less stringent additional disease burden is 10^{-4} (rather than 10^{-5}) DALY loss pppy; this approach could be used as the first step in areas where there is currently extensive use of untreated wastewater for irrigation.

1.6. Conclusions

The 2006 WHO guidelines represent a radical departure from the 1989 guidelines, but they are much more soundly based on the protection of human health. The starting point is the acceptance of the tolerable additional burden of disease used in the 2004 WHO drinking water quality guidelines of $\leq 10^{-6}$ DALY loss per person per year that translates to a tolerable rotavirus infection risk of 10^{-3} pppy. The use of quantitative microbial risk analyses based on likely human exposure scenarios results in robust estimates of the risks to human health from, and the corresponding pathogen reductions required for, both restricted and unrestricted irrigation. National governments have to decide whether this baseline value of 10^{-6} DALY loss pppy is appropriate or whether to adopt, at least initially, a higher value (10^{-5} or even 10^{-4} DALY loss pppy). The recommendations in the 2006 guidelines can be confidently used without the general need in all cases to undertake case-specific estimates of the risks to human health resulting from the use of wastewater and greywater for crop irrigation.

References

Bartram J, Fewtrell L, Stenström T-A. (2001). Harmonized assessment of risk and risk management for water-related infectious disease: an overview. In: Fewtrell L, Bartram J, eds. Water quality: Guidelines, Standards for Health, Assessment of Risk and Risk Management for Water-Related Infectious Disease. IWA Publishing, London, pp. 2–16.

DeGarie CJ, Crapper T, Howe BM, Burke BF, McCarthy PJ. (2000). Floating geomembrane covers for odour control and biogas collection and utilization in municipal lagoons. Water Sci Technol 42:291–298.

Feachem RG, Bradley DJ, Garelick H, Mara DD. (1983). Sanitation and Disease: Health Aspects of Wastewater and Excreta Management. John Wiley & Sons, Chichester.

Gilbert RJ, de Louvois J, Donovan T, et al. (2000). Guidelines for the microbiological quality of some ready-to-eat foods sampled at the point of sale. Commun Dis Public Health 3:163–167.

Haas CN, Rose JB, Gerba CP. (1999). Quantitative Microbial Risk Assessment. John Wiley & Sons, New York.

Hall GV, Kirk MD, Ashbolt R, Stafford R, Lolar K. (2006). Frequency of gastrointestinal illness in Australia 2002: regional, seasonal and demographic variation. Epidemiol Infect 134:111–118.

Havelaar AH and Melse JM. (2003). Quantifying public health risk in the WHO guidelines for drinking-water quality: a burden of disease approach (RIVM Report No. 734301022/2003). Rijksinstituut voor Volksgezondheid en Milieu, Bilthoven.

Jackson S, Rodda N, Salukazana L. (2006). Microbiological assessment of food crops irrigated with domestic greywater. Water SA 32:700–704.

Mara DD. (2004). Domestic wastewater treatment in developing countries. Earthscan Publications, London.

Mara DD, Sleigh PA, Blumenthal UJ, Carr RM. (2007). Health risks in wastewater irrigation: comparing estimates from quantitative microbial risk analyses and epidemiological studies. J Water Health 5:39–50.

Mathers CD, Stein C, Ma Fat D, et al. (2002). Global Burden of Disease 2000, Version 2: Methods and Results. World Health Organization, Geneva.

Mead PS, Slutsker L, Dietz V, et al. (1999). Food-related illness and death in the United States. Emerg Infect Dis 5:605–625.

Murray CJL and Lopez AD. (1996). The Global Burden of Disease, Volume 1: A Comprehensive Assessment of Mortality and Disability From Diseases, Injuries, and Risk Factors in 1990 and Projected to 2020. Harvard University Press, Cambridge, MA.

Prüss A and Havelaar A. (2001). The global burden of disease study and applications in water, sanitation, and hygiene. In: Fewtrell L and Bartram J, eds. Water quality: Guidelines, Standards for Health, Assessment of Risk and Risk Management for Water-Related Infectious Disease. IWA Publishing, London, pp. 43–59.

Shuval HI, Lampert Y, Fattal B. (1997). Development of a risk assessment approach for evaluating wastewater reuse standards for agriculture. Water Sci Technol 35:15–20.

von Sperling M and de Lemos Chernicharo CA. (2005). Biological Wastewater Treatment in Warm Climate Regions. IWA Publishing, London.

World Health Organization (WHO). (1989). Health Guidelines for the Use of Wastewater in Agriculture and Aquaculture (Technical report series 778). WHO, Geneva.

World Health Organization (WHO). (2001). Depleted Uranium: Sources, Exposure and Health Effects (Report WHO/SDE/PHE/01.1). WHO, Geneva.

World Health Organization (WHO). (2004). Guidelines for Drinking-Water Quality, third edition. WHO, Geneva.

World Health Organization (WHO). (2006a). Guidelines for the Safe Use of Wastewater, Excreta and Greywater, Volume 2: Wastewater Use in Agriculture. WHO, Geneva.

World Health Organization (WHO). (2006b). Guidelines for the Safe Use of Wastewater, Excreta and Greywater, Volume 4: Excreta and Greywater Use in Agriculture. WHO, Geneva.

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Chapter 2
EMWater Guide and Recommendations on Wastewater Treatment and Water Reuse

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Abstract EMWater Guide: Improving wastewater treatment and reuse practices in the Mediterranean countries–A Practical Guide for Decision-Makers has been developed for the MEDA region within the EMWater Project, which is funded by the EU MEDA Water Programme and InWEnt Capacity Building International, Germany. The overall objective of the EMWater Guide is to provide guidance on taking decisions in wastewater management. The target groups are officials and decision makers mainly on municipal level, including people without engineering background and NGOs and consultants, active in the wastewater field. The EMWater Guide shall encourage its users to take into account all relevant framework conditions and alternative solutions when selecting appropriate technologies for wastewater treatment and reuse. It shall enable decision makers in the field of wastewater treatment and reuse through an easy-to-understand, concise manual. The Guide does not replace in-depth analysis of specific conditions and expert consultation once the decision to start a wastewater project has been taken.

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I. Al Baz et al. (eds.), *Efficient Management of Wastewater.*

2.1. Introduction

The Mediterranean region is one of the areas in the world most affected by water shortage. Moreover, the demand for water is expanding due to population growth, rising standards of living, urbanization, increasing economic activities and expanding areas of irrigated agriculture. Therefore, improved water demand management and the development of new water resources are urgently needed. On the other hand, today, in many MEDA countries, wastewater is not always adequately treated, leading to (further) deterioration of the quality of existing freshwater resources and the Mediterranean Sea. Hence, the proper management, treatment and reuse of wastewater could be a valuable alternative to using freshwater resources, especially in water-scarce countries.

2.1.1. Wastewater Treatment and Reuse Situation in the EMWater Partner Countries

Fresh water availability varies significantly between the four EMWater partner countries: Jordan, Lebanon, Palestine and Turkey. Jordan and Palestine are already seriously affected by water scarcity. Lebanon and Turkey, on the other hand, are today still classified as water-rich countries but are likely to face water scarcity within the next decades. In Turkey, the water availability varies strongly from region to region.

In the whole MEDA region, the largest consumer of water is by far irrigated agriculture with around 70% of water usage. And the demand for irrigation water is growing constantly. Here wastewater could be a valuable alternative to fresh water resources, if managed and treated properly. Agricultural irrigation has the major potential for water reuse applications. In Jordan, for example, the desperate need for water has led to the reuse of treated wastewater in agriculture for many years. In Lebanon, Palestine and Turkey, on the other hand, water reuse application is very limited to date. Proper wastewater management is required to allow for water reuse, but today, even if the standard of wastewater management differs in the four countries, in most cases it can be described as insufficient in general.

Reasons for failing to promote proper wastewater management and reuse in the region have mainly socio-cultural and technical origins. In addition, the lack of laws and regulations or the lack of their enforcement contributes to rejection of reuse.

2.1.2. Methods and Means Used for Developing the Guide

Methods and means used for the development of the EMWater Guide have been know-how and experience of the EMWater project partners, literature and Internet research, a background paper on water reuse guidelines by Adelphi Research as well as a questionnaire survey on the guide's contents: 50 questionnaires have been filled in the MEDA partner countries. The respondents included staff at municipalities, authorities, ministries, universities, utilities, user groups, etc. The general outline of the guide has been developed according to the results of this questionnaire survey. After preparing a draft version of the guide, a multistage review process by all project partners, experts in the field and from the region and by local stakeholders was initiated. Especially the MEDA partners of the project team have been involved in collecting local feedback on the guide's structure and content. The response was not as widespread/extensive as hoped for, but sufficiently qualified and substantial comments have been received, the final version of the guide being available since mid-March 2007.

2.1.3. Objective, Target Groups and Structure of the Guide

The EMWater Guide shall support decision making in wastewater management and the planning of related projects. The main target groups are officials on the municipal level (i.e., people who do not necessarily have a background in engineering or natural sciences). Therefore, the guide does not aim to present detailed information on, for example, technological or biological aspects of wastewater management. Rather, it intends to present the main criteria for decision making in a concise way—easy to understand and in short form, using figures and tables as much as possible. For detailed information, the reader is referred to existing literature (e.g., information that can be found in the Internet). The guide also provides lists of references and other sources of information for planning and implementing wastewater projects. It could, therefore, also be useful for other stakeholders, such as NGOs or consultants active in the field or authorities on the national level.

It is important to point out that the guide cannot replace in-depth analyses of the existing framework conditions, feasibility studies and other surveys. For implementing wastewater projects, the involvement of experts from different disciplines remains crucial. To give the reader a short overview of what to expect and also to clarify the limitations of the guide, a fact sheet is included in the beginning of the guide to summarize the objectives, target groups, structure, focus, methods and means.

The guide consists of two parts, one on wastewater collection and treatment and a second on wastewater reuse.
Part I, "Wastewater Collection & Treatment," focuses on:

- small communities
- small centralised and
- small decentralised systems

Part II, "Wastewater Reuse," focuses on:

- small communities
- reuse of treated municipal wastewater
- reuse for irrigation in agriculture

In addition to the two main parts, the guide also incorporates an introduction and an annex. The Introduction contains the EMWater recommendations (see Chapter 2.4 of the guide), a short review of existing policies and legislation regarding wastewater treatment and reuse in the four EMWater partner countries and a glossary. The Annex lists limit values of selected standards (e.g., World Health Organization, U.S. Envionmental Protection Agency) and regulations in the MEDA region and elsewhere (e.g., Mexico, Tunisia, Jordan, Turkey and Palestine), a link-list of regional and international project experiences with wastewater reuse and a link-list for awareness raising materials. Furthermore, the Annex presents selected organisations involved in wastewater treatment and reuse, and other useful sources of information.

2.2. Part I: Guide for Wastewater Collection & Treatment

Part I of the EMWater Guide focuses on small centralized and decentralized treatment systems for small communities in rural areas. Wastewater treatment systems for small communities are of high concern in the MEDA region. They represent the majority of the existing systems and have to deal with specific conditions, such as high fluctuations of hydraulic and organic loads (on a daily, weekly and monthly basis) and the need for easy management and operation.

The following criteria are considered most important for long-term sustainability of wastewater collection and treatment concepts in suburban and rural areas of the MEDA countries:

- affordability; especially low operation costs
- operability; operation must be easily possible with locally available staff and support
- reliability; producing a safe effluent for water reuse
- environmentally sound; for example, little sludge production and low energy consumption
- suitability in Mediterranean climate; average wastewater temperature, for example, in Istanbul is 23 °C in July and 15 °C in January)

Part I of the EMWater Guide provides an overview of different collection and treatment systems and supports the selection process of the most feasible solutions. Therefore, the strengths and the weaknesses of each alternative are highlighted with a special focus on low-cost and easy-to-manage treatment techniques. Also, non-technical criteria such as social participation and acceptance are included to provide sustainable sanitation.

In the first part, the advantages and disadvantages of centralized and decentralized wastewater treatment systems are discussed as the decision "decentralized or centralized" is the first choice that planers and decision makers have to make. Information is given on how to find an appropriate solution.

In the following, different wastewater treatment technology options are described, including on-site systems as well as small treatment systems serving urban centres and peri-urban areas. Aspects and parameters affecting the selection of the appropriate treatment system are highlighted. The presented technologies include various extensive systems, such as constructed wetlands, waste stabilization ponds (natural lagoons) and aerated lagoons, as well as intensive treatment systems, such as Imhoff tanks, biofilm systems, activated sludge systems, hybrid technology and anaerobic systems (UASB* reactors). Furthermore, tertiary treatments are shortly explained and also the very important topic of sludge production and management is discussed.

Part I concludes with explaining the process of selecting an appropriate small wastewater treatment system. The selection of a certain type of wastewater treatment technology depends not only on technical considerations, but also on many non-technical factors and issues linked to the local context, such as regulatory requirements, economic and environmental factors associated with a sanitation program, and social factors that are important for the long-term acceptance and sustainability of a system. Therefore, several criteria need to be taken into account when selecting an appropriate sanitation system, such as:

- population to be treated (PE)
- water supply availability (per capita)
- type of the final destination (reuse, ground, underground, rivers, lakes, sea, transition water bodies)
- quality level of the final destination
- effluent standards
- financial aspects: construction and operation and maintenance costs
- land availability
- energy availability
- sludge production and disposal management
- local climate
- operator expertise
- management model to be applied

The selected treatment system should be assembled in a way that ensures that requirements for effluent quality are met and yet is compatible with other site limitations. Therefore, the different criteria need to be balanced. Part I of the EMWater Guide presents information to support the decision of which treatment system might be appropriate under certain conditions. Table 2.1 provides a range of treatment options that are described in the guide.

2.3. Part II: Guide for Water Reuse

Part II of the EMWater Guide focuses on small communities, reuse of treated municipal wastewater and reuse for irrigation in agriculture. The reuse of treated wastewater can be a valuable alternative to freshwater resources, especially in water-scarce

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* Upflow Anaerobic Sludge Blanket (UASB)

Table 2.1 Treatment processes suitable for removal of the main sewage components

Contaminant	Treatment Process
Organic material	• Anaerobic pre-treatment (e.g., UASB) • Facultative ponds • Biofilm systems • Activated sludge systems • Land treatment
Suspended solids	• Pre-treatments (screening and comminution) • Septic tanks • Anaerobic ponds • Primary sedimentation
Ammonia	• Biofilm systems • Activated sludge systems
Nitrogen (total and oxidised)	• Activated sludge systems
Phosphate	• Activated sludge systems • Chemical addition
Excreted pathogens	• Maturation ponds • Disinfection (chlorination, ozonation, UV, membrane UF and MF)

countries. Today, technically proven wastewater treatment and purification processes exist to produce water of almost any quality desired. Treated wastewater can be reused for many different applications. Most common is the reuse for agricultural irrigation. Industrial reuse and groundwater recharge are also largely applied, and the reuse in aquacultures and for landscape irrigation is becoming more and more common.

In the beginning of Part II of the EMWater Guide, an overview of the most common reuse applications is given (irrigation in agriculture or for landscaping, reuse in aquacultures, groundwater recharge, industrial recycling and reuse). The quality requirements, benefits, risks and potential constraints for each type of reuse are described. The main benefits, risks and potential constraints of water reuse of each reuse application are summarized in Tables 2.2 and 2.3.

The next chapter of Part II supports the reader in selecting a feasible reuse application. The process of selecting an appropriate reuse application starts from a very general identification of reclaimed water supply and demand. Step by step, through further analyses of framework conditions and requirements, the range of potential reuse applications will be reduced. The selection process includes the following steps:

1. Inventory of potential sources and demand for wastewater
2. Identification of legal requirements and responsible institutions
3. Detailed analysis of reuse alternatives (benefits and risks)
4. Economic evaluation
5. Financial feasibility check

Table 2.2 Main benefits of wastewater reuse

Irrigation	Aquaculture	Groundwater Recharge	Industrial reuse
Agriculture: Additional water available to farmers Nutrients of WW (N, P) can be used as fertilizer. Need for artificial fertilizer reduced -> Cost reduction. Landscaping: Reuse of treated WW can help to control desertification and support desert reclamation.	Reclaimed WW can be profitably used as a fertilizer in aquacultures to increase plankton growth and thus to produce natural food for fish.	Advantages are the same as for groundwater recharge with fresh water, such as: • Establishment of salt-water intrusion barriers in coastal aquifers • Replenishment of aquifers • Control or prevention of ground subsidence • Storage of reclaimed water for future uses Additional WWT through infiltration and percolation though the soil. This may eliminate the need for further advanced WWT processes.	In-plant recycling and reuse help to meet or avoid stringent regulatory standards for effluent discharges. Recycling and reuse of the water can also facilitate the recycling and reuse of other valuable constituents in the WW. It often saves costs.

WW, wastewater; WWT, wastewater treatment; N, nitrogen; P, phosphorus.

Table 2.3 Main risks and potential constraints of wastewater reuse

Irrigation	Aquaculture	Groundwater Recharge	Industrial reuse
Irrigational reuse often associated with environmental and health risks. Acceptability depends on whether health risks and environmental impacts are tolerable. Water quantity requirements vary seasonally -> water storage systems required. Adapted irrigation management required (e.g., risk of clogging irrigation systems, risk of increased salinity in soil and deterioration of soil quality, etc.)	WW needs to be treated before reuse. The amount of wastewater fed into the ponds needs to be managed properly to prevent overload.	Poorly planned recharge can contaminate the aquifer (e.g., with pathogens, chemicals or trace organic compounds) and therefore harm the environment and human health.	Constituents in reclaimed water can be related to: • scaling • corrosion • biological growth • fouling Health issues need to be considered, particularly because of aerosol transmission of pathogens in cooling water.

Figure 2.1 is a flow chart that gives an overview of the selection process.

Also included in Part II of the EMWater Guide is the information on ways to prevent health risks caused by pathogens and parasites when using reclaimed water for irrigation, and on the importance of awareness raising, education, and

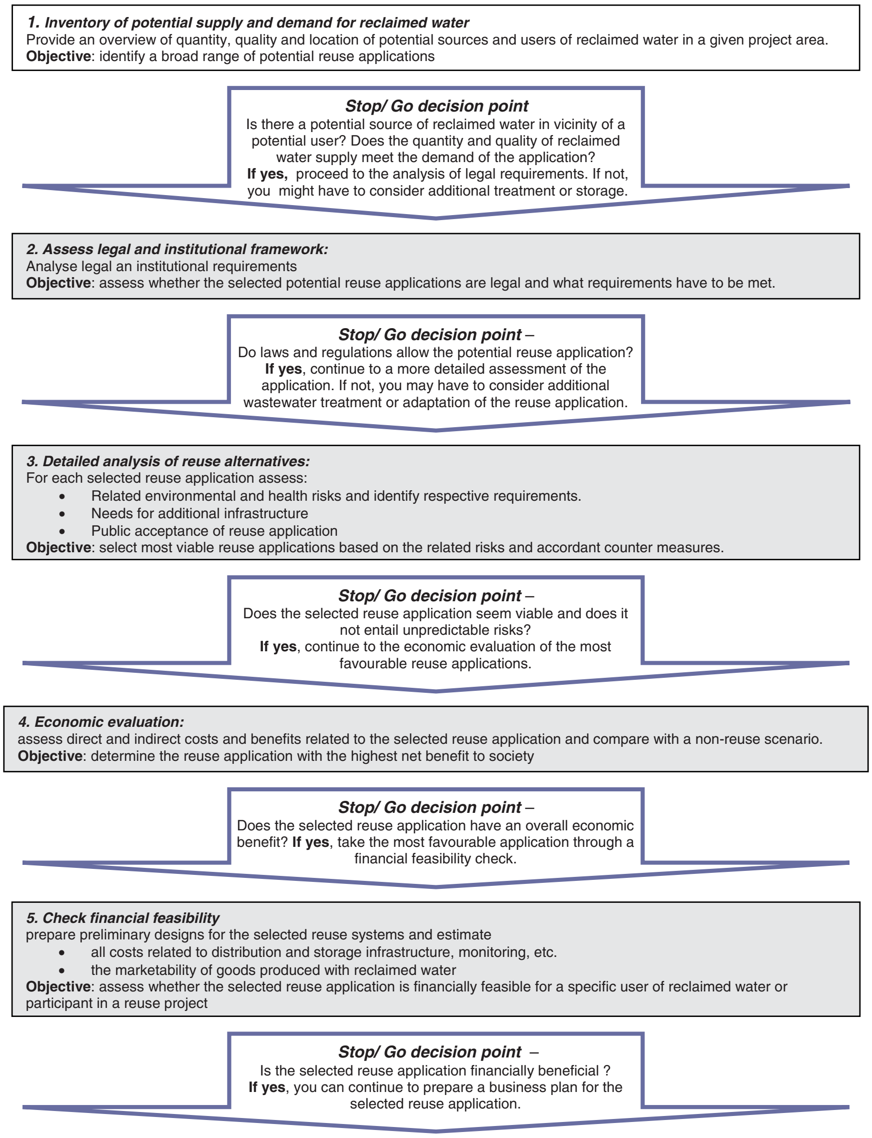

Figure 2.1 Overview of the selection process

capacity building. Reclaimed water reuse is one of the main options considered as a new source of water in water-scarce regions. However, such practice can also entail risks, and therefore adequate information and training of the potential users and proper handling and management of the treated wastewater is most important.

2.4. EMWater's Wastewater Treatment and Reuse Recommendations

Based on research, studies and project work, the EMWater project team has summarized the following recommendations to consider when implementing a wastewater treatment and reuse system:

- Decentralized wastewater treatment should always be considered at first since costs for sewerage networks can make up to 80% of wastewater treatment costs in total.
- Local water management in the long run and the protection of scarce water resources has to be taken into account to identify sustainable wastewater treatment and reuse projects.
- Operation and maintenance costs, including energy costs, should carefully be calculated when selecting a wastewater treatment technology.
- The development of tariffs for treated effluent should be considered to ensure recovery of costs for operation and maintenance of treatment systems.
- Consider appropriate technology for wastewater treatment. High technology is not always the best option, it should be ensured that there are sufficient financial and human capacities to operate and maintain the facilities properly.
- Consider source separation right from the beginning (e.g., domestic <-> industrial wastewater, rainwater ⇔ greywater ⇔ blackwater.
- Pathogen removal is essential for reuse in agriculture, removal of suspended solids and organic matter is therefore important.
- Nitrogen-removal is not always necessary when the reclaimed water is reused in agriculture or aquaculture. However, it has to be ensured that groundwater aquifers are not contaminated through seepage of reclaimed water. Also, seasonal fluctuations of nutrient requirements of crops need to be taken into account.
- Water reuse has major benefits since it can be a valuable alternative to freshwater resources. It allows costs to be saved for wastewater treatment and for fertilizers and can increase agricultural production.
- Market assessments need to be done for reclaimed water and produce to make sure that reuse projects are accepted and financially feasible.
- Additional costs for transfer, storage, distribution and drainage need to be considered when planning a reuse project.
- Microbiological water quality standards are only one way to prevent health risks, other measures such as crop restriction and human exposure control should also be taken into account.
- Any legal standards to regulate water reuse need to be adapted to local conditions. They should not be too strict, in order to promote reuse.
- Regulations for reuse should be affordable, achievable and enforceable.
- Awareness raising is a major issue in reuse projects. Campaigns for farmers and consumers should be included at the project planning stage.

- Water reuse could be promoted by subsidising pilot-scale and demonstration project, which can be visited by local farmers, decision makers and the interested public.

References

The article is based on and summarizes the "EMWater Guide: Improving wastewater treatment and reuse practices in the Mediterranean countries – A Practical Guide for Decision-Makers". The EMWater Guide can be downloaded from the website: http://www.emwater.org

InWEnt Capacity Building International. (2007). EMWater Guide: Improving wastewater treatment and reuse practices in the Mediterranean countries – A Practical Guide for Decision-Makers. InWEnt, Germany.

Chapter 3
Integrated Wastewater Management: A Review

Bassim Abbassi(✉) and Ismail Al Baz

Abstract In this chapter, aspects of wastewater management are explored with integrated perspective. This chapter shows that a holistic view of the entire wastewater system is required for proper wastewater management, starting from the wastewater generation until the ultimate disposal schemes. The functional elements of integrated wastewater management system are generation and composition, collection, treatment (including sludge treatment) and disposal and reuse. A successful wastewater management decision requires a comprehensive, impartial evaluation of centralized and decentralized treatment systems. However, centralized systems should be evaluated based on the investment of the associated collection sewers and their operation and maintenance (O&M). Selecting appropriate technology for wastewater treatment should be based on area-specific integrated factors such as land availability, wastewater quality, desired finished water quality, socio-economic factors and local and provincial regulations.

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3.1. Introduction

The global demand for water is expanding due to population growth, rising standards of living, urbanization, increasing economic activities and expanding areas of irrigated agriculture. Therefore, improved water demand management and development of new water resources are needed. Inadequate wastewater treatment resulted in a serious deterioration of existing water resources as well as seas and oceans in many regions.

The modern wastewater management system takes into consideration the life cycle of the wastewater, from generation until ultimate reuse schemes (Durham et al., 2003). The major functional elements of wastewater management are generation and composition, collection, treatment (including sludge treatment) and disposal and reuse (Abbassi et al., 2000). Proper wastewater management can contribute efficiently in solving the problems associated with the scarce freshwater resources. The relationship between the different functional elements and the water demand represents the integration of wastewater management system (InWEnt, 2006).

The most important criteria for long-term sustainability of wastewater management implementation are affordability (capital and O&M costs), functionality (possibly with locally available staff and support), reliability (e.g., safe effluent for water reuse), environmentally sound (e.g., little sludge production and low energy consumption) and climate suitability (temperature specific).

The overall objective of this chapter is to provide guidance on decision making in wastewater management. The specific aim of integrated wastewater management is to incorporate the selection of cost-effective and efficient wastewater treatment technologies with the ultimate disposal and reuse schemes.

3.2. Generation and Decomposition

Wastewater generation represents the first element in a wastewater management system that ensures a proper design of successive unit operations and unit processes for both collection and treatment. Generally, municipal wastewater stream can be separated into three components, namely black water (water containing feces), yellow water (water containing urine) and grey water (wastewater from washing machines, showers, baths or cleaning and possibly from kitchen sinks). These components differ greatly in composition and concentration of the various constituents.

Streams like black water and kitchen refuses are high in organic load and conversion into biogas via anaerobic treatment appears to be attractive. Black water is of major concern with respect to health risks (pathogens and pharmaceutical residuals). Yellow water contains high amounts of nitrogen and phosphorus and could be used as a source for fertilizer. Grey water can be purified relatively easily and used for several reuse purposes (e.g., flushing toilets, cleaning and irrigation).

Different types of waste stream separation can be carried out. However, for each waste stream or combination of streams, different treatment technologies based on different quality requirements are available (Lesjean and Gnirss 2006; Ramon et al., 2004), which is discussed later in this chapter (Sect. 3.4.1).

3.3. Wastewater Collection and Maintenance

Wastewater collection represents the major element in an integrated wastewater management system and conventionally accounts for 50 to 70% of the total implementation cost of the entire functional elements of wastewater management. Different types of collection systems are available and the choice between them is based on generated wastewater stream characteristic, wastewater treatment system (centralized or decentralized), area-topographical condition and financing affordability. Table 3.1 shows these different sewerage collection types and their advantages and disadvantages.

The sewer maintenance program consists of visual inspections, scheduled sewer cleanings based on maintenance history, unscheduled sewer cleanings as determined by visual or closed circuit inspections and follow-up practices to determine the cause of backups and overflows. Visual inspections are carried out with a mirror attached to a pole; however, the use of portable cameras has been recently introduced to enhance the effectiveness of visual inspections. Older areas of the sewer system are inspected every two years, whereas inspection of relatively new areas may be completed every three to four years (U.S. Environmental Protection Agency [EPA], 1999).

Cleaning is an important part of pipe maintenance. Sewer line cleaning is prioritized based on the age of the pipe and the frequency of the problems within it. Rodding and pressurized cleaning methods are most widely used to maintain the pipes. Bucket machines are rarely used because cleaning by this method tends to be time consuming. However, mechanical cleaning is used rather than chemical methods to remove grease and roots. Introducing chemicals into the cleaning program requires hiring an expert crew, adopting a new program and instituting a detention time to ensure the chemicals' effectiveness. Table 3.2 defines the conditions under which certain cleaning methods are most effective.

Table 3.1 Types of sewerage systems and their characteristics

Type of sewer	Advantages	Disadvantages
Gravity sewer (separate and combined systems)	Low energy requirement	Exfiltration and infiltration possible
Pressurized sewer	Small diameter Little excavation	High energy consumption Exfiltration possible
Vacuum sewer	Small diameter Little excavation No exfiltration	High energy consumption
Solids free sewer	Minimum gradients	Need of septic/settling tank Smaller diameter

Table 3.2 Effectiveness of cleaning techniques (U.S. EPA, 1999)

	Type of problem				
Solution to problem	Emergency stoppages	Grease	Roots	Sand, grit, debris	Odors
Balling		4		4	3
High-velocity cleaning	1	5		4	3
Flushing					2
Sewer scooters		3		3	
Bucket machines, scrapers				2	
Power rodders	4	1	3		
Hand rods	3	1	2		
Chemicals		2	3		3

5 = Most effective solution for a particular problem
1 = Least effective solution for a particular problem

3.4. Wastewater Treatment

Centralized wastewater treatment represents the conventional approach in many countries and is characterized by the collection of sewage and, in some cases, storm water as well. Lower capital and operating costs of one large treatment plant compared to many small-scale plants serving the same urban area is the major advantage of this system. In addition, effective control of effluent quality and plant operation is ensured. Centralized wastewater systems are becoming nonfavorable solutions due to:

- associated high costs of collection system
- soil and groundwater pollution due to undetected sewer leak
- possible entry of variety of recalcitrant and toxic material (heavy metals, endocrine disrupters, pharmaceutical residues, pathogens, etc.) that hinder the reuse schemes
- hydraulic and organic load fluctuating in case of combined collection systems

In recent years, increasing attention has been given to decentralized wastewater systems offering an appropriate solution for low-density communities and sensitive areas (Nhapi, 2004). Decentralized management system employs collection, treatment and disposal/reuse of wastewater from small communities (from individual homes to clusters of houses in a community). Such systems apply a wide spectrum of low-tech solutions (septic tanks or natural systems like ponds or constructed wetlands) and advanced technical solutions (activated sludge, trickling filters, rotating biological contactors, sequence batch reactor etc.).

The basic concept of decentralization of wastewater treatment is to maintain both the solid and the liquid fractions of the wastewater near the point of origin and to minimize the wastewater collection network. Decentralized systems can contribute greatly in wastewater management cost reduction, especially to those costs related to sewerage collection systems. However, concerns related to the decentralized systems are the risk of low effluent quality, especially those that are operated under low-tech solutions.

Generally, centralized and decentralized wastewater treatment using biochemical reaction principles can be broadly classified into high- and low-rate systems. High-rate (intensive) systems imply a treatment in small reactor volume at high microbial concentration. In low-rate (extensive), wastewater is treated at larger reactor volume with lower microbial concentration.

3.4.1. Selection of Appropriate Wastewater Treatment System

The selection of specific unit processes depends not only on the nature of wastewater, including degradability and treatability by selected processes, but also on discharge requirements. Other important factors are environmental impact, land availability, projected life of plant design and cost. All the relevant factors in process selection should be considered, although their relative importance will vary due to social, environmental and political differences as well as technological availability and expertise. Consideration and selection procedure will vary according to whether the treatment plant serves an urban or rural catchments area, or in a developed or non-developed country (Metcalf and Eddy, 1991). Gray (1999) suggested guidelines in the form of algorithms (Figure 3.1) to aid process selection for biological wastewater treatment processes. These were based on yes/no responses and so should be used in conjunction with more detailed accounts of the application and suitability of unit processes, such as those of Metcalf and Eddy (1991).

The choice between high- and low-rate systems for both decentralized and centralized options have implications beyond environmental sustainability, as economical aspects and financial sustainability represent one of the most relevant criteria to be considered. Generally, to select an appropriate treatment technology, a detailed cost–benefit analysis should always be given in terms of capital financing and plant O&M.

The decision about the most appropriate treatment to be adopted, aimed to guarantee the protection of the water bodies and the respect of their own quality, has to be taken on the basis of wastewater origin and characteristics and the type of the ultimate disposal (groundwater, rivers, lakes, sea, transition water bodies). The treatment process to be applied will be defined according to several criteria to be balanced, which include land availability, capital and operational costs and sludge production.

Land availability represents the main criteria to be considered to select an intensive/extensive centralized treatment system. A practice rule can be the following (InWEnt, 2007):

- For land availability of less than 1 m^2/PE, intensive systems will be chosen.
- If available area is higher (up to 5 m^2/PE) mixed systems (biological secondary treatment followed by finishing lagooning, drained vertical sand filters, etc.) can be considered.
- If area occupation is higher than 6 m^2/PE is acceptable, extensive systems shall be applied.

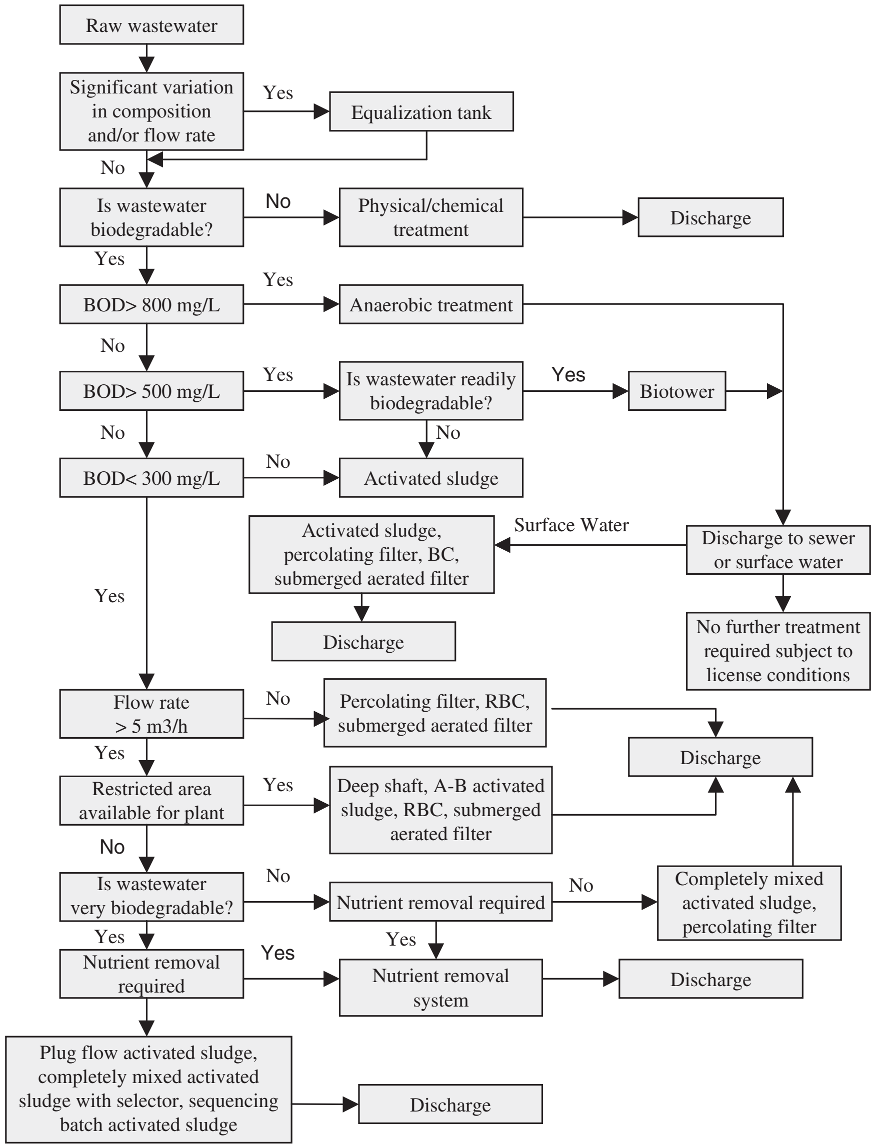

Figure 3.1 Process selection algorithm for biological wastewater treatment (Gray, 1999)

As a general rule, as a process become more complex (intensive systems), the amount of land it requires decreases while total costs and sludge production increase. Selection criteria may additionally involve sludge production and management issues and reuse considerations related to water and nutrients.

Financial aspects represent, indeed, the most effective constrain in the wastewater treatment system selection process. Capital and O&M costs are clearly situation-specific, thus making an absolute assessment very hard to perform. Generally, it is possible to state that extensive systems allow fewer operational costs, especially those required for energy consumption and sludge treatment and disposal. Furthermore, these techniques do not require specialized manpower. Taken as a whole, the use of extensive processes should allow, with identical capacities, a savings of, on average, 20 to 30% on capital costs, and from 40 to 50% on O&M costs, compared to intensive purification systems. Berland and Cooper (2001) calculated the capital and operational costs including energy (€/PE) of small wastewater treatment plants for 1,000 PE in France (Table 3.3). Table 3.4 shows the relative construction and operational costs (€/PE) in Italy. Different types of wastewater treatment systems are taken into account, for a population varying from 100 to 10,000 PE, considering equal to 100 the lowest cost per unit (oxidation pond for 10,000 PE) and expressing other costs as a comparison to that. Reported data include sewer system costs.

Table 3.5 shows a broad line qualitative evaluation of the most relevant parameters that should be taken into consideration during the decisional process. This is, however, a site-specific comparison that should be continuously reviewed based on local conditions.

Table 3.3 Capital and annual operational costs (€/PE) of small wastewater treatment plants for 1000 PE in France

Treatment process	Capital cost	Operational cost
Activated sludge	230 (± 30%)	11.5
Rotating Biological Contactors	220 (± 45%)	7
Imhoff tank + Constructed Wetland	190 (± 35%)	5.5
Biofilters	180 (± 50%)	7
Aerated lagoons	130 (± 50%)	6.5
Waste stabilization ponds	120 (± 60%)	4.5

Table 3.4 Construction and annual operation and maintenance costs (€/PE) of some wastewater treatment systems

	Number of inhabitants (PE)						
Type of system	100	200	500	1000	2000	5,000	10,000
Oxidation pond	1600	1050	610	400	265	150	100
	36.8	27.0	19.0	14.0	10.5	7.5	1.0
Primary settling	1975	1065	800	650	520	390	320
	80.5	68.8	56.0	47.8	41.0	34.0	28.3
Activated sludge	1690	1390	1100	925	765	600	505
	139.5	118.3	95.0	80.3	68.0	54.0	46.0
Biofilters	1625	1345	1050	885	730	575	480
	155.5	122.3	88.5	69.8	54.8	39.5	31.8
Sewer systems	6300	5350	4300	3650	3120	1030	2130
	21.0	18.5	15.0	13.5	12.0	9.8	8.5

Legend : $\frac{\text{Construction costs}}{\text{Operation costs}}$

Table 3.5 Qualitative evaluation of decisional criteria for suitable treatment system selection

Treatment system	Compliance with discharge standards	Area requirement	Sludge production	Energy requirements	Construction costs	O&M costs
Constructed wetlands	G	H	L	L	L	L
Waste stabilization ponds	G	H	L	L	L	L
Aerated lagoons	G	M	L	M	L-M	M
Biofilters	M-G	L	M	M-H	M-H	M-H
RBCs	M-G	L	M	M	H	M
Extended aeration	VG	L	H	VH	M-H	H
Sequence batch reactor	VG	L	H	VH	H	H
UASB systems	M	L	M	L	M	L

VH = very high; H = high; VG = very good; G = good; M = medium; L = low

3.4.2. Sludge Treatment

Sewage sludge (biosolids) is a nutrient-rich organic material resulting from wastewater treatment and processing. The amount of excess sewage sludge production from treatment plants is continuously increasing. Therefore, proper management of excess sludge treatment is becoming increasingly important (Abbassi, 2003). Table 3.6 shows a summary of potential benefits of innovative technologies as compared to established technologies (U.S. EPA, 2006). Criteria used to select the technologies include applicability, judgment about critical assessments needed to promote the technology to the next level of development, promise for further development and current interest.

3.5. Wastewater Reuse

The reuse of treated wastewater can be a valuable alternative to freshwater resources, especially in water-scarce countries. Water reuse should be viewed as one of several alternative sources of new water, all of which will be important tools in the toolkit of the water manager of the 21st century (Miller, 2006). According to Sheikh (1998), water reuse projects are often undervalued when compared to other projects as a result of failure to properly quantify reuse benefits, such as watershed protection, local economic development and public health improvement. Today, technically proven wastewater treatment and purification processes exist to produce water of almost any quality desired. In the planning and implementation process, the intended wastewater reuse applications dictate the extent of wastewater treatment

Table 3.6 Summary of sludge treatment technologies (U.S. EPA, 2006)

Technology and advancement	Low capital cost	Low annual cost	Reduces solids or thickens	Production class "A" biosolids	Reduces odor	Beneficial use (non agriculture)
	Potential benefit as compared to established technologies					
Conditioning						
Established; chemical conditioning and heat conditioning						
Innovative						
Chemical cell destruction		•	•			
Ultrasonic disintegration		•	•			
Thickening						
Established; centrifuge, flotation, gravity built, gravity thickening, and rotary drum						
Innovative						
Flotation thickening – Anoxic gas	•	•	•		•	
Membrane thickening	•	•	•			
Recuperative Thickening	•	•	•			
Stabilization						
Established; aerobic digestion, alkaline stabilization, anaerobic stabilization, composting, pasteurization, solidification and synox						
Innovative						
Aerobic/anoxic	•	•	•			
Anaerobic baffled reactor	•	•	•			
Columbia biosolids flow	•	•	•	•		
High-rate plug flow	•	•	•			
Temperature phased anaerobic digestion	•	•	•	•		
Thermal hydrolysis						
Thermophilic fermentation	•	•	•	•		
Vermicomposting	•	•	•	•		
	•	•	•	•		
Dewatering						
Established; belt filter press, centrifuge, chamber press, drying beds, and vacuum filters						
Innovative						
Quick-dry filter beds	•	•	•			
Electro-dewatering	•	•	•			
Inclined screw press	•	•	•			
Bucher hydraulic press	•	•	•			
Thermal conversion						
Established; fluidized-bed furnace, multiple-hearth furnace and wet air oxidation						
Innovative						
Reheat and oxidize		•	•		•	
Supercritical water oxidation		•	•			
Minergy			•		•	
Drying						
Established; direct drying, flash drying and indirect drying						
Innovative						
Belt drying	•		•	•		•
Direct microwave	•		•	•		•
Fluidized bed drying	•		•	•		•

required or vice versa the available wastewater quality limits possible reuse applications (Salgot et al., 2006).

Wastewater reuse reduces the demand on conventional water resources. In addition, for certain types of reuse, constituents of the wastewater (such as nitrogen and phosphorous) can be used for beneficial purposes. In this case, wastewater treatment costs will reduce as nitrogen and phosphorous do not need to be removed. However, health aspects, groundwater pollution and soil contamination are important constraints to wastewater reuse and should be carefully taken into consideration.

3.5.1. Selecting Appropriate Reuse Applications

For proper selection of appropriate reuse technology, an integrated approach is required, where technological, economical, legal, social, environmental and institutional aspects are considered. While technological and legal issues might be easier to identify and tackle, special attention should be given to market assessment for reuse options and to public acceptance of reuse. The selection process should involve the following steps:

- inventory of potential sources and demand for wastewater
- identification of legal requirements and responsible institutions
- detailed analysis of reuse alternatives
- economic evaluation
- financial feasibility check

The process leads from a very broad assessment of potential supply and demand for wastewater to a more detailed evaluation of related benefits and risks and assessment of costs. Table 3.7 summarizes risks to be considered when selecting

Table 3.7 Categories of municipal wastewater reuse and potential constraints (Metcalf and Eddy, 1991)

Wastewater reuse categories	Risks to consider
Agriculture and landscape irrigation	
Crop irrigation	Surface-and groundwater pollution, if not properly managed
Commercial nurseries	Marketability of crops and public acceptance
Park/school yards	Effect of water quality, particularly salts, on soils, grasses and crops
Freeways (median strips)	Public health concerns related to pathogens (bacteria, viruses and parasites)
Golf courses	
Cemeteries	
Greenbelts	
Residential areas	
Industrial recycling and reuse	
Cooling	Constituents in reclaimed wastewater cause scaling, corrosion, biological growth and fouling
Boiler feed	

(continued)

Table 3.7 (continued)

Wastewater reuse categories	Risks to consider
pathogens in cooling water	Public health concerns, particularly aerosol transmission of
Process water	
Heavy construction	
Groundwater recharge	
Groundwater replenishment	Organic chemicals in reclaimed wastewater and their toxicological effects
Salt water intrusion control	
Subsidence control	Total dissolved solids, nitrates and pathogens in reclaimed water
Recreational/environmental uses	
Lakes and ponds	Health concerns from bacteria and viruses
Marsh enhancement	Eutrophication due to nitrogen and phosphorus in receiving waters
Stream flow augmentation	
Fisheries	Toxicity to aquatic life
Snowmaking	
Nonpotable urban uses	
Fire protection	Public health concerns on pathogens transmitted by aerosols
Air conditioning	Effects of water quality on scaling, corrosion, biological growth and fouling
Toilet flushing	
Potable uses	
Blending in water supply reservoirs	Constituents in reclaimed water, especially trace organic chemicals and their toxicological effects
Pipe-to-pipe water supply	Aesthetics and public acceptance
	Health concerns about pathogen transmission, particularly viruses

reuse applications. Detailed surveys of the local situation will be required to be able to assess actual risks and constraints, and to select the most appropriate technology and applicable risk prevention measures. In the next step, these risks will be compared with the benefits linked with the specific application of wastewater reuse.

Many factors decide on viability of reuse projects because such projects require the establishment or adjustment of existing infrastructure, change in water use habits, etc. In order to decide on viability of reuse projects, a more detailed evaluation of applications should cover suitability of soils and crops, environmental and health risks, need for additional infrastructure and public acceptance of reuse.

3.6. Conclusion

This chapter shows that a holistic view of the entire system is required for a proper wastewater management, starting from the wastewater generation until the ultimate disposal, including reuse schemes. The decision approach for wastewater management should include a comparison between centralized and decentralized treatment systems for the specific area of study. However, the cost of sewerage networks that can make up to 80% of wastewater treatment costs should be considered. O&M costs should be calculated carefully when selecting a wastewater technology. The selection of the appropriate technology should consider the financial and human capacities to properly operate and maintain the facilities. Wastewater quality (e.g., combined, black, yellow and grey water) and wastewater source (domestic or industrial) should be considered. Pathogen, suspended solids and organic matter removal is essential for wastewater reuse. However, nutrient removal (such as nitrogen and phosphorous) is not always necessary when the wastewater is reused in agriculture or aquaculture. Legal standards to regulate wastewater reuse need to be adapted to local conditions.

References

Abbassi B, Dullstein S, and Räbiger N. (2000). Minimization of excess sludge production by increase of oxygen concentration in activated sludge flocs; experimental and theoretical approach. Water Res 34:139–146.

Berland JM and Cooper PF. (2001). Extensive Wastewater Treatment Processes Adapted to Small and Medium Sized Communities (500 to 5000 Population Equivalents). Office of Official Publications of the European Union, Luxembourg.

Durham B, Yoxtheimer D, Alloway C, Diaz C. (2003). Innovative water resource solutions for Islands. Desalination 156:155–161.

InWEnt Capacity Building International. (2007). EMWater Guidelines for Decision Makers in the Field of WWT and Reuse—Short Review (Draft). InWEnt, Germany.

Gray NF. (1999). Water Technology; An Introduction for Environmental Scientists and Engineers. Arnold Publisher, London.

Lesjean B and Gnirss R. (2006). Grey water treatment with membrane biorector operated at low SRT and low HRT. Desalination 199:432–434.

Metcalf and Eddy. (1991). Wastewater Engineering: Treatment and Disposal, third edition. McGraw Hill, New York.

Miller G. (2006). Integrated concept in water reuse: managing global water needs. Desalination 187:65–75.

Nhapi I. (2004). A framework for the decentralised management of wastewater in Zimbabwe. Phys Chem Earth 29:1265–1273.

Ramon G, Green M, Semiat R, Dosoretz C. (2004). Low strength grey water characterization and treatment by direct membrane filtration. Desalination 170:241–250.

Salgot M, Huertas E, Weber S, Dott W, Hollender J (2006) Wastewater reuse and risk: definition of key objectives Desalination 187:

Sheikh B (1998). Accounting for Benefits of Water Reuse AWWAWEF Conference on Water Reuse at Orlando, Florida, pp. 1–4.

U.S. Environmental Protection Agency. (1999). Collection systems, operation and maintenance fact sheet. EPA 832-F-99-031.

U.S. Environmental Protection Agency. (2006). Emerging technologies for biosolids management. EPA 832-R-06-005.

Chapter 4
Egyptian Effluent Standards for Treated Sewage: Evaluation and Recommendations

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Abstract The characteristics of sewage discharged from four Egyptian villages and four Egyptian cities were determined. The results showed that the sewage of the Egyptian villages is a concentrated wastewater with chemical oxygen demand (COD) as high as 1100 mg/L. Moreover, the experiments indicated that the filtered COD after aerobic biodegradability ranged between 50 and 70 mg/L for the sewage of villages and 40 and 60 mg/L for the sewage of the cities. Therefore, it is difficult to achieve the Egyptian effluent standards (EES) for COD (80 mg/L), especially for the sewage of the villages. Accordingly, the EES of the treated sewage are evaluated. Moreover, applicable recommendations for the EES and recommended treatment systems are presented. For the sewage of Egyptian cities, stringent effluent standards (class 1) should be applied, as compared to that of the rural areas and small communities (classes 2, 3 and 4). The EES is proposed to be in classes and phases to guarantee achieving the required effluent standards in short and long term. The required time for reaching the target effluent standards (in the last phase) should be selected based on the development in Egypt.

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4.1. Introduction

In Egypt, more than 90% of the rural areas are not served with wastewater treatment plants (WWTPs). Aerobic systems, like conventional activated sludge, tricking filter and extended aeration, are the most applied systems for sewage treatment in Egypt. Therefore, determining the characteristics of Egyptian sewage is needed for assessing the feasibility of application of aerobic systems in Egypt and for evaluating the Egyptian effluent standards (EES).

Elmitwalli (1992) found that the sewage of an Egyptian village (Meat Mazah) was a concentrated sewage (chemical oxygen demand [COD] > 1000 mg/L) and is more concentrated than that of El-Mansoura city (600 mg/L). He claimed that the high concentration of the sewage in the village was mainly due to discharge of animal manure in the sewer system, not low water consumption, as the village had a sufficient water supply. Ibrahim (1995) evaluated the performance of some Egyptian WWTPs and found that most were not working properly and the effluent COD was higher than that of the EES of 80 mg/L. According to EES, the treated domestic sewage in Egypt should be discharged to drains.

The characteristics of the Egyptian sewage were determined in the present study. Moreover, the EES is evaluated based on the characteristics results and suggested EES for the most important parameters are proposed.

4.2. Material and Methods

The sewage used in the experiments was taken from the sewage pump stations of four villages (Shawa, Meat El-Aamel, Nawasa El-Gheat and Nawasa El-Bahr) and four cities (El-Mansoura, Aga, Samanoud and El-Senblawein). The wastewater from each source was represented by a grab sample, which was taken at mid-day.

For each wastewater source, a 4-L container was filled with 3.5 L of raw sewage. Each container was aerated for 10 days by an air diffuser connected to a small air compressor. For raw sewage samples, COD, biochemical oxygen demand (BOD_5), pH, total PO_4, NH_4, SO_4 and Cl were determined. Every day (except day 5) during the aerobic biodegradability tests, a sample of about 40 mL was withdrawn from each container. For each sample, total COD and filtered (Whatman filter-paper No. 1) COD were measured. All measurements were carried out according to standard methods (APHA, 1998).

4.3. Results and Discussion

4.3.1. Sewage Characteristics

Tables 4.1 and 4.2 show the characteristics of the sewage from the four villages and four cities, respectively. It is clear from these tables that the average COD of the villages was significantly higher than that of the Egyptian cities and represented a

Table 4.1 Characteristics of the sewage for the selected four Egyptian villages

		Village				
Parameter	Unit	Shawa	Meat El-Aamel	Nawasa El-Gheat	Nawasa El-Bahr	Average
BOD_5	mg/L	596	708	454	434	508
Raw COD	mg/L	1,037	1,498	998	922	1,113
Filtered COD	mg/L	403	365	365	307	360
NH_4^+	mg/L	48	48	52	96	61
Total PO_4^{3-}	mg/L	13.0	15.7	13.0	11.7	13.4
SO_4^{2-}	mg/L	79.5	68.9	72.4	81.8	75.7
Cl^- mg/L	mg/L	441	414	306	486	412
pH	—	8	7.7	7.5	8	7.8

Table 4.2 Characteristics of the sewage for the selected four Egyptian cities

		City				
Parameter	Unit	El-Mansoura	Aga	Samanoud	El-Senblawein	Average
BOD_5	mg/L	164	422	392	256	309
Raw COD	mg/L	346	768	653	499	567
Filtered COD	mg/L	77	173	211	211	168
NH_4^+	mg/L	10	51	48	51	40
Total PO_4^{3-}	mg/L	1.95	3.13	11.41	12.06	7.14
SO_4^{2-}	mg/L	55	61	81	62	65
Cl^- mg/L	mg/L	80	405	261	162	237
PH	—	7.5	7.3	7.4	7.5	7.4

Table 4.3 Characteristics of the sewage from literatures

Total COD (mg/L)	SS (mg/L)	VSS (mg/L)	Kj-N (mg/L)	NH_4^+-N (mg/L)	Total P (mg/L)	Reference
500	252	217	—	25	6	Pretorius, 1971
288	118	98	—	33	3	Kobayashi et al., 1983
627	376	297	54	30	10	Barabosa and Sant'Anna, 1989
475	190	155	30	14	25	Sanz and Fdz-Polanco, 1990
160	285	230	43	19	44	Sanz and Fdz-Polanco, 1990
585	321	238	88	53	10	Garuti, 1992
500	150	75	55	—	—	Chudoba and Pannier, 1994
650	217	—	57	45	10	Wang et al., 1995
635	356	249	43	40	—	Eliosov and Argamon, 1995
410	210	195	43	30	7.2	Orhon et al., 1997

very strong sewage. The characteristics of the sewage of the four Egyptian villages were higher than that found in the literature (Table 4.3) and similar to that of the villages of Damietta governorate in Egypt (Ibrahim, 1995). On the other hand, the wastewater of the aforementioned Egyptian cities had similar characteristics to that of Cairo city (El-Gohary and Nasr, 1999; El Monayeri and Smith, 2004) and to that mentioned in the literature (Table 4.3). The particulate COD represents the major part of total COD (>65%) for both Egyptian cities and villages. The latter result was also found by Levine et al. (1985), Odegaard (1998) and Elmitwalli et al.

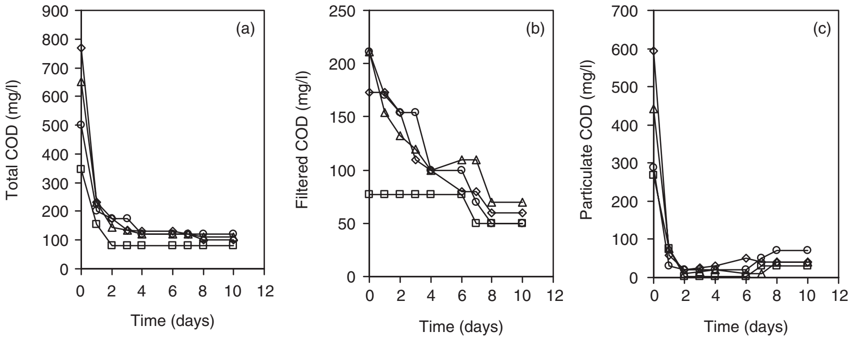

Figure 4.1 Course of total, filtered and particulate COD for the four Egyptian cities during the batch aerobic experiments. (△), Samanoud; (◇), Aga; (○), El-Senblawein; (□), El-Mansoura

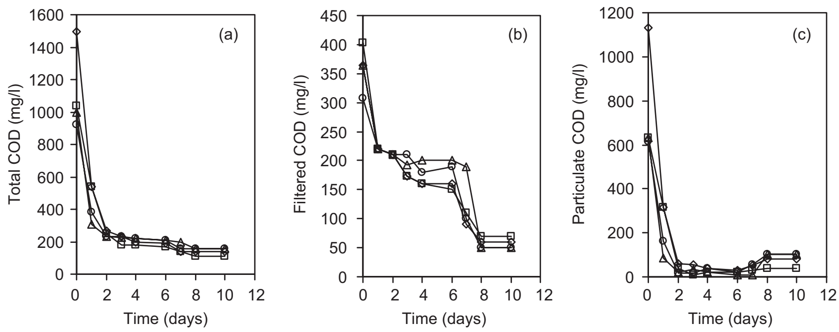

Figure 4.2 Course of total, filtered and particulate COD for the four Egyptian villages during the batch aerobic experiments. (△), Nawsa El-Bahr; (◇), Meat El-Aamel; (○), Nawsa El-Gheat; (□), Shawa

(2000) for the wastewater of the United States, Norway and the Netherlands, respectively.

Figures 4.1 and 4.2 show the course of COD (total, filtered and particulate) in the batch aerobic experiments for the sewage of the selected cities and villages, respectively. Although the soluble COD represented the minor part of Egyptian sewage, the high degradation of this fraction was needed for achieving the EES for COD (80 mg/L). As the particulate COD after batch experiments can be separated by settling (similar to what happens in the secondary sedimentation tank), the filtered COD at the end of the batch experiments can be considered the minimum COD after aerobic treatment. The results showed that the minimum COD after aerobic process was higher for the wastewater of villages (50 to 70 mg/L) than that of the cities (40 to 60 mg/L). Accordingly, it seems that achieving the EES of 80 mg COD/L is not easy to obtain for aerobic treatment of the sewage of Egyptian villages. Moreover, application of high-rate activated sludge or contact aeration process seems to be unsuitable for Egyptian sewage, as these processes have poor removal efficiency for soluble COD.

4.3.2. Evaluation of the Existing ESS

Ibrahim (1995) evaluated the applied different technologies for sewage treatment in the rural areas of Egypt. He found that the effluent of these systems did not comply with EES (Table 4.4) for COD. The same conclusion based on the aerobic biodegradability experiments for Egyptian villages can also be withdrawn. Accordingly, the EES for COD of the treated sewage should be oriented to be achieved, if the WWTPs are sufficiently operated.

Elmitwalli (1992) claimed that the high concentration of the sewage of Meat Mazah village was due to wastes of animal discharge in the sewer system. From visual observation during wastewater collection in this research, accumulated manure was clearly observed in the manholes in the aforementioned Egyptian village. There is no proper method in the Egyptian rural areas to dispose of manure, organic waste and agricultural wastes. This results in many environmental problems, not only in the rural areas, but also in the urban areas, like accumulation of manure and garbage in the sides of the Egyptian canals (main source of water supply in Egypt), frequent clogging of gravity sewers in the rural areas and pollution of ground water. Therefore, for serving any rural areas in Egypt with domestic wastewater collection and treatment facilities, proper systems for collection and treatment of all solid wastes (garbage, manure and agricultural wastes) should be installed at the same time to guarantee a successful operation of all systems.

The effluent standards in most of developed and developing countries are arranged in classes, depending on the discharge. The effect of the effluent on the water quality is also monitored. Table 4.5 presents the main parameters of the effluent standards for different countries.

In comparing the EES of the treated sewage with that of the developed and developing countries, the following can be withdrawn:

1. The EES for treated sewage is a single effluent standard. This means that there is no difference between the effluent standards of a large WWTP with high wastewater flow and a small WWTP for a small community in rural areas having limited wastewater flow. For example, the treated sewage of Cairo city with a population of more than 15 million per capita has the same effluent standard of the treated wastewater of a small village with a population of less than 5000 per capita. Most of effluent standards for both developed and developing countries have different classes for the effluent standards. These classes depend on the

Table 4.4 Egyptian effluent standard according to Law 48 in 1982 for discharging the treated domestic-sewage to drains

Parameter	Unit	Maximum permissible value
BOD_5	mg/L	60
COD	mg/L	80
NO_3-N	mg/L	50
Fecal coliforms	FC/100 mL	5,000

Table 4.5 The main parameters of the effluent standards for different countries

Country	Parameter (mg/L)				WWTP		Receiving water	Time of application	Reference
	COD	BOD	Total N	Total P	Size (PE*)	Type			
Germany	150	40			<1000	Secondary			Jacobsen and Warn, 1999
	110	25			1000–5000	Secondary			Jacobsen and Warn, 1999
	90	20			5000–10000	Secondary			Jacobsen and Warn, 1999
	90	20	18	2	10000–100000	Tertiary			Jacobsen and Warn, 1999
	75	15	18	1	>100000	Tertiary			Jacobsen and Warn, 1999
France	125	25			>2000		Any surface water	31.12.2005	Jacobsen and Warn, 1999
	125	25	15	2	10000–100000		Sensitive water	31.12.1998	Jacobsen and Warn, 1999
	125	25	10	1	>100000		Sensitive water	31.12.1998	Jacobsen and Warn, 1999
Netherlands	125	20	15	2	1800–18000	Tertiary	Any surface water	31.12.1998	Jacobsen and Warn, 1999
	125	20	10	2	18000–90000	Tertiary	Any surface water	31.12.1998	Jacobsen and Warn, 1999
	125	20	10	1	>90000	Tertiary	Any surface water	31.12.1998	Jacobsen and Warn, 1999
Austria	90				50–500	Secondary	Rivers	31.12.2006	Jacobsen and Warn, 1999
	75	20		2	500–5000	Tertiary	Rivers	31.12.2005	Jacobsen and Warn, 1999
	75	20		1	5000–50000	Tertiary	Rivers	31.12.2003	Jacobsen and Warn, 1999
	75	15		1	>50000	Tertiary	Rivers	31.12.2001	Jacobsen and Warn, 1999
	75	15		0.5	>10000	Tertiary	Lakes	31.12.1998	Jacobsen and Warn, 1999
Sweden			15	0.3–0.5	10000–100000				Jansen and Johansen, 1999
			10	0.3–0.5	>100000				Jansen and Johansen, 1999
EU	125	25			2000–10000	Secondary		2000-2005	Jacobsen and Warn, 1999
	125	25	15	2	10000–100000	Tertiary	Sensitive areas	31.12.1998	Jacobsen and Warn, 1999
	125	25	10	1	>100000	Tertiary	Sensitive areas	31.12.1998	Jacobsen and Warn, 1999
Taiwan	250	80			<50 m^3/d				Jou, 2004
	150	50			50–250 m^3/d				Jou, 2004
	100	30			>250 m^3/d				Jou, 2004
Jordan	500	150	100				Irrigation of cooked vegetables, fruit, forestry tree		Al-Mulqui *et al.*, 2002
	200	50	50	15			Irrigation of lawns and parks; ground water recharge; discharge to streams, wadis and reservoirs		Al-Mulqui *et al.*, 2002
	700	250					Irrigation of fodder crops		Al-Mulqui *et al.*, 2002

* PE: population equivalent, which is defined as a BOD_5 load of 60 g/day

served population, the flow of the WWTP and/or the required water quality of the receiving streams.

2. The EES are mainly focused on the removal of BOD, COD and pathogens, not for the nutrients (P and N) removal. Therefore, the secondary treatment will be required. This also applied in most of the developing countries, while nutrient removal is mainly required in developed countries.
3. The EES of the treated sewage are mainly based on the characteristics of the discharged effluent and not on the effect of the treated wastewater on quality of the surface and ground water. According to EES, the treated domestic sewage in Egypt should be discharged only to drains.
4. The water in Egyptian drains and Canals flows from the south to north. Therefore, the coastal areas and the delta of the Nile in the north of Egypt are the most polluted areas. Accordingly, these areas should be considered as sensitive areas for pollution, where more stringent effluent standards should be applied.
5. The maximum permissible value for COD concentration according to the ESS (80 mg COD/L) is higher than that for the secondary effluent in many developed countries (as shown in Table 4.5), especially for the effluent standards of small community in developed countries.
6. The EES for BOD_5 concentration (60 mg/L) is higher than that of developed countries and some of developing countries. A lower value than 60 mg BOD_5/L can be achieved in Egypt, especially if the WWTPs are properly operated. Therefore, the EES for BOD_5 for large WWTP, like Cairo WWTP, should be lowered.
7. The fecal coliform value according to the EES is higher than that of the World Health Organization (WHO) for unrestricted irrigation (1000 FC/100 mL; WHO, 1989). This is mainly because the treated Egyptian effluent should be discharged to drains. However, most of the water in Egyptian drains is reused illegally by farmers or legally by mixing it with canal water in mixing stations at the end of main drains in the north of Egypt (Baza, 2002). Therefore, such value for fecal coliform should be lowered to be 1000 cell/100 mL for large WWTPs.

4.3.3. Recommended Effluent Standards for Treated Domestic Wastewater in Egypt

Most of the effluent standards in many developing countries are copied from that of developed countries, with the attempt to achieve these standards too quickly, without considering the economical and technological capacities (von Sperling and Chernicharo, 2002). Some standards in developing countries are excessively stringent, which leads to increase the distance between desirable and achievable, between law and reality. This results in illegal discharge of wastewater, even without any treatment, to the surface water (like illegal discharge of raw domestic wastewater to the Egyptian canals and drains). To decrease the gap between standards and achievement in developing countries, the effluent standards should be placed to be achieved in a short period and to minimize pollution. This can be done by implementation

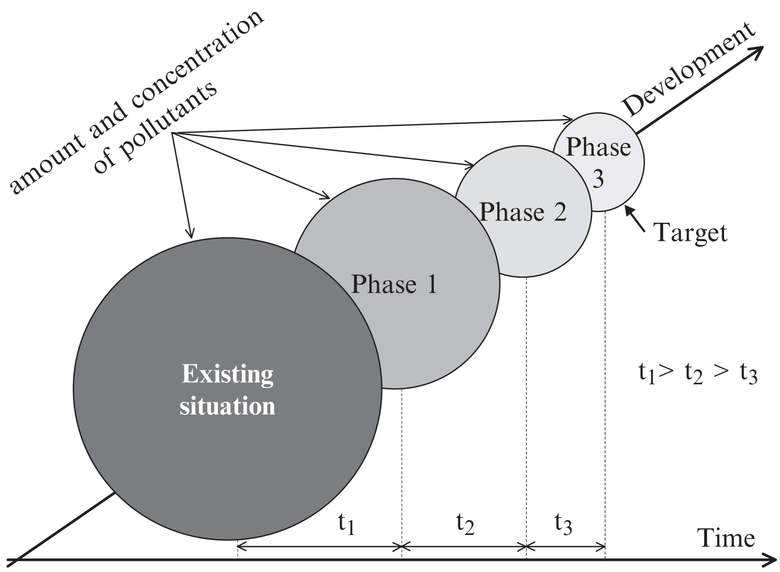

Figure 4.3 Schematic diagram for the application of the stepwise effluent standards to achieve the required target

of standards in stepwise, in phases, to achieve the final target value for the effluent standards in the last phase, as shown in Figure 4.3. The time required for each phase should be parallel and represent the economical, institutional and technological development in the developing countries. Moreover, the effluent standards for treated wastewater should be divided into classes depending on wastewater flow and the required water quality of the receiving water streams.

Based on the effluent standards in many developed and developing countries and characteristics, aerobic biodegradability (this study) and anaerobic biodegradability (Elmitwalli et al., 2002) of Egyptian sewage, the EES of treated domestic wastewater are recommended to be divided into the following four classes in phase 1:

1. **Class 1**: for the effluent of the WWTPs in the cities. The results showed that the domestic wastewater of the Egyptian cities, which represents the major part of the Egyptian domestic wastewater, is less concentrated as compared to that of villages. Therefore, it is possible to achieve a high effluent quality, if these WWTP are operated properly.
2. **Class 2**: for the effluent from the WWTPs in villages and this class can have a lower effluent quality as compared to that in class 1.
3. **Class 3**: for the effluent of on-site treatment systems for remote houses and communities, which will be installed in any area without any sanitation. This means that any new houses or communities should have pre-treatment facilities.
4. **Class 4**: for the existing houses in the rural areas without any sanitation. For this class, a permissible period of 5 to 10 years should be applied before application of such class.

Table 4.6 Recommended maximum permissible concentration for COD, BOD_5 and pathogens for the recommended classes of the EES for treated domestic sewage in phase 1

Parameter	Class 1	Class 2	Class 3	Class 4
COD (mg/L)	100	250	350	350
BOD_5 (mg/L)	40	100	200	200
Pathogen (FC/100 mL)	1,000	50,000*	—	—
Recommended treatment systems	Pre-treatment (primary sedimentation tank or high-rate anaerobic system) followed by aerobic treatment and disinfection	Upflow anaerobic sludge blanket (UASB) + post-treatment, extended aeration, oxidation ditch or ponds	Option 1: (Sedimentation tank, septic tank or anaerobic treatment) + (filter, wetland or pond) Option2**: Separate treatment of black water and grey water	(Sedimentation tank, septic tank or anaerobic treatment) + (filter, wetland or pond)

* When the recommended system is properly operated, the permissible values for pathogen can be achieved as indicated by ASCE (1999) and Elmitwalli et al. (2002). Moreover, it is difficult to apply disinfection in the rural areas, due to high operational costs.
** This can be applied in areas lacking in water resources, like the tourist areas in the Red Sea.

These classes have to be upgraded and modified stepwise, in each phase (as shown in Figure 4.3), until achieving the target effluent standards. In phase 2 and 3, the EES can be focused on, respectively, water quality of the receiving streams in the sensitive areas (like the coastal areas and the delta of the Nile in the north of Egypt) and nutrient removal. The adaptation and implementation of such classes and recommendations in this chapter will result in a control and a reduction of the pollution from domestic wastewater in a short period and will reduce the illegal discharge of untreated wastewater to the canals and drains in Egypt. Table 4.6 presents the recommended maximum permissible concentration (for COD, BOD_5 and pathogens) and treatment processes for the suggested classes of the EES for the treated sewage in phase 1.

4.4. Conclusions

The following conclusions can be withdrawn from this study:

1. The Egyptian sewage of villages represents a very strong sewage with total COD as high as 1100 mg/L, while the sewage of cities was a strong sewage with an average total COD of 570 mg/L. Moreover, the particulate COD represented the major part of total COD (>65%) for both Egyptian cities and villages.
2. The filtered COD after aerobic biodegradability ranged between 50 and 70 mg/L for the sewage of Egyptian villages and 40 and 60 mg/L for cities sewage. Therefore, the EES of COD (80 mg/L) is difficult to achieve for the sewage of the villages.
3. For the sewage of Egyptian cities (which represents the major amount of the Egyptian wastewater), applying more stringent effluent standards (class 1) is recommended, as compared to that of rural areas and small communities (classes 2, 3 and 4). The EES are proposed to be in classes and phases to guarantee achieving the required effluent standards in short and long term. The required time for reaching the target effluent standards (in the last phase) should be selected based on the development in Egypt.

References

Al-Mulqui H, Bataineh F, Malkawi S. (2002). Wastewater Reuse—The Hashemite Kingdom of Jordan. Water Demand Management Forum on Wastewater Reuse. March 26–27 2002, Rabat, Morocco.

American Public Health Association, American Water Works Association, Water Pollution Control Facilities. (1998). Standard Methods for the Examination of Water and Wastewater, 20th edition. Washington, DC.

American Society of Civil Engineering. (1999). Design of Municipal Wastewater Treatment Plants. ASCE Manuals and Reports on Engineering, No 76.

Barabosa R and Sant'Anna G. (1989). Treatment of raw domestic sewage in an UASB reactor. Water Res 23:1483–1490.

Baza M. (2002). Wastewater recycling and reuse in the near east region: experience and issues. Proceeding of the IWA Conference in Water Recycling in the Mediterranean, Greece, pp. 43–59.
Chudoba P and Pannier M. (1994). Nitrification kinetics activated sludge with both suspended and attached biomasses. Water Sci Technol 29:181–184.
El-Gohary F and Nasr F. (1999). Cost-effective pre-treatment of wastewater. Water Sci Technol 39:97–103.
Eliosov B and Argamon Y. (1995). Hydrolysis of particulate organics in activated sludge systems. Water Res 29:155–163.
Elmitwalli T. (1992). Determination of kinetic coefficients for the domestic wastewater in Egypt using activated sludge process. MSc thesis, Mansoura University, El-Mansoura, Egypt.
Elmitwalli T, van Dun M, Bruning H, Zeeman G, Lettinga G. (2000). The role of filter media in removing suspended and colloidal particles in anaerobic reactor treating domestic sewage. Biores Technol 72:235–242.
Elmitwalli T, Zeeman G, Oahn K, Lettinga G. (2002). Treatment of domestic sewage in a two-step system anaerobic filter/anaerobic hybrid reactor at low temperature. Water Res 36:2225–2232.
El Monayeri O and Smith E. (2004). Efficiency Control of Aeration Tanks in Activated Sludge Units: A Case Study. Proceeding of 4th IWA World Water Congress, September 19–24 2004, Marrakech, Morocco, paper ID 116107.
Garuti G, Dohanyos M, Tilche A. (1992). Anaerobic-aerobic combined process for the treatment of sewage with nutrient removal: the ananox process. Water Sci Technol 25:383–394.
Ibrahim GEA. (1995). Comparative study to evaluate different technologies for sewage treatment in rural areas in Egypt. PhD thesis, Alexandria University, Alexandira, Egypt.
Jacobsen BN and Warn T. (1999). Overview and comparison of effluent standards for urban waste water treatment plants in European countries. Eur Water Manage 2:25–39.
Jansen J and Johansen N. (1999). Comparison of procedures for compliance assessment for emission of nitrogen and phosphorous in Sweden, Norway, Finland, Denmark and the EU. Proceeding 6th Nordic Conference on Nitrogen and Biological Phosphorous Removal, February 2–4 1999, Oslo, Norway.
Jou JB. (2004). Environment, irreversibility and optimal effluent standards. Aust J Agric Resour Econ 48:127–158.
Kobayashi H, Stenstrom M, Mah R. (1983). Treatment of low strength domestic wastewater using the anaerobic filter. Water Res 17:903–909.
Levine D, Tchobanaglous G, Asano T. (1985). Characterization of the size distribution of contaminants in wastewater: treatment and reuse implications. J Water Pollut Control Fed 57:805–816.
Odegaard H. (1998). Optimised particle separation in the primary step of wastewater treatment. Water Sci Technol 37:43–53.
Orhon D, Ates E, Sozen S, Cokgor E. (1997). Characterization and COD fractionation of domestic wastewater. Environ Pollut 95:191–204.
Pretorius V. (1971). Anaerobic digestion of raw sewage. Water Res 24:463–469.
von Sperling M and Chernicharo C. (2002). Urban wastewater technologies and the implementation of discharge standards in developing countries. Urban Water 4:105–114.
Sanz I and Fdz-Polanco F. (1990). Low temperature treatment of municipal sewage in anaerobic fluidized bed reactors. Water Res 24:463–469.
Wang K, Zeeman G, Lettinga G. (1995). Alteration in sewage characteristics upon aging. Water Sci Technol 31:191–200.

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Chapter 5
Groundwater Contamination as Affected by Long-Term Sewage Irrigation in Egypt

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Abstract The use of treated sewage water for irrigating the desert sandy soil in Egypt has been practiced in Cairo. Abu-Rawash sewage farm is one that was established in 1944. The farm is irrigated by the flood system. Seepage water beneath the irrigated land is a result of the continuous use of sewage irrigation. Such seepage water, or so-called "groundwater," is the only source for daily domestic use, including cooking, for the farmers. The physical and chemical characteristics of sewage irrigation water as well as fecal coliform and level of heavy metals were studied extensively. The quality of the resulted groundwater was also investigated through 36 samples to investigate level of the level contamination. Results showed that the total dissolved solids (TDS) of groundwater samples vary from 306 to 9,808 mg/L. The minor constituents in these samples include phosphates, nitrates, nitrite, ammonia and sulfide, which exhibited high levels. About 88 and 92% of these samples were over the permissible level in terms of the biochemical oxygen demand and chemical oxygen demand, respectively. For the fecal coliform count, around 60% of the groundwater samples and 45% of the canal water samples were over the permissible limits for drinking water. The suitability of the studied groundwater for human risk consumption was, therefore, evaluated.

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5.1. Introduction

Water scarcity is the case for many developing countries; it is considered a chronic situation mainly due to rapid increasing population (Abdel-Shafy and Aly, 2002). Consequently, exploitation of marginal water resources has become a necessity. Recycled urban wastewater is an important component of marginal resources. Recycled wastewater is used for irrigation, urban and industrial needs as well as indirect potable water supply (Abdel-Shafy and Aly, 2002; Chang et al., 1995). In Mediterranean countries, as a region suffering from water deficit, irrigation needs are seasonal with peak values in late spring and summer. Tourist activity is at its maximum in the same period, entailing an increase of potable water demand (Chang et al., 1995; Bouwer, 1989).

Water budget to Egypt is estimated to be 55.5 BC m^3/y that serve 76 million Egyptians. At present, water income per capita is below 1,000 m^3, and that is estimated to fall below 500 m^3 by the year 2025 (i.e., absolute water scarcity; Abdel-Shafy and Aly, 2002). Therefore, the only option is to increase the water income by using the non-conventional water resources. In Cairo, there are two sewage farms that were established around Cairo sandy soil desert using treated domestic wastewater for irrigation (Abdel-Shafy et al., 2003). One of them is Abu-Rawash farm that was established on 1944. Seepage water beneath the irrigated land is a result of the continuous use of sewage irrigation. Such seepage water, or so-called "groundwater," is the only source for domestic use for the farmers (Abdel-Shafy et al., 2003; Nicholson et al., 2001), who suffer from a shortage of drinking water.

A major issue is the health risks encountered when using the water withdrawn from aquifers recharged with recycled water (Nicholson et al., 2001). Contaminants of concern are microorganisms, nutrients, heavy metals and trace organic pollutants (Abdel-Shafy and Abdel-Sabour, 2006). Related risks depend on the quality and use of the recovered water. Such an issue has already been addressed through health-related regulations and/or guidelines on different uses of water and on reuse of treated wastewater (Abdel-Shafy and Abdel-Sabour, 2006). Many questions have been raised concerning the degree of groundwater contamination in terms of minor constituents, biochemical oxygen demand (BOD), chemical oxygen demand (COD), fecal coliform and heavy metals. The suitability of the groundwater for human consumption should be evaluated.

5.2. Materials and Methods

5.2.1. Geological and Hydro-Geological Settings of Abu-Rawash Area

The investigated area is located at the edge of the Nile river western old flood plain between latitudes 30° 02′ and 30° 07′ N and longitude 31° 02′ and 31° 80′ E. The land surface is flat within the farm, slopes from 20 to 80 m (amsl) towards

Cairo-Alexandria Highway. The major part of the area is covered with Miocene sediments. The unconsolidated sandy soil of sewage farm, partly intercalated with clay, is underlined to the west by sandy limestone of Miocene age. To the east, the area is adjacent to the Quaternary Nile deposits. These deposits comprise sands, gravel and clay that were deposited by the Nile during the development of the Delta. The edge of flood plain is shown as a fault of *horst* and *graben* complex. The fault follows the northwest/southeast trend. The occurrence of Oligocene basalt southwest of the study area is considered more complex tectonic situation. The geomorphology of the western delta region shows a great variation of surface formation. The most predominant are silts, sands, sand dunes, carbonates and sand and gravel.

The studied area located within the Quaternary aquifer exhibits the best conditions in favor of surface recharge. The direction of regional groundwater flow in the western Nile-delta is governed by two factors: Location of recharged area and topographic features. The study area comprises of two aquifers systems. The Quaternary aquifer system is directed to the east, while the tertiary is directed at the west. The Delta Quaternary aquifer is highly productive and consisting of Quaternary graded sand and gravel, intercalated by clay lenses.

There are two main drainage water canals running across Abu-Rawash. The first drainage is non-treated sewage water. The second is mixture of both sewage and agriculture drainage water. The groundwater that resulted from of the two types of wastewater is the only source for domestic use, including drinking by the farmers. Meanwhile, the continuous recharge from both irrigation water and Nile river also contributes in forming such groundwater. The depth of groundwater in the region varies from 15 m in the east to more than 90 m in the west. This indicates that, initially, the feasibility of artificial recharge increases toward the west.

Sampling: An extensive program was conducted for a period of 12 months to collect samples of wastewater, groundwater and Nile canal water for the determination of the physical, chemical characteristics including BOD, COD, and total suspended solids (TSS), according to the American Public Health Association (APHA, 2005). Determinations of the fecal coliform count were also carried out (APHA, 2005).

5.2.1.1. Wastewater Samples

Three different types were collected as follows: treated sewage water that is used for irrigation purposes, Barakat sewage water drain and El–Mouheet drain (mixture of sewage and agricultural drainage water).

5.2.1.2. Groundwater samples

Thirty-six groundwater samples were collected at different depths, ranging from 6 m to 95 m. Of the water samples, 60% were considered shallow groundwater (i.e., <30 m).

5.2.1.3. Surface Water Samples

Eleven canal water samples were collected from five different locations, namely EL Mansouria canal, Kafr Hakim, Beny Magdol, Nahia and Kerdasa canals.

5.2.1.4. Metal determination

A portion of 100 mL from each collected samples was prepared for the determination of trace metals according to (APHA, 2005) using ICP POEMS III Thermo Jarrel Ash Corporation.

5.3. Results and Discussion

5.3.1. Physical and Chemical Characteristics

The physical and chemical characteristics of sewage water that is used for irrigation, groundwater, and the canal water are given in Tables 5.1, 5.2 and 5.3 respectively. These results showed that; the pH values are within the permissible level. The total dissolved solids (TDS) of groundwater samples vary from 306 mg/l to 9808 mg/l in the western part. About 85% of the samples in the investigated area contained > 1500 mg/l reflecting saline water type, 20% of the samples ranged from 1000-1500 mg/l i.e. fresh to brackish water types and 71.5% of the samples are <1000 mg/l indicating fresh water types. While the TDS of canal water samples range from 296 mg/l to 2039 mg/l. All canal water samples (except sample No. 113) are < 1000 mg/l as freshwater type. Meanwhile, the TDS of the sewage water samples ranged from 617 mg/l to 2664 mg/l.

5.3.2. Minor Constituents

5.3.2.1. Phosphates

Tables 5.2 and 5.3 showed that 81% of the groundwater samples exceeded the permissible limits of drinking water (1 mg/L), while canal water samples exhibited higher concentrations. Of the samples, 45% were over permissible limits. On the other hand, all sewage water samples showed high values.

5.3.2.2. Nitrates, Nitrite and Ammonia

The studied groundwater samples exhibited high concentrations, which reflected the impact of sewage irrigation (Tables 5.2 and 5.3). For nitrites and ammonia, both groundwater and canal water samples were within the permissible limits for drinking

Table 5.1 Physical and chemical characteristics of different groundwater samples at Abu Rawash

New Well no.	pH	EC	TDS mg/L	Ca^{++} mg/L	Mg^{++} mg/L	Na^{+} mg/L	HCO_3 mg/L	SO_4^{-2} mg/L	Cl^{-} mg/L
1	6.96	17,180	8,908	707.84	323.4	2,200	201.3	1,100	4,400
3	7.13	2,250	1,249	99	72.17	250	274.5	150	508
10	765	2320	1,322	87.12	43.3	350	237.9	45	650
11	7.31	1642	821	87.12	67.35	130	311.1	100	253
16	7.65	1563	896	55.44	58.69	160	205.87	120	250
18	7.56	715	418	27.72	26.94	80	183	100	55
23	8.41	808	445	15.84	36.08	100	285.48	45	73.5
24	7.3	944	520	39.6	40.89	100	164.7	80	110
26	7.9	909	458	59.4	21.65	85	210.45	55	77
27	7.4	890	475	23.76	28.86	120	323.12	40	82.5
34	7.8	2,290	1302	19.8	67.35	350	237.9	300	400
35	8.76	1,509	866	47.52	26.83	230	335.5	200	122.5
36	8.3	1,898	1100	55.44	19.72	320	488	220	196
37	7.95	1,816	1018	31.68	23.09	320	481.9	120	240.1
41	7.33	1,398	733	47.52	43.3	165	379.72	95	165
43	7.4	1,553	915	51.48	54.84	215	343.12	115	280
44	7.6	1,115	621	59.4	21.65	150	283.65	50	165.8
46	7.25	1,281	674	31.68	40.89	160	384.3	105	110
48	7.28	562	306	19.6	20.65	70	192.15	25	35
49	7.14	1,047	635	55.44	21.65	150	420.9	92.5	55
52	7.5	1,237	631	39.2	42.91	140	384.3	50	110
55	7.4	882	498	31.68	21.65	130	336	43	71.5
56	7.42	723	361	27.72	16.21	88	200.2	22	66
66	7.03	1,574	881	39.6	43.3	220	407.17	145	165
67	7.68	1,880	1,060	71.28	43.3	250	430.05	220	225.5
68	7.81	1,092	616	19.5	15.87	190	339.2	75	110
73	7.24	1,740	846	63.36	57.73	170	398.02	95	236.5
77	7.6	1,764	813	35.64	45.7	200	228.75	125	242
80	7.9	1,532	864	35.64	42.91	220	390.4	160	183.75
81	7.27	843	463	27.72	26.46	115	300.75	38	55
84	7.5	1,755	893	35.64	36.08	255	375.15	95	260
87	7.7	1206	654	39.6	40.89	150	445.5	75	99
91	7.64	1,937	1077	59.4	57.73	250	549	290	100
92	7.18	8.53	452	23.76	24.05	120	283.65	48	71.5
94	7.44	5,000	2615	31.68	137.01	740	558.15	560	820

water (6.5 mg/L), according to the World Health Organization (WHO 1993, 1996; Cotruvo, 1999). For nitrates, all groundwater samples showed high concentration, but only 14% of these samples were over the permissible limits (WHO 1993, 1996; Cotruvo, 1999). On the other hand, all canal water samples were within the permissible limits for drinking water (50 mg/L). As expected, sewage water samples showed high concentration. It is worth mentioning that the Iso contour map of groundwater samples shows that the concentration of nitrate increased toward both the sewage farm and Beny Magdol area, which lies rear the drain El-Mouheet. On the contrary, sewage water samples contained higher concentrations. The average values are 62.67 mg N_2/L for ammonia nitrogen and 101.67 mg N_2/L for the total nitrogen.

Table 5.2 Minor elements constituents of different groundwater samples at Abu Rawash area

New well no.	Site	Total P	NO_3^-	NO_2^-	NH_3	S^-	Total depth
1	Sabih comp	0.524	54.92*	ND	0.4	2.6	95
3	Sabih comp	0.09	63.59*	0.082	0.28	3.2	76
10	Sabih comp.	0.019	45.67	0.026	0.18	1.6	90
11	Al Saliba	0.276	21.36	ND	0.02	1.64	28
16	Al Saliba	0.342	29.94	0.039	0.02	1.44	45
18	Al Saliba	0.228	15.32	0.003	0.22	ND	40
21	Al Saliba	ND	47.85	ND	ND	ND	6
23	Olwea AL Sharkia	0.1	45.32	ND	0.02	2.88	10
24	Olwea AL Sharkia	0.198	47.45	0.01	ND	ND	8
26		0.38	55.15*	ND	0.02	6.5	60
27	Kobry Edris	ND	7.4	ND	0.02	7.6	40
34	Arab Abu Rawash	0.52	45.67	0.024	0.01	8.0	38
35	Arab Abu Rawash	1.88	55.15*	0.043	0.12	7.36	35
36	Arab Abu Rawash	1.516	46.17	0.033	0.14	7.36	30
37	Arab Abu Rawash	1.11	40.7	0.034	0.15	13.12	20
41	Arab Abu Rawash	0.59	46.45*	0.025	0.06	11.5	50
43	Arab Abu Rawash	0.519	47.07	0.014	0.02	11.6	60
44	Arab Abu Rawash	0.502	31.2	0.011	0.02	10.88	25
46		0.214	6.7	ND	ND	7.5	60
48	Kafr Hakim	0.22	2.97	0.017	0.06	ND	12
49		ND	32.4	0.014	0.04	7.68	15
52	Ezbet EL Shemy	0.02	20.7	0.027	0.06	ND	22
55	Kobry El Balah	0.032	0.709	ND	ND	ND	12
56	Kobry El Balah	0.003	5.78	ND	ND	ND	12
66	Abu Rawash	0.03	46.17	ND	ND	ND	22
67	Nahia Road	0.09	2.95	ND	ND	ND	
68	Bany Magdol Road	0.082	11.2	0.006	0.04	5.12	20
73	Nahia	0.05	32.12	0.004	0.04	4.16	18
77	Nahia	0.08	41.5	0.004	ND	4.35	
80	Bany Magdol	0.03	20.7	0.043	0.06	5.12	7
81	Kafr Abu Hmida	ND	1.71	0.003	0.04	5.12	
84	Bany Magdol	0.065	34.56	0.017	ND	11.2	7
87	Kafr Abu Hmida	0.026	36.49	0.001	0.02	6.08	
91	Kerdasa	0.026	20.7	0.008	0.012	6.08	30
92	Kerdasa	ND	9.25	ND	ND	10.56	20
94	Kerdasa	0.214	46.72	0.001	ND	9.28	

ND, below detection limits; *, values over the permissible limits.
According to WHO (1993) and EU (1998), standards limits for drinking water
All groundwater samples are within permissible limits with respect to NO_2,NH_4, S.
For nitrate, about 14% of the groundwater samples exceeds the permissible limits and most of the samples show high concentrations of nitrates.

5.3.2.3. Sulfide

Most of the studied samples showed high levels of sulfide (Tables 5.2 and 5.3). It was found that 75% of the groundwater samples and 36% of canal water samples were over the permissible limits for drinking water (>1 mg/L; WHO 1993, 1996; Cotruvo, 1999; Egyptian Environmental Affairs Agency, 1992), as shown in Table 5.3.

Table 5.3 Minor elements constituents of canal water samples at Abu Rawash area

New well no.	Well no.	Locality	Total P	NO_3	NO_2	NH_3	S^-
1	90	EL Mansoria canal	1.08	7.42	0.076	0.02	ND
2	113	Kafr Hakim canal	1.2	21.9	0.024	0.02	7.2
3	14	EL Mansoria canal	0.26	9.25	0.085	0.02	ND
4	15	EL Mansoria canal	0.368	5.56	0.05	0.02	ND
5	16	EL Mansoria canal	0.058	12.1	0.055	0.02	0.16
6	18	EL Mansoria branch	0.03	21.1	0.131	ND	ND
7	36	EL Mansoria branch	2.66	48.41	0.02	ND	7.6
8	37	EL Mansoria canal	0.19	13.69	0.027	ND	ND
9	41	Bany Magdol	4.3	35.31	0.388	0.26	9.28
10	115	Nahia	0.198	12.1	ND	ND	ND
11	116	Kerdasa	2.115	21.1	0.059	ND	9.6

The levels of sulfide in the canal water samples at Kerdasa, Bany Magdol and at Kafr Hakim canal were 9.6, 9.28 and 7.2 mg/L, respectively, and they are examples of such high sulfide contents in the canal. It is necessary to mention that the Iso sulfide contour map for ground water samples showed the same trend as nitrate.

5.3.2.4. BOD, COD and TSS

About 88% of the groundwater samples exceeded the permissible limits for drinking water, which is an indication of sewage impact. In addition, all canal water samples contained higher than the permissible levels for drinking water. Sewage water samples showed very high values of BOD (Table 5.4).

For COD, the values were very high; 92% of the groundwater samples and all canal water samples were over the permissible limits for drinking water (10 mg/L).

For TSS, only 7% of the groundwater samples were over the permissible limits for drinking water (10 mg/L; WHO 1993, 1996; Cotruvo, 1999; Egyptian Environmental Affairs Agency [EEAA], 1992). For canal water samples, about 18% of the samples exceeded the limits for drinking. On the other hand, TSS for sewage water ranged from 45 to 134 mg/L, which can be considered as group 1 of treated wastewater (EEAA, 1992). It is worth mentioning that there is a strong correlation between organic matter content and fecal coliform counts in this study (Table 5.4), which is a positive indication of the sewage water impact.

5.3.2.5. Fecal Coliform Count

Around 60% of the groundwater samples and 45% of the canal water samples were over the permissible limits for drinking water in terms of fecal coliform (3 cells/100 mL; EEAA, 1992), as shown in Table 5.4. For raw sewage water, the total bacterial count at 22 °C, total the bacterial count at 73 °C, fecal coliform and the fecal streptococci were 2.1×10^{11} count/mL, 9.2×10^{10} MPN/100 mL, 2.2×10^{10} MPN/100 mL and 5.2×10^{8} MPN/100 mL, respectively (Table 5.6).

Table 5.4 Chemical oxygen demand, biochemical oxygen demand, total suspended solids and Coliform count of groundwater at Abu Rawash area

New well no.	Total depth	Sample	COD (mg/L)	BOD (mg/L)	TSS (mg/L)	Coliform count/ 100 mL
3	76	Sabih comp no. 23	116*	73.65*	50*	400
4	205	Sabih comp no. 25	60	41.4	7	<300
7	95	Sabih comp no. 0	100*	63.63*	12	300
11	28	Adel Tamoum	40	24	6.5	<300
18	40	El Saliba	120*	75*	4	900
23	6	Ezbet El Olwia	20	10	6.5	<300
27	60	Kobri Edris	0	0	4.5	<300
32	65	Mohamed Osman	0	0	11.5	<300
33	40	Maher(shahin)	110*	68*	26*	450
34	40	Toblat comp	120*	72*	6.5	1,500
35′	38	Beside the lagoon	103*	64*	1.3	300
35	35	Farmer's houses	80	49	4	300
36	30	Farm mosque	50	30.6	9	300
37	40	Inside treatment plant	80	30.3	4.5	300
38	40	Beside the farm	60	30	4	300
39	Salama 60 m	Beside Brakat sewage drain	30	15.6	3.5	<300
40	Salama 9 m	Beside Brakat sewage drain	60	27.6	4	600
41	20	Beside Brakat sewage drain	60	34.5	7.5	400
42	22	Beside Brakat sewage drain	100*	60*	5.5	400
45	60	Koki comp	60	31.45	11.5	<300
46	25	In front of Mnsoria	40	25	11.5	<300
50	16	In front of Mnsoria	60	37	5.5	<300
55	22	Kobri El Balah	20	12	4	<300
63	12	Ezbet Abu Roash	12	7.4	0.6	400
92	30	Beside Kerdasa drain	10	6	2	700

*, values over the permissible limits.

About 29% of the groundwater samples are not suitable for irrigation with respect to chemical oxygen demand and biochemical oxygen demand.

Only 7% of the groundwater samples are not suitable for irrigation with respect to total suspended solids.

All groundwater samples are suitable for irrigation with respect to fecal coliform except sample no. 34.

5.3.2.6. Trace Constituents

Generally, the level of metals in all the studied samples showed low concentrations (Figure 5.1). However, some samples of groundwater were over the permissible limits for drinking water, as an indication of sewage impact. For canal water, all investigated samples were within the permissible levels, except for Fe, Mn and Ni of a few samples, which again reflects the sewage influence of these particular metals. The Iso contour maps of groundwater shows the same trend of the contamination increase toward both the sewage farm and Beny Maogdol area, which lies rear the drain of El-Mouheet.

Table 5.5 Standard permissible limits WHO, 1993 and EU, 1998. All parameters are in mg/L except pH

Parameter	WHO	EU	Egyptian?
pH	6.5–8.5	Not mentioned	6.5–9.2
Ammonia NH_4	No guideline	0.5	0.5
Nitrate NO_3	50	50	50
Nitrite NO_2	0.5	0.5	0.5
Cadmium Cd	0.003	0.005	0.005
Chromium Cr	0.5	0.5	0.5
Copper Cu	2.0	2.0	2.0
Iron Fe	0.3	0.2	1.0 mg/L for underground water
Lead Pb	0.01	0.01	0.01
Manganese Mn	0.5	0.05	0.5 mg/L for underground water
Nickel Ni	0.02	0.02	0.02
Phosphate PO_4	—	—	1.0
Sulfide S	—	—	1.0
COD	—	—	10.0
BOD	—	—	6.0

Table 5.6 Total count 22 °C, total count 73 °C, fecal coliform and the fecal streptococci in the sewage water (as average values)

Parameter	Unit	Raw	Final
Total count 22 °C	Count/mL	2.1×10^{11}	9.2×10^{5}
Total count 73 °C	MPN/100 mL	9.2×10^{10}	3.5×10^{4}
Fecal coliform	MPN/100 mL	2.2×10^{10}	17×10^{4}
Fecal streptococci	MPN/100 mL	5.2×10^{8}	3.3×10^{3}

5.4. Conclusion

The contamination of the ground water by nitrites, nitrates, ammonia and sulfides is an indication of sewage impact. Meanwhile, the presence of fecal coliform in 60% of the groundwater samples is a strong indication of such contamination. It is worth mentioning that there is a strong correlation between organic matter content and fecal coliform counts in this study ,which is a positive indication of the sewage water impact. Therefore, this groundwater should not be used for potable purposes. Groundwater systems should be protected from any sewage contact, particularly for the site selection of a sewage farm. Therefore, avoiding groundwater jeopardy is a pre-requisite of any sewage reuse. Distinction between potable and non-potable groundwater is essential. The important distinction is also to be made between indirect (surface spreading) and direct (injection wells) recharge.

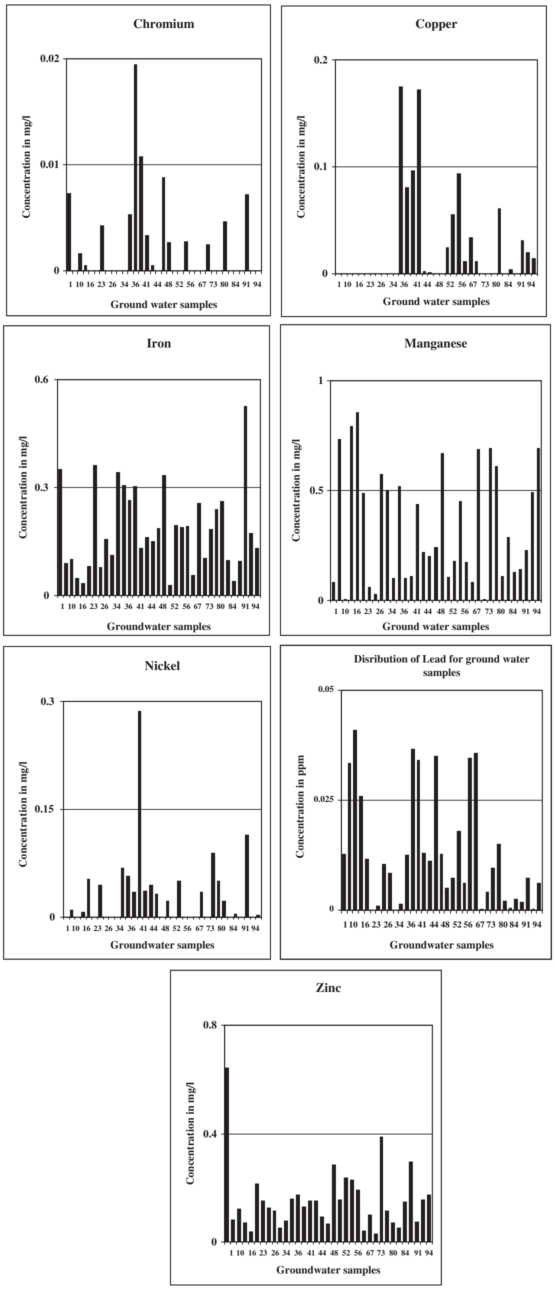

Figure 5.1 Level of metals in all the studied samples showed low concentrations

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References

Abdel-Shafy HI and Abdel-Sabour MF. (2006). Wastewater reuse for irrigation on the desert sandy soil of Egypt: long-term effect. In: Hlavinek et al., eds. Integrated Urban Water Resources Management. Springer Publisher, the Netherlands, pp. 301–312.

Abdel-Shafy HI and Aly RO. (2002). Water issue in Egypt: resources, pollution and protection endeavors. Cent Eur J Occupation Environ Med 8:1–21.

Abdel-Shafy HI, Anwer S, Hassan S, Yahia T. (2003). Risk assessment of sewage reuse on the sandy soil of Abu-Rawash Desert, Egypt. J Environ Protect Engineer 29:5–19.

American Public Health Association (APHA). (2005). Standard Method for the Examination of Water & Wastewater. American Public Health Association, American Water Works Association, Water Pollution Control Facilities, Washington, DC, p. 385.

Bouwer H. (1989). Groundwater recharge with sewage effluent. Wat Sci Tech 23:2099–2108.

Chang AC, Page AL, Asano T. (1995). Developing human health-related chemical guidelines for recycled wastewater and sewage sludge applications in agriculture. World Health Organization, Geneva, Switzerland.

Cotruvo JA. (1999). Guidelines for Drinking Water Quality: Assessment Methodologies. Unpublished report to World Health Organization on a comparative analysis of risk assessment methodologies used for development of drinking water standards and guidelines. WHO, Geneva, Switzerland.

Egyptian Environmental Affairs Agency (EEAA). (1992). Environmental Action Plan, Law 4/1994, Guidelines for Drinking Water Quality. Cairo, Egypt.

Nicholson B, Gibert M, Le Gal C, Vanderzalm J, Banning N. (2001). Management of ongoing and planned water reuse related project in USA and Mediterranean countries. World Health Organization Expert Consultation on Health risks in Aquifer Recharge by Recycled Water, November 8–9, 2001, Budapest.

World Health Organization (WHO). (1993). WHO Guidelines for Drinking Water Quality, vol.1. WHO, Geneva, Switzerland.

World Health Organization (WHO). (1996). WHO Guidelines for Drinking Water Quality, vol. 2. WHO, Geneva, Switzerland.

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Chapter 6
Effluent and Sludge Management in Yemen

J.E. Hall(✉) and R. Ebaid

Abstract Yemen has rapidly declining water resources as a result of over-exploitation of groundwater for crop irrigation and the incidence of water-borne diseases is high due to the low provision of sanitation services. Significant improvements to public health are being made by the construction of sewerage systems and wastewater treatment plants in some towns, but none of the wastewater projects have considered how effluent and sludge should be reused safely, and farmers in particular are highly exposed to risks of infection through uncontrolled reuse practices. This chapter describes the strategy that was developed by MHW Arabtech Jardaneh, in association with GKW Consult, to achieve sustainable reuse of effluent and sludge in Yemen by adopting simple and pragmatic measures that ensure maximum recovery of the agricultural resource value of wastewater while protecting human health within limited financial and institutional resources.

6.1. Introduction

Through the German Financial Co-operation with the Republic of Yemen, Kreditanstalt für Wiederaufbau (KfW) has been financing the design and construction of sewerage systems and wastewater treatment plants (WWTPs) in Yemen for more than 20 years.

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The priority has been to protect public health by avoiding uncontrolled discharges of raw sewage and the resulting contamination of potable water sources as well as crops where raw sewage is used for irrigation. However, scant regard has been given to the management and reuse potential of the treated effluent and sludge that is produced by the WWTPs. Currently, reuse is haphazard as guidance to farmers is not provided and there is no control as the institutional responsibilities are not clearly defined.

To redress the situation, KfW financed a study undertaken by MWH Arabtech Jardaneh in association with GKW Consult (MWH Arabtech Jardaneh and GKW Consult, 2005) to address the general and specific issues arising from WWTPs recently installed in five towns in Yemen (Aden, Amran, Hajjah, Ibb and Yarim). While providing specific solutions for the eight WWTPs serving these towns, the study also provided the strategic, practical framework for the management of effluent and sludge reuse throughout Yemen. This chapter describes the current situation in Yemen and summarizes the principal recommendations of the study.

6.2. The Need for Effluent and Sludge Reuse in Yemen

Yemen is a water-scarce country with a rich natural environment and agricultural diversity due to its varied terrain and climatic conditions. The natural resources are the basis of the national economy but the depletion and degradation of these resources is undermining sustainable development and Yemen is facing a water crisis.

The climatic zones of Yemen can be divided according to the UNESCO classification, based on the ratio between average annual precipitation and annual reference evaporation (E_o). By this classification, Aden is hyper-arid (P/E_o <0.03), Amran and Hajjah arid (P/E_o 0.03<0.25), Yarim semi-arid (P/E_o 0.25<0.50) and Ibb subhumid (P/E_o <0.50<0.75). The boundaries between these zones approximately correspond respectively to the 100, 500 and 700 mm isohyets (Figure 6.1).

The annual renewable water resources in Yemen are estimated at 2.5 billion m^3 (Gun and Ahmed, 1995) but this falls well short of the current annual consumption of 3.4 billion m^3; a deficit approaching 1 billion m^3/year. This is due to the rapid expansion of groundwater exploitation for agricultural irrigation since the 1980s, resulting in aquifers being depleted at a much faster rate than natural recharge. This is exacerbating an already difficult situation for potable water supplies. At the present rate of consumption, fresh groundwater resources may be exhausted in 50 to 100 years in some regions (World Bank, 1997), and as little as 10 years in the Sana'a basin where annual groundwater decline is up to 8 m. Annual per capita water availability in Yemen has progressively fallen from 1,098 m^3 in 1955 to 460 m^3 in 1990 to a current level of 137 m^3, and could fall to 66 m^3 by 2026. Compared with the average for the MENA region of 1,250 m^3/day or the global average of 7,500 m^3/day, the amount of water available per capita in Yemen is among the lowest in the world.

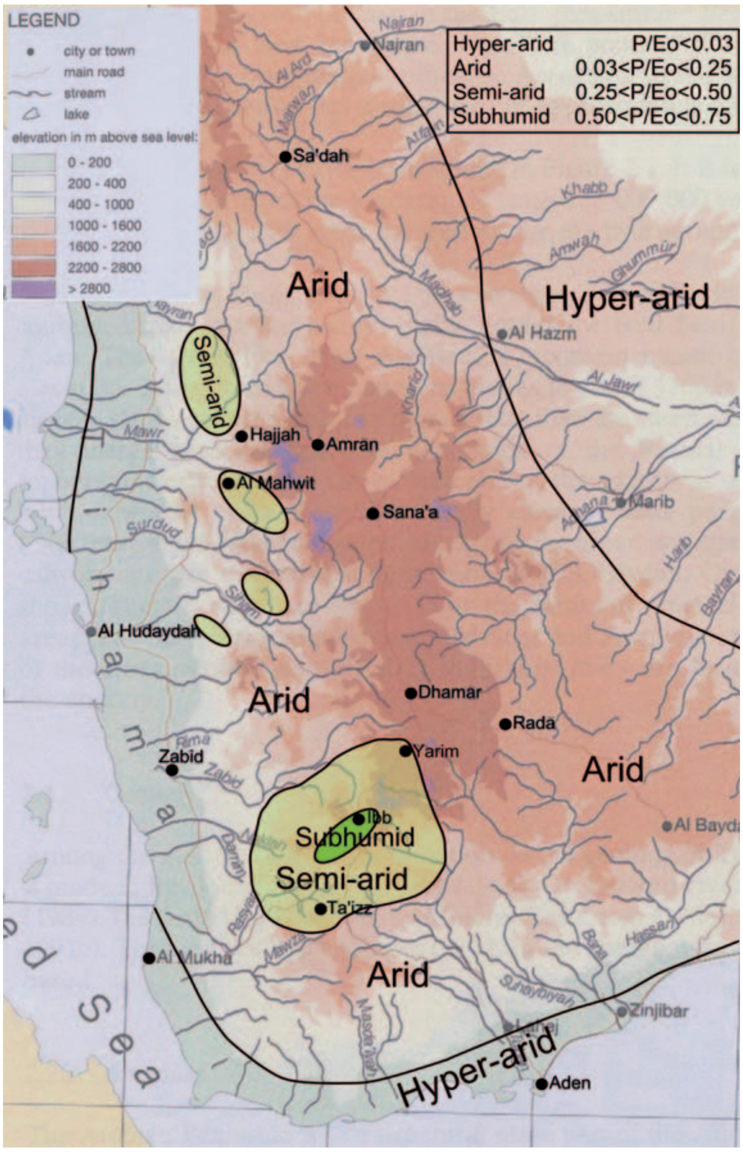

Figure 6.1 Climatic zones of western Yemen

With agriculture consuming the most water (>90%), only 40 L per capita per day remains on average for domestic consumption, well below the level regarded as the minimum necessary for human needs. Public health and clean water supplies are also at risk from the poor provision of sewage collection and treatment systems. Currently, only 57% of the population has public water supply and only 6.2% have sewerage, mostly in the urban centres (Ministry of Water and Environment [MWE], 2004).

These difficult conditions are not going unchallenged. Prior to and since the reunification of Yemen in 1990, there have been numerous development programs in all sectors involving many international donor organisations. Recent important developments in the water and environment sectors have been the adoption of water and environmental laws; the creation of a combined MWE; and the progressive decentralisation of the water supply and sanitation services. There is an ongoing program of rehabilitation and installation of water supply and sanitation systems in the cities and provincial towns, and Millennium Development Goals sets targets of halving the unserved urban population by 2015. A consolidated strategy, action plan and investment program has recently been devised as the National Water Sector Strategy and Investment Program (NWSSIP) for the water sector as a whole, addressing water resource management, urban and rural water supply and sanitation, irrigation and the environment, with an investment program of $1.54 billion for 2005 to 2009 (MWE, 2004).

The provision of wastewater treatment inevitably results in the production of treated effluent and sludge. These are valuable resources for agricultural irrigation and soil fertilization, particularly under the conditions in Yemen. The current water-scarce conditions emphasize the need and urgency of reusing all treated wastewater, and the NWSSIP considers reuse as a means of substituting fresh water resources. This raises new issues for the management and control of effluent and sludge that hitherto have not been fully addressed in Yemen. While it has been stated government policy for some years to reuse effluent for agricultural irrigation, there has been no clear strategy, particularly in relation to institutional responsibilities, appropriate legislation and practical knowledge of the ways and means of realizing the resource value of effluent and sludge in a sustainable, safe manner. Standards for effluent reuse have been adopted in Yemen (Yemen Standardisation, Metrology and Quality Control Organisation, 2001), based on the World Health Organzation (1989) and Food and Agriculture Organization of the United Nations (1992) guidelines, but the institutional and practical means of compliance are absent. So far, the issues of sludge quality and control have been ignored.

There are few laboratories in Yemen capable of analyzing the comprehensive list of parameters required by the effluent reuse standards. Since heavy industries are absent from all catchments, heavy metal concentrations in effluent and sludge are trivial but the major challenge is ensuring that the microbiological quality of effluent and sludge is suitable for the reuse conditions due to the high prevalence of enteric diseases in the population. Most of the WWTPs in Yemen, even the most recent, are unlikely to reliably achieve the microbiological quality standards necessary for unrestricted reuse.

6.3. The Potential Benefits of Effluent and Sludge Reuse

Effluent and sludge must be treated and managed appropriately to avoid potentially adverse impacts on the environment and human health, so that the resource value of using these products can be realized safely. The use of effluent and sludge must also be practicable and economic to ensure operational reliability and affordability, particularly under the difficult conditions in Yemen.

The use of effluent and sludge has a number of direct and indirect benefits under Yemeni conditions. Agriculture can utilize lower quality water than that required for potable purposes, and so effluent reuse for crop and tree irrigation can release precious clean water resources for more sensitive uses. As in many Middle Eastern countries, there is a shortage of animal manure traditionally used to maintain soil fertility. Sludge (and other organic wastes) can have a strategic role in soil management and conservation to enhance soil productivity and help control soil loss through erosion, an important issue in Yemen. Both effluent and sludge contain nutrients that have direct benefits for the farmer by increasing crop yields and cropping intensity. Traditional rain-fed crop production is low yielding and high risk, limiting cropping to the rainy season, but the continuous flow of effluent allows reliable cropping throughout the year. Fertilizer consumption in Yemen is low, applied only to high-value irrigated crops, and agricultural improvements are slow to be adopted, largely due to traditional land tenure conditions. The nutrient content of effluent and sludge can provide a low-cost means of enhancing farm profitability as well as allowing import substitution of fertilizer. Although this will be small at the national scale, this will be important at the local level.

Demand for effluent and sludge is likely to be high where the benefits are realized and this may make it feasible for the water utilities to charge farmers for supplies, thus generating much-needed revenue to cover operating costs. Agricultural diversification and forestry are also feasible and attractive options, providing rural development and employment opportunities.

The agronomic values of treated effluent and sludge are well recognized internationally, and reuse in agriculture is usually the most sustainable option. However, the whole process must be considered holistically, encompassing wastewater quality, treatment and reuse/disposal options, and with a long-term view to potential impacts and overall sustainability. This process also necessarily requires appropriate policies, strategies, legislation and institutional structures with adequate resources for implementation and control, so that the maximum benefits from effluent and sludge reuse can be realized at the lowest cost while protecting the environment and human health.

6.4. Wastewater Production and Treatment

Wastewater problems occur when communities expand to the point that the natural assimilation capacity of the environment cannot deal with traditional methods of disposal. Most towns in Yemen have grown very rapidly since the first Gulf War due to the return of migrant workers and refugees, which resulted in the degradation of water quality from the existing town wells. This situation was exacerbated where piped water systems were installed, resulting in higher water consumption and overloading of cesspits. With the rapid rate of expansion of many towns, earlier sewerage systems were quickly overloaded resulting in the discharge of raw and partially treated sewage creating perennial streams in wadis which were used for uncontrolled crop irrigation.

The first of the existing WWTPs in Yemen was constructed in Aden in the 1970s and in recent years the number of sanitation projects has grown rapidly, mostly financed by World Bank and German (KfW) funds. There are now at least 25 WWTPs in Yemen, either operating or under construction with a total treatment capacity that will reach about 307,000 m^3/day (112 million m^3/year), although further WWTPs are at the design or concept stage (see Table 6.1). The majority of WWTPs are waste stabilization pond systems, which is the most appropriate treatment for the local conditions, and if designed correctly, should produce high-quality effluent suitable for unrestricted reuse. Conventional treatment is provided in Sana'a and Ibb by extended aeration, and in Hajjah by Imhoff tank and percolating filter but these WWTPs produce relatively poor quality effluents; the lack of space precluded the use of pond treatment (see Figure 6.2).

Table 6.1 Wastewater treatment plants in Yemen

Location	Design capacity (m^3/d)	Type of treatment	Date	Funding
Aden (Ash Shaab)	11,000	3 stage stabilization ponds	1970s extended 1989	KfW WB/IDA
Ash Shaab (upgrade)	30,000	3 stage stabilization ponds	Construction	WB/IDA
Aden (Al Arish)	70,000	3 stage stabilization ponds	2002	KfW
Amran	1,480	3 stage stabilization ponds	2002	KfW
Bait El Faqih	2,544	3 stage stabilization ponds	2007	KfW
Bajil	4,151	3 stage stabilization ponds	2007	KfW
Dhamar	11,000	3 stage stabilization ponds	1992	KfW
Hajjah Main	2,428	Imhoff tank/two-stage filter	1998	KfW
Hajjah LS6	724	Imhoff tank	1998	KfW
Hajjah LS8	253	Imhoff tank	1998	KfW
Hodeidah (existing)	12,000	3 stage stabilization ponds	1983	
Hodeidah (upgrade)	51,500	3 stage stabilization ponds	2006	WB/IDA
Ibb (current)	5,200	Activated sludge	1991	KfW
Ibb (upgrade)	10,000	Imhoff tanks/activated sludge	Construction	KfW
Mukalla	14,000	Stabilization ponds	Construction	WB/IDA
Ghail Bawazir	3,600	Stabilization ponds	Construction	WB/IDA
Al Qa'edah	2,650	Imhoff/trickling filter	Construction	WB/IDA
Rada	1,880	2 stage stabilization ponds	1996	KfW
Sa'da	1,449	3 stage stabilization ponds	Designed	KfW
Sana'a	50,000	Activated sludge	2000	AFESD
Seiyun	9,300	Stabilization ponds	Designed	AFESD
Sheher	3,000	3 stage stabilization ponds	Designed	KfW
Taiz	17,500	3 stage stabilization ponds	1982	WB/IDA
Tarim	8,000	Stabilization ponds	Designed	AFESD
Yarim	1,771	3 stage stabilization ponds	2003	KfW
Zabid	1,146	Imhoff tank/two-stage ponds	2005	KfW
Zinjibar	3,880	3 stage stabilization ponds	Designed	KfW

KfW, Kreditanstalt für Wiederaufbau; IDA, International Development Association.
AFESD, Arab Fund for Economic and Social Development.

Figure 6.2 Imhoff Tank and sludge drying beds at Hajjah

While it is easy to be critical of the designs of WWTPs after they have commenced operation, our evaluation of eight WWTPs revealed a number of design weaknesses, believed to be due largely to limitations placed on construction budgets from international donors. Designing WWTPs down to a price will not result in the best long-term investment as the WWTPs may not treat sewage to the necessary standards for reuse, thus requiring subsequent investment to rectify the deficiencies and improve effluent quality, which may not be physically or financially feasible after the WWTP has been constructed. In recent recognition of this and the government's commitment to effluent reuse, any shortfall in investment to ensure effluent quality standards for reuse are achieved will be rectified. The technical competence of the local corporations and at the ministerial level clearly needs to be enhanced so that WWTP designs made by external consultants can be critically reviewed to ensure best long-term value for money and that the required effluent and sludge quality standards for reuse will be achieved.

A consistent shortcoming in WWTP designs is the assumption made on water consumption and sewage strength, resulting in designs that do not adequately balance organic and hydraulic loads. Sewage strengths in Yemen are high due to low per capita water consumption resulting from the introduction of cost-recovery tariffs, which reduced domestic water use. As a result, WWTPs generally exceed their organic loading well before their hydraulic capacity. For existing stabilization pond treatment systems, the simplest and lowest-cost solution is to increase the retention time in the anaerobic ponds (i.e., build additional ponds), as this will enhance overall treatment efficiency by reducing the organic loading of the facultative ponds.

Optimistic assumptions are also made for the fecal coliform (FC) load in the sewage; actual numbers are generally one log greater than assumed in the design due to the high strength of the sewage. This will result in greater FC numbers in the treated effluent compared with the modeling of the design assumptions. Achieving FC counts of less than 1,000 MPN/100 mL in the effluent is the standard necessary for unrestricted reuse and well-designed stabilisation ponds should be able to achieve this but few WWTPs are able to do so in practice. Where space for additional ponds is limited, effluent polishing and disinfection by sand filters and chlorination may be the most practicable means of achieving appropriate effluent quality for unrestricted reuse. Ibb WWTP is the only plant in Yemen where effluent is chlorinated but this is not very effective due to the current overloaded condition of the treatment plant.

Sludge from anaerobic ponds and Imhoff tanks have undergone digestion, but this is not effective at reducing pathogen numbers to sufficiently low levels in the sludge to be safe for manual sludge handling. Most WWTPs are provided with an area for temporary storage of sludge after removal from anaerobic ponds or drying beds, prior to being collected by farmers. However, the storage area design does not usually permit the long-term storage necessary to reduce the usually high pathogen and parasite contents of the sludge to levels that will comply with the proposed standard. Under local conditions, storage for six months with sludge spread in a thin layer (~15 cm) to maximize exposure to solar radiation is regarded as the most practicable and lowest-cost means of achieving hygienic sludge, approaching U.S. Environmental Protection Agency (1993) Class A.

The issues of WWTP design, referred to earlier, concern the final quality of effluent and sludge in relation to their intended outlet. The additional costs of ensuring that reuse standards can be met reliably add little to the overall cost of WWTP construction if they are designed for at the outset. However, the development of effluent and sludge management plans is done after the WWTP is constructed, rather than before. This can lead to either reuse of effluent and sludge that does not comply with the standards, or effluent and sludge that are disposed of in less sustainable ways. For effluent, this would be discharge to the local wadi, where farmers would use the effluent in any case. For sludge, this would most likely be dumped haphazardly as there are no sanitary landfills in Yemen, creating a significant environmental and health hazard.

In designing a new WWTP, it is logical that the most practicable and sustainable options for effluent and sludge management should be identified first, so that the WWTP can be designed to achieve reliably the necessary quality requirements for the identified effluent and sludge outlets. However, in practice, this is rarely happens in a systematic manner. International donors and design consultants should adopt a holistic approach to the development of management strategies and WWTP design in order to achieve the most cost-effective, sustainable treatment and reuse.

6.5. Effluent Reuse Practices and Prospects

With the introduction of sewerage systems, farmers were quick to exploit this new resource, either by diverting new perennial wadi flow or by blocking sewer mains to flood irrigate their land, unconcerned about the health risks to them, their families or consumers of the crops. Farmers (who are often armed) may threaten water utility workers if attempts are made to repair broken sewer mains, such is the value placed on this source of irrigation water. Furthermore, farmers who have experienced the crop yield benefits (and profits) from using raw sewage may be difficult to persuade to use treated effluent.

The potential quantity of effluent produced the design capacities of the current and planned WWTPs could service the irrigation requirement of about 5,600 ha (MWH Arabtech Jardaneh and GKW Consult, 2005). This assumes an irrigation duty of 20,000 m^3/year to meet the average water requirements of two crops grown per year under the generally arid climatic conditions of Yemen and the prevailing inefficient irrigation methods. The potential area that could be irrigated with effluent is equivalent to about 0.3% of the cultivatable land in Yemen, and while the quantity of effluent is very small in relation to overall water requirements, this will be significant within the localities of effluent production.

Following the principle that the simplest and most pragmatic approaches are generally the most achievable and sustainable, the recommended approach for inland WWTPs is to discharge the effluent to the nearest wadi and allow farmers to use the effluent as they wish (unrestricted reuse) but with more control and monitoring of treatment to ensure the necessary standards are achieved.

The irrigation command area of each WWTP is essentially limited to that which can be reached by gravity flow since centrally provided pumping should be avoided to minimize costs and reliability problems. The most accessible land will be alongside the wadi into which the treated effluent is discharged. This is often a narrow strip of land stretching for several kilometers. However, farmers are increasingly using portable pumps to lift water (and effluent) to irrigate higher ground where they have sufficient land and yield potential to justify the costs or can share equipment with neighboring farmers (see Figure 6.3). Nevertheless, formal irrigation schemes should be avoided, as they are expensive to install, operate and monitor, although these may be successful if organized through water user associations (WUAs) since the costs of schemes can be shared and will benefit more users. While the government could part finance schemes as an incentive to conserve groundwater, the operation of the system should be entirely the responsibility of the WUA. The financial commitment of the farmers is considered essential to ensure that they have a sense of "ownership" and a vested interest in the system, thus ensuring its sustainability. Previous initiatives to improve local irrigation supplies through wholly government-owned and operated projects have generally failed as local farmers have not been incentivized by investing in the scheme themselves.

With the exception of one formal effluent reuse scheme (Hodeidah Green Belt), effluent is discharged to the nearest wadi (or to the sea). Wadis normally experience

Figure 6.3 Farmer pumping effluent from wadi to irrigate sorghum

periodic infrequent flows, but effluent discharge inevitably results in perennial flow and this presents both opportunities and risks. Farmers are presented with a reliable source of water for crop irrigation so that they can increase their cropping intensity and income, but they utilize the effluent without any knowledge of its risks. Usually, there are wells near wadis, and wadis are commonly used as roads so there are potentially high indirect and direct exposure risks to local inhabitants. The perennial flow of effluent in wadis may also make the wadis impassable for traffic, an issue commonly disregarded in the design of WWTPs where contractors are required merely to provide "discharge to wadi."

Where there is no previous experience, achieving reuse of treated effluent and sludge depends crucially on farmer acceptance but many farmers are equivocal about reuse, principally due to concern of damage to their land (salinisation). Once there has been some local experience and the benefits are apparent to the local farming community, demand usually increases rapidly. There is a clear need for demonstration field trials to show farmers how to maximize benefits and minimize potential problems such as salinity, safe handling practices to minimize risk of infection to themselves, and the most appropriate crops to grow to avoid risks to consumers. This should be a function of the agricultural extension authority but this service does not operate effectively and their staff is not trained in modern irrigation practices or the reuse of effluent.

There will always be a portion of effluent that cannot be used for crop irrigation due to the seasonality of crop production, even when two or more crops are grown per year. Effluent not reused will eventually infiltrate in the receiving wadi bed to provide indirect aquifer recharge but there is considerable scope for formal aquifer recharge where discharge to wadi is inappropriate and this could be targeted to reduce the rate of decline of groundwater in specific areas. Because most aquifers are deep, the soil cover will provide good filtration and adsorption of pathogens and pollutants in the effluent (with the exception of nitrate), so the impact of groundwater quality should be minimal. Monitoring of farm wells would be necessary as these are often used as potable supplies, as well as for crop irrigation.

The reuse of effluent and sludge for trees (forestry and amenity) is well established in many countries and this is a potentially attractive option for Yemen where natural forests are limited and declining rapidly due to over-exploitation for fuel wood and animal fodder. The green belt scheme in Hodeidah is the only formal "forestry" effluent reuse project in Yemen, designed to control sand dune encroachment of the urban area (Omer 2001), and this has also been considered for Aden. Effluent is used to irrigate urban planting of trees in Sana'a, transported by tanker, and this may be an attractive option in other towns to enhance the urban environment. There has been no development of commercial forestry in Yemen due to limited water resources but this may be a feasible option where effluent is available and agricultural land is limited, as this could stimulate economic development of secondary industries, such as furniture manufacturing, as well as provide fuel wood and fodder.

Our study has shown clearly the problem that sources of effluent do not always coincide with easily accessible land for irrigation and would require investment to maximize direct reuse. For the same reason, it will be difficult to achieve substitution of fresh water by effluent, as envisaged by NWSSIP, since the majority of farms rely on rain-fed crop production. Consequently, effluent reuse is likely to result in the expansion of irrigated agriculture rather than conservation of groundwater.

6.6. Sludge Use Practices and Prospects

Soils in Yemen are characterized by low organic matter and nutrient contents, resulting in low levels of agricultural production. Manure is used to supplement soil fertility, usually from the farmer's own animals but there is an active market for manure that supplies larger farmers. Fertilizer consumption in Yemen is low and restricted exclusively to nitrogen (urea) applied only to high value irrigated crops: phosphorus and potassium fertilizers are virtually unknown. Due to the uncertainty of rainfall and consequential financial risk, fertilizer is not applied to rain-fed crops.

Under these conditions, sludge offers a cheap, effective alternative to manure and fertilizer to provide organic matter and nutrients, at the cost of the farmer loading and transporting sludge from the WWTP. Currently, arrangements between the WWTP and farmers are ad hoc, with no control or recording of user and the land to which the sludge is applied.

It is estimated that the total sludge production in Yemen may exceed 60,000 tds/y when the current and planned WWTPs are operating at capacity (MWH Arabtech

Jardaneh and GKW Consult, 2005). About 15,000 ha (1.4% of total cultivatable area of Yemen) would be required annually to use this quantity of sludge, assuming an annual rate of application rate to land is 4 tds/ha. While this area is small in relation to the overall cultivatable area of Yemen, sludge is produced in only a few locations and so will require a significant proportion of the land locally. Unlike effluent reuse, where the area of reuse is usually constrained to land immediately downstream of the WWTP, the principal limitation for sludge reuse is transport distance (i.e., the cost to the farmer of collecting sludge from the WWTP). While the main sludge reuse area would normally be expected to be within a few kilometers of the WWTP, there are exceptions; for instance farmers from Tihama and Amran have taken sludge from Hajjah WWTP, a distance of more than 50 km, as they see a value in sludge in excess of the transport cost.

For the small stabilization pond WWTPs, finding sufficient farmers in the locality to take the sludge is unlikely to be a problem, but for the major sludge production centers of Sana'a, Ibb, Aden and Hodeidah, this could be much more difficult as the WWTP would rely on a much larger proportion of the farming community to take sludge. For Aden and Hodeidah, this is made more difficult with the WWTPs being located on the coast with only limited agricultural land nearby.

The sludge storage facilities on the WWTPs are limited, and in some cases, WWTP designs made no provision. Consequently, there is anxiety about sludge handling, storage and disposal, particularly because stabilization pond systems are desludged only periodically. Clearly, sufficient storage must be allowed for in the design but also, the demand by farmers must be encouraged through demonstration trials, marketing and provision of appropriate agronomic advice.

All sludges currently produced in Yemen are air-dried, either *in situ* in anaerobic ponds or on drying beds. However, at Ibb, which experiences the highest rainfall in Yemen, the WWTP has a seasonal problem due to low demand during the wet summer season when air drying sludge is difficult, and currently exacerbated by the overloaded condition of the WWTP. Sludge accumulates to the extent that the WWTP is obliged occasionally to discharge liquid sludge to the wadi. The current proposal is to install mechanical dewatering, which should alleviate the immediate problem but could result in more difficult issues. Mechanical dewatering is not only expensive (recurrent costs for polymer and power), but also the physical quality of the sludge (sticky) will be unattractive to farmers, being impossible to handle and spread on the land manually. This will also increase potential health risks to the farm laborers (see Figure 6.4). Expanding the drying beds and improving their efficiency (e.g., solar drying, reed beds, etc.), with additional sludge storage space, are considered more sustainable and lower-cost options.

The principal concern of sludge reuse is the high exposure risk of WWTP and farm laborers handling sludge and the potential for acquiring infection. Farmers can be adequately protected by long-term storage of sludge to reduce the pathogen load to acceptable levels for manual spreading on the land. However, WWTPs are highly exposed as sludge is lifted from drying beds by hand, and ensuring workers take elementary precautions (i.e., gloves, boots, personal hygiene) has proved difficult, resulting in high levels of infection and absenteeism.

Figure 6.4 Manual lifting of sludge from drying beds at Ibb wastewater treatment plant

Sewage treatment results in most of the sewage pollutants being retained in the sludge, but with the limited quantity of industrial effluents discharged to sewer, heavy metal concentrations in sludges are very low and typical of domestic catchments. However, the long-term environmental concern is that sludge use on land results in accumulation of heavy metals in soil. Modeling of potential accumulation, based on the natural background concentrations of heavy metals in the soil, indicates that it would take at least 200 years of regular use before precautionary limit values for protecting soil quality would be approached. Consequently, heavy metals in sludge are not an immediate environmental or health concern, with the benefit of reducing the need for frequent and costly monitoring programs.

6.7. Strategy Development

International experience shows that the key to successful, sustainable effluent and sludge reuse programs is to control the potential risks to human health and the environment, and to create and maintain farmer and public confidence in the effectiveness, safety and benefits of wastewater treatment and effluent and sludge reuse.

While there is a general acceptance in Yemen that the reuse of effluent (and by implication, sludge) is an integral component of water resource and pollution control policies, there is no cohesive implementation strategy. There is no clear definition of the institutional responsibilities or the mechanisms for implementing and controlling reuse projects.

Regulations on the treatment of wastewater and effluent and sludge reuse are widely adopted internationally, based on extensive scientific research and adapted to local conditions. The standards applied by different countries generally reflect the actual and perceived risks and the level of precaution deemed necessary to protect health and the environment. For Yemen, the appropriateness and practicability of standards need to be addressed, but finance is required to enhance WWTPs to ensure that compliance with the standards for safe reuse can be achieved in practice.

The steps considered necessary for Yemen to approach the required levels of safety, control and operational security for sustainable reuse of effluent and sludge, include:

- Clearly defined reuse strategies, in particular how effluent reuse can be integrated into the emerging water resource management strategy (e.g., NWSSIP).
- Clearly defined institutional responsibilities for effluent and sludge management at central government and local levels. Coordination mechanisms between the relevant ministries are required at the national level and forums need to be created at the local level at which all of the stakeholders are represented, with farmer representation through WUAs.
- Specific regulations on effluent and sludge qualities and monitoring requirements for different reuse conditions. International quality standards adapted to the local conditions should make compliance feasible and encourage monitoring and control. The process should be transparent and auditable to provide confidence to all of the stakeholders involved.
- Ensuring that WWTPs (stabilization ponds) are designed and operated to comply with effluent quality standards for unrestricted reuse and safe discharge to the environment.
- Provision of high-quality central laboratories to analyze expensive and infrequently required parameters, such as heavy metals, and provide quality checks on local laboratories.
- Technical guidelines on the management of effluent and sludge reuse programs, and the provision of appropriate agricultural advice for users.
- Promotion of beneficial reuse of effluent and sludge, and the safe practices that should be observed. This is best achieved by a program of demonstration trials and a community-based approach through WUAs.

References

Food and Agriculture Organization of the United Nations (FAO). (1992). Wastewater treatment and use in agriculture. Irrigation and Drainage Paper 47. FAO, Rome.

MWH Arabtech Jardaneh and GKW Consult. (2005). Feasibility study for effluent and sludge reuse in Aden, Amran, Hajjah, Ibb and Yarim. Project report to Kreditanstalt für Wiederaufbau and Ministry of Water and Environment, Yemen.

Gun JAM van der and Ahmed AA. (1995). The water resources of Yemen. A summary of available information. Report WRAY-35. Ministry of Oil and Mineral Resources, General Department of Hydrology, and The Netherlands Embassy, Sana'a.
Ministry of Water and Environment. (2004). National Water Sector Strategy and Investment Program, MWE, Yemen.
Omer MA. (2001). Evaluation of Hodeidah sewage water quality used for irrigating greenbelt. Tihama Environment Protection Project, Tihama Development Authority, Yemen.
U.S. Environment Protection Agency. (1993). Part 503—Standards for the Use or Disposal of Sewage Sludge. Fed Reg 58:9387–9404.
World Bank. (1997). Towards a water strategy. An agenda for action. Report No. 15718-YE. WB, Yemen.
World Health Organization (WHO). (1989). Health guidelines for the use of wastewater in agriculture and aquaculture. Technical Report No. 778. WHO, Geneva.
Yemen Standardisation, Metrology and Quality Control Organisation. (2001). Irrigation Water No. 150/2001. YSMQCO, Yemen.

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Chapter 7
Fate of Pathogens In Tomato Plants and Soil Irrigated With Secondary Treated Wastewater

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Abstract Pathogenic indicators, namely total coliform, *E. coli* and *Enterococcus* were measured on harvested tomato fruits and leaves and in soil irrigated with fresh water, effluent of extended aeration wastewater treatment plant and effluent of upflow anaerobic sludge blanket-rotating biological contactors integrated pilot treatment system. Plantation was taking place in a greenhouse during summer in Jordan. A drip irrigation system was applied in which laterals were covered with mulch to minimize contact between irrigation water and plants. Results showed that total coliform and *Enterococcus* counts in all tomato fruit samples (except one) and *E. coli* count in all harvested tomato fruit samples were less than 1 MPN/g dry plant. Although secondary treated wastewater had indicator pathogenic counts of 2 to 5 log units, a considerable reduction was noticed in the collected soil samples after 10 days of the last irrigation. All soil samples contained less than 1 MPN/g dry soil of *E. coli*, while total coliform counts ranged from less than 1 to 19.23 MPN/g dry soil. The results suggest that disinfection of the reclaimed wastewater may not be necessary with respect to the measured indicator pathogens when proper agricultural practices are applied.

7.1. Introduction

Wastewater reuse for agricultural production is a common practice in several countries. However, there is a growing concern about the related environmental and health impacts of wastewater reuse. In the case of domestic wastewater, emphasis

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is given to the control of microbiological contamination and many developing countries adopted the World Health Organization (WHO) guidelines (1989) with respect to the maximum allowable limits of indicator pathogens in the effluent of treatment plants. The guidelines were applied as national standards in many cases and treated as absolute values. However, the production of "standard effluent" becomes irrational when irrigation techniques and agricultural practices are used to act as barriers for pathogenic contaminants as suggested by the water chain approach (San'a expert group meeting on municipal wastewater use for irrigation, 2006). In this approach, irrigation technique is of crucial importance for the selected wastewater treatment system. For example, using sprinklers as irrigation technique for raw fresh crops requires advanced systems for pathogens removal; however, suitable application of drip irrigation may require poor water quality with less purified effluents and sophisticated treatment plants. In other words, the same quality of crops can be obtained with lower quality water if proper irrigation techniques are used.

The integrated approach combines both risk assessment and risk management as introduced by Stockholm expert meeting (Fewtrell and Bartram, 2001). In fact, the newly suggested WHO guidelines (2006) do not require pathogens removal to occur solely at the wastewater treatment plant. Natural die off, farming practices, applied irrigation systems and produce washings are considered very effective in reducing pathogens to acceptable, safe limits. This concept is in consistent with the sanitation coverage target adopted by the World Summit on Sustainable Development in Johannesburg in 2002. Risk management is of particular importance even when the treated effluent is disinfected. There is evidence that the disinfection process, especially when chlorine is used, is not effective in preventing microorganism regrowth downstream of the treatment plant (Gantzer et al., 2001; Tchobanoglous et al., 2003) particularly when nutrients are available (Rajkowski and Rice, 1999). When this is the case, health risks associated with reclaimed water reuse is still present and agricultural practices become crucial in determining the safe usage of this water resource. There are indeed only few publications investigating the fate of pathogens in soil and plants when irrigating with either disinfected treated effluent or treated wastewater that is not subjected to disinfection.

According to Jordanian standards for reclaimed wastewater reuse in agriculture 893/2006, treated wastewater cannot be used for irrigating crops that are eaten raw. However, when risk management options are taken into account, it could be possible to use primary or secondary treated wastewater for irrigating different crops, including those eaten raw. The main objective of this research is to investigate the fate of pathogens in tomato plants and soils irrigated with different qualities of wastewater under Jordanian conditions, provided that precautions are taken into account when considering agricultural practices. The latter includes the application of a drip irrigation system that is covered with black mulch to prevent direct contact between the plants and treated irrigation wastewater.

7.2. Materials and Methods

The experiment was performed in a 500 m^2 greenhouse built 60 m downstream of the domestic Abu-Nusier wastewater treatment plant located north of Amman. Half of the greenhouse was utilized for growing tomato with the following three different water treatments (Figure 7.1):

1. Effluent of the existing activated sludge treatment plant. Irrigation water was taken before chlorination.
2. Effluent of a pilot integrated anaerobic-aerobic system consisting of a combined upflow anaerobic sludge blanket (UASB) reactor followed by -rotating biological contactors (RBCs).
3. Fresh water as a control.

The pressurized irrigation system consisted of a pressure regulator at the inlet, a disc filter, sand filter and fertilizer injection unit at the storage tanks. A main line from each storage tank carries the water to the greenhouse. Every plot in the greenhouse contains two laterals from the mains. Every lateral contains five drippers supplying 4.3 L/hour. The discharge at the beginning of the mains is 258 L/hour for each type of wastewater, while it was 21.5 m for each lateral. Plantation took place at the second half of the green house. The field was ploughed twice before starting the experiment. The planting area was divided into blocks with paths 0.5 m between blocks. The plots were randomly distributed for every water quality. The area subjected to irrigation water was covered with mulch (as shown in Figure 7.2) to prevent direct contact between irrigation water and the plants. This system is also common in Jordan, especially in the Jordan valley, where it is mainly used to prevent evaporation.

Tomato plant seedlings were prepared in a nursery before planting in May 2003. Tomato's growing season is around 90 days and two fruits pickings were performed at the end of the season. The soil was irrigated for six hours one day before plantation and for 30 minutes the same day of plantation. After tomato was planted, soil was irrigated for extra two hours. After two weeks, the plants were irrigated weekly for two hours. After three more weeks, plants were irrigated two times per week for

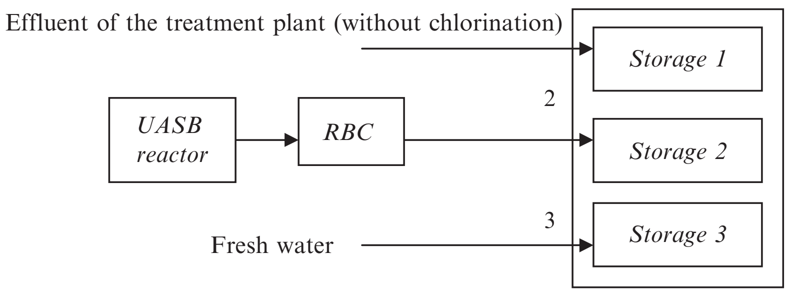

Figure 7.1 Schematic representation of the presumed experiment

Figure 7.2 Laterals covered with mulch to reduce direct contact of irrigation water with plants

two hours until the end of the growing season. Irrigation requirements were calculated according to the pan evaporation measurements carried out at the greenhouse through the whole experimental period.

Soil samples were collected after five and 10 days of the last irrigation, respectively. Samples were taken at four depths: 0 to 15 cm, 15 to 30 cm, 30 to 45 cm and 45 to 60 cm. Water samples were collected prior to each irrigation and leaves and fruits of tomato plants were collected at the end of the experiment. Plants and soil samples were collected using zip bags and transferred directly to the laboratory for analysis. Wastewater samples were collected using sterile bottles that included sodium thiosulfate and transferred directly to the lab for analysis. All collected samples were analysed for total coliform, *E. coli, Enterococus* and *Salmonella*. A certain weight of sample (soil or plant) was diluted to 100 mL using 0.85% NaCl as described by Mackie and McCartney (1989). Pathogens were then measured following the APHA (1995) enzyme substrate test. For wastewater samples, measurements followed the APHA (1995).

7.3. Results and Discussion

The measured average total coliform, *E. coli* and *Enterocoсuss* in the water used in treatments are shown in Table 7.1. It should be mentioned that no *Salmonella* was detected in all analyzed water samples. The measured values of biological indicators

Table 7.1 Biological indicators measured for the three treatments

Parameter	WWTP × 10^5	UASB-RBC × 10^5	FW
Total coliform (MPN/100 mL)	4.8	68.69	<1
E. coli (MPV/100 mL)	1.5	31.77	<1
Enterococous (MPN/100 mL)	0.02	0.008	

RBC, rotating biological contactor; UASB, upflow anaerobic sludge blanket; WWTP, wastewater treatment plant.

Table 7.2 Biological indicators in tomato crop (leaves and fruits) for all treatments*

	T. Coliform (MPN/g dry plant)		
	FW	WWTP	UASB-RBC
Leaves	<1	<1	<1
Fruits	<1	<1 (71.57)	<1
	E. Coli (MPN/g dry plant)		
Leaves	<1	<1	<1
Fruits	<1	<1	<1
	Enterococous (MPN/g dry plant)		
Leaves	<1	<1 (24.29)	<1 (2.3×10^3)
Fruits	<1	<1	<1 (20.45)

* Only one reading had the value between brackets.

on tomato fruits, leaves and the soils are shown in Tables 7.2 and 7.3, respectively. It is very interesting to note that fruits (Figure 7.3) in all cases (except one) did not contain more than 1 MPN/g dry plant of each biological indicator, although wastewaters (Table 7.1) contained high pathogenic load. According to WHO guidelines (1989), wastewater with qualities reported in Table 7.1 should not be used for unrestricted irrigation to prevent health risks associated with wastewater reuse. However, the results obtained in this study show that when irrigation techniques and agricultural practices are taken into account, health risks are minimized at lower treatment costs. The results obtained in this study were better than those obtained by Manios et al. (2005), who concluded that some pathogenic contamination on the surface of tomato and cucumber occurred due to indirect move of pathogens through insects, which were present in the greenhouse. Mulch cover used in this experiment did minimize the contact between plants and wastewater. However, one measurement for the total coliform showed a value of 72 MPN/g dry plant while one measurement for the *Enterococcus* had a value of 21 MPN/g dry plant. According to the WHO suggested guidelines (2006), produce washing would result in 1 log unit reduction in pathogens, which means that washing tomato before consumption will guarantee a safe product.

Indicator pathogens counts in different soil layers for all treatments are shown in Table 7.3. With reference to the measured values of total coliform in FW (Table 7.1), and the counts measured in the soil five days after last irrigation, it can be noticed that soil cannot be considered free of this indicator pathogen. Ruffete et al. (2006)

Table 7.3 Biological indicator concentrations for soil samples taken at different depths for all applied treatments

	5 days after the last irrigation			10 days after last irrigation		
	Total Coliform /g dry soil			Total Coliform (MPN/ g dry soil)		
Depth	FW	WWTP	UASB-RBC	FW	WWTP	UASB-RBC
0–15	339	8.95	70.45	15.48	19.23	<1
15–30	1.19×10^3	4.36	19.60	<1	<1	<1
30–45	5.67	1.98	199	<1	2.74	<1
45–60	38.90	133	33.38	<1	3.82	<1
	E.coli (MPN/g dry soil)			E.coli (MPN/g dry soil)		
Depth	FW	WWTP	UASB-RBC	FW	WWTP	UASB-RBC
0–15	<1	1.499	24.08	<1	<1	<1
15–30	3.13	1.195	6.92	<1	<1	<1
30–45	<1	<1	<1	<1	<1	<1
45–60	6.69	<1	2.74	<1	<1	<1
	Enterococcus (MPN/g soil)			Enterococcus (MPN/g dry soil)		
Depth	FW	WWTP	UASB-RBC	FW	WWTP	UASB-RBC
0–15	1.28×10^3	96.74	891	22.39	14.93	8.79
15–30	513	42.53	715	14.43	13.44	23.36
30–45	57.98	40.04	36.52	1.17	20.02	7.24
45–60	22.96	43.11	1.31×10^3	0	7.04	8.11

Figure 7.3 Tomato plants just before harvesting

measured a total coliform count in soil without additional treatment around 10^4 cfu/g soil dry matter, which indicates that soils may originally contain background total coliforms. Regrowth of indicator pathogens in soil was reported by Gibbs et al. (1997) and high humidity was found to be among factors that help in growth of enteric bacteria (Entry et al., 2000; Rufete et al., 2006). The count of total coliforms decreased with time and soil dryness as shown after 10 days of irrigation cessation (Table 7.3). Although, total coliform was present in soil, transmission to tomato plants can be ignored as discussed before.

E. coli did not survive in soil after irrigation. *E. coli* count decreased considerably with maximum values of 24 MPN/g dry soil reported when UASB-RBC effluent was used for irrigation (Table 7.3). After 10 days of terminating the experiment, *E. coli* count did not exceed 1 MPN/g dry soil in all analyzed soil samples. Short survival time for *E. coli* was also reported by other researchers (Entry et al., 2005) who did not detect *E. coli* in any soil sample after the first day of diary manure and compost application to soil. With respect to *Enterococcus*, there was a 2 to 3 log unit reduction in soil samples taken at all depths after five days of the last irrigation compared with counts present in irrigation wastewater. Although *Enterococcus* counts were traced after 10 days of the last irrigation in all soil layers (except one), there was a considerable reduction in their concentration when comparing measurements taken for soil samples after five and 10 days of the last irrigation. The survival of these pathogens in soil layers is expected to decrease with time; however, exact time needed for their disappearance was not measured.

7.4. General Discussion

A preliminary model established for the assessment of risk of infection and disease associated with wastewater irrigation of vegetables eaten uncooked (Shuval et al., 1997) showed that irrigating with wastewater effluent that meets WHO guidelines (1989) with respect to fecal coliform would provide a factor of safety of 1 to 2 orders of magnitude greater than that used by U.S. Environmental Protection Agency for microbial accepted standards for drinking water. With respect to the quality of surface water used for unrestricted irrigation, it is common to find lenient standards compared with those implemented for irrigation using reclaimed wastewater (Carr et al., 2004). Indeed public health protection is a major concern; however, wastewater is a resource that should be fully exploited and irrational standards will limit the use of this resource. In many water-stressed countries, violations are reported and farmers use partially or untreated wastewater for irrigation (Raschid-Sally et al., 2005). Moreover, protection of public health cannot be achieved solely at the treatment plant, as regrowth of pathogens has been reported, especially when chlorine is used for disinfection (Gantzer et al., 2001; Tchobanoglous et al., 2003). Instead, health protection can be achieved by using a "multiple barriers" approach (Carr et al., 2004) that interrupts the flow of pathogens to human. In this approach, soil, wastewater treatment, irrigation technique and human exposure control are all important in determining the fate of pathogens in wastewater. At the same time, the cost of wastewater treatment is reduced as soil and crops serve as biofilters (Haruvy, 1997). This will be of special interest in countries with limited financial resources, as it will encourage the implementation of lower cost wastewater treatment technologies and maximize reclaimed water reuse.

7.5. Conclusions

Pathogenic indicators measured on tomato plants and in soil irrigated with treated wastewater from an extended aeration treatment plant and a pilot plant consisting of UASB-RBC integrated system showed that total coliform and *Enterococcus* counts in all tomato fruit samples (except one) and *E. coli* count in all harvested tomato fruit samples were less than 1MPN/g dry plant. Although secondary treated wastewater had indicator pathogenic counts of 2 to 5 log units, a considerable reduction was noticed in soil samples collected five and 10 days, respectively, after the last irrigation. After 10 days of the last irrigation, all soil samples collected from all depths contained less than 1 MPN/g dry soil of *E. coli*, while total coliform counts ranged from less than 1 to 19.23 MPN/g dry soil. With respect to the measured indicator pathogens, the results suggest that disinfection of reclaimed wastewater may not be necessary when proper agricultural practices are applied downstream of the treatment plant.

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References

Carr RM, Blumenthal UJ, Duncan Mara D. (2004). Guidelines for the safe use of wastewater in agriculture: revisiting WHO guidelines. Water Sci Technol 50:31–38.

Choi C, Song I, Stine S, Pimentel J, Gerba C. (2004). Role of irrigation wastewater reuse: comparison of subsurface irrigation and furrow irrigation. Water Sci Technol 50:61–68.

Entry JA, Hubbard RK, Thies JE, Furhmann JJ. (2000). The influence of vegetation in riparian filterstrips on coliform bacteria I. Movement and survival in surface flow and groundwater. J Environ Qual 29:1206–1214.

Entry JA, Leytem AB, Verwey S. (2005). Influence of solid diary manure and compost with and without alum on survival of indicator bacteria in soil and on potato. Environ Pollut 138:212–218.

Fewtrell L and Bartram J, eds. (2001). Water Quality: Guidelines, Standards and Health Assessment of Risk and Risk Management for Water Related Infectious Disease. London, IWA publishing on behalf of the World Health Organization.

Gantzer C, Gillerman L, Kuznetsov M, Oron G. (2001). Adsorption and survival of feacal coliforms, somatic coliphages and F-specific RNA phages in soil irrigated with wastewater. Water Sci Technol 43:117–124.

Gibbs RA, Hu CJ, Ho GE, Unkovich I. (1997). Regrowth of fecal coliform and salmonellae in stored biosolids and soil amended with biosolids. Water Sci Technol 35:269–275.

Haruvy N. (1997). Agricultural reuse of wastewater: nation-wide cost-benefit analysis. Agric Ecosys Environ 66:113–119.

Mackie and McCartney. (1989). Practical Medical Microbiology, 13th edition.

Manios T, Papagrigoriou I, Daskalakis G, Sampathianakis I. (2005). Evaluation of primary and secondary treated and disinfected wastewater irrigation of tomato and cucumber plants under greenhouse conditions, regarding growth and safety considerations. Proceedings to the First International Conference on Sustainable Urban Wastewater Treatment and Reuse, September 15–16, Nicosia-Cyprus.

Rajkowski KT and Rice EW. (1999). Recovery and survival of Escherichia coli O157:H7 in reconditioned pork-processing wastewater. Food Prot 62:731–734.

Raschid-Sally L, Carr R, Buechler S. (2005). Managing wastewater agriculture to improve livelihood and environmental quality in poor countries. Irrig Drain 54:S11–S22.

Rufete B, Perez-Murcia MD, Perez-Espinosa A, Moral R, Moreno-Caselles J, Paredes C. (2006). Total and fecal coliform bacteria persistente in a pig slurry atended soil. Livestock Sci 102:211–215.

Shuval H, Lampert Y, Fattal B. (1997). Development of risk assessment approach for evaluating wastewater reuse standards for agriculture. Water Sci Technol 35:15–20.

Tchobanoglous G, Burton F, Stensel H. (2003). Wastewater Engineering: Treatment, Disposal and Reuse, fourth edition. Metcalff & Eddy, Inc.

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

World Health Organization (WHO). (2006). WHO Guidelines for the Safe Use of Wastewater, Excreta and Greywater. Volume II: Wastewater Use in Agriculture. WHO, Geneva.

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Chapter 8
Alleviation of Salinity Stress Imposed on Broad Bean (*Vicia faba*) Plants Irrigated With Reclaimed Wastewater Mixed With Brackish Water Through Exogenous Application of Jasmonic Acid

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Abstract Farmers in Palestine suffer from a continuous shortage of water due to its scarcity. It is important to note that both brackish water and reclaimed wastewater represent major sources, although both resources are problematic, as they impose stress to growing plants. Consequently, alleviation of these stresses is required, particularly salt stress, imposed by the use of brackish water or reclaimed water. The aim of this study is to search for the means to alleviate stress through irrigation with reclaimed wastewater mixed with brackish water (mix). Jasmonic acid (JA), a plant growth regulator, proved to be efficient in alleviating various types of stresses, such as chill and drought stress. JA was tested in this study to determine whether

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it would alleviate salt stress imposed through irrigation of broad bean (*Vicia faba*) plants by a mix of reclaimed wastewater and brackish water (Electrical conductivity [EC] = 7 dS/m). Broad beans plants are considered sensitive to salinity. Results showed that treating plants with JA lessened, although only slightly, the negative impact of mix. Moreover, applying treated wastewater using drip irrigation in addition to cultivating plants in pots prevented the contamination of fruits with the pathogens. Treating plants with JA enhanced the plant's tolerance to stress conditions imposed through irrigation of plants with alternative water resources.

8.1. Introduction

Palestine is located in southwest Asia on the eastern shore of the Mediterranean, in the center of the Middle East. There are two distinct climatic seasons in Palestine: a wet winter and a dry summer. The rainy season extends from mid-November to the end of April, with an average annual rainfall in the Palestinian Territories (PT) of approximately 450 mm. However, temperatures are relatively high: January is the coldest month with average temperatures between 8 and 12 °C, while August is the warmest month, with temperatures ranging between 22 and 34 °C. The Jordan River system is the only surface water resource in the PT, but there are two aquifers: the Mountain Aquifer underlying the West Bank, and the Coastal Aquifer underlying the Gaza Strip. The problems related to water that Palestine presently faces are abundant and varied. Palestine and Jordan, as well as most other Middle Eastern countries, are generally characterized as arid and lacking in water resources. The PT are expected to experience a serious water deficit in the year 2020, the shortage of which will be 271×10^6 m^3. Numerous studies and plans for expanding water resources exist, including desalination and water transfers from other basins. However, in most cases, these plans are expensive and face daunting logistical and political barriers (Mimi et al., 2003).

In recognition of the scarcity of water and inevitable population growth in the region, it has become vital to conserve existing water. The supply and management of water resources and wastewater remain a key priority for the PT. The wastewater-related problems in the environment have been documented as continuously increasing due to the increasing discharge of wastewater as a result of the increasing demand of fresh water. These environmental problems include the gradual increase in nitrates of both groundwater wells and some freshwater springs (Ministry of Planning and International Cooperation, 1998). The use of treated wastewater in the PT to meet increasing agricultural water demands has been identified as one of the main objectives of the Palestinian water sector. The total volume of treated urban wastewater suitable for reuse is projected to be 12.1×10^6 m^3/year for the main Palestinian cities by the year 2010. In comparison, total agricultural water demand is projected to increase by 50×10^6 m^3 over the years 2006 to 2010 (Meerbach, 2004). Due to the current political climate, an increase in the fresh water supply is not a viable option. Therefore, water reuse is the key to agricultural development. Various studies have been conducted to assess the feasibility of reuse at several locations in the West Bank, but implementation of a comprehensive water reuse project is still pending the approval and construction of wastewater treatment plants.

From the agronomic side, several factors and conditions restrict the use of treated wastewater in agriculture, the most important of which are crop type, irrigation system and socio-cultural factors. However, some of the potential hazards of using reclaimed wastewater for plants are salinity, specific ion or element toxicity, direct injury to leaves (Johnson and Parnell, 1998), nitrogen overdose (Feigin et al., 1991) and water stress. Consequently, using reclaimed wastewater forced horticulturists to adopt various techniques and means to deal with these hazards. Among these techniques are the leaching of accumulated salts by over-irrigation and mixing water of different sources to attain a lower irrigation water salinity level, application of Ca-source (e.g., $CaSO_4$) to counteract the sodicity and treating plants with growth regulators (e.g. Jasmonic acid [JA]) to lessen the negative impact of salinity.

The main goal of this research is to investigate the suitability of using reclaimed wastewater mixed with brackish water (mix) as alternative water resources in irrigating agricultural crops in the Palestinian highlands, combined with the use of the natural growth regulator (JA) to alleviate the negative impact of reclaimed wastewater and/or brackish water.

8.2. Materials and Methods

8.2.1. Location

The experiments were conducted at the wastewater treatment plant in Al-Bireh City. The treatment plant processes approximately 1.25×10^6 m^3/year of raw municipal wastewater. It consists of oxidation ditches and secondary clarifiers. The reclaimed water has a tested quality of 10/10 mg/L biochemical oxygen demand/total suspended solids (BOD/TSS; BOD <10 mg/L, TSS <10 mg/L), 30 to 40 mg/L total nitrogen and less than 100 CFU/100 mL fecal coliform level. Because there are no adjacent agricultural lands, the effluent is not normally reused and is discharged to the Wadis.

8.2.2. Planting Material

Broad bean seeds (*Vicia Faba*) cv. Primarence were planted in 12-L pots filled with soil mixture composed of peat moss, sand and clay in 2:1:1 ratio (by volume). Each pot was fertilized before planting with 10 g of 14:7:28 ($N:P_2O_5:K_2O$) starter fertilizer. Plants were distributed in the greenhouse and divided to three groups. Water was applied to all plants using drip irrigation. Plants in group one were irrigated with reclaimed water (RW), plants in the group two were irrigated with mix and plants in group three were irrigated with fresh water. The electrical conductivity (EC) of the mix was 1.5 dS/m at the start of treatment period, and increased gradually to 7.0 mmohs.cm^{-1} by the end of the growing season. During the first three weeks of the growing season, all plants were irrigated with freshwater three times per week until plants reached a height of 15 cm. After that, each pot received the designated treatment. Each plant was fertilized with 2 g of 14:7:28 ($N:P_2O_5:K_2O$) at weekly intervals, and the quantity of water applied was increased gradually in proportion to plant grown and climatic conditions, and reached 2.5 L by the end of experiment.

8.2.3. JA Treatments

Within each group, there were four JA treatments (0.0, 0.5, 1.0 and 1.5 mM). JA solutions, containing a few drops of Tween-20, were applied exogenously to plants three times during the plant growing season: the first time was 42 days from planting, the second application was at the beginning of flowering (after 70 days from planting) and the final application was 120 days from planting. Each plant was sprayed with 0.25 L at the first application time and with 0.5 L at the following application times. JA was synthesized from methyl Jasmonate (MJ), according to Farmer and Rayan (1992). In brief, JA was prepared by dehydration of MJ under mild alkaline conditions, by mixing 1.5 mL MJ with 15 mL MeOH, 450 µL H_2O, and 1.5 g K_2CO_3. The mixture was incubated at 60 °C for 45 minutes in a sealed vial, then added to 90 mL water. This aqueous mixture was extracted four times with 45 mL pentane to remove MJ, and the aqueous phase was then titrated to pH 4.5 with 2.0 M HCl and extracted four times with 60 mL diethyl ether. The diethyl ether was removed by rotary evaporation, leaving JA. After that, JA was mixed with 3 mL acetone and the volume was increased to 2 L with distilled water, which provided a stock solution with a concentration of 1.5 parts per million (ppm) JA. The designated solutions were prepared through dilution with distilled water. Accordingly, there were nine treatments with five replicates (total 50 pots), and the experimental design was randomized complete block design (RCBD).

8.2.4. Assessment of Vegetative and Reproductive Parameters and Mineral Composition of Plants

To assess the impact of the treatments on vegetative and reproductive growth, the plant height, number of leaves, and number of branches were recorded weekly, whereas fruit number and weight were recorded more than once per week. For mineral composition determination, 20 leaves were taken randomly from each plant after 110 days of planting. Ten leaves were taken from the upper plant and 10 leaves from the lower part and then shock-frozen in liquid nitrogen. Tissues were then stored at −10 °C until required. Total nitrogen was determined according to the Kjeldahl method based on the ICARDA Manual (Ryan et al., 2001), and total phosphorus, total potassium and total calcium (Ca) were determined by using dry–ash procedure bases (Ryan et al., 2001). Phosphorus readings of samples, standards and blanks were taken at 410 nm using the spectrophotometer, whereas readings for K and Ca were taken at 768 and 620 nm, respectively, using a flame photometer.

8.2.5. Visual Inspection

Plants were inspected visually for yellowing, wilting and salt injuries after 80, 120 and 135 days of planting. These parameters were visually evaluated on a scale of 1 to 5, in which 1 indicates no salt injuries, no wilting symptoms and greener plants, and 5 indicates extreme wilting and yellowing plants.

8.2.6. Assessment of Soil Properties

Soil samples were tested at the end of the growing season for the following properties: pH value, which was determined for soil paste extract using an HI 9017 microprocessor pH meter; EC, using an TH-2400 auto-ranging EC meter; and organic matter, which was estimated by digesting soil with concentrated sulfuric acid, further addition of potassium dichromate in the presence of concentrated phosphoric acid, and finally titration with ferrous sulfate.

8.2.7. Assessment of Fruit Contamination

Harvested fruits were tested for fecal coliform, total coliform, *Salmonella* and *Staphylococcus aureus*. Determination of fecal coliform and total coliform on products was conducted according to the Bacteriological Analytical Manual (Bennett and Lancette, 1998). Samples were prepared by adding 50 g of test sample to 200 mL peptone water, and then blended in a stomacher for one minute at medium speed. To test fecal coliform, samples were diluted in eosin methylene blue medium by the spread plate technique, and incubated for 18 to 24 hours at 44.5 °C. To test total coliform, samples were diluted on violet red bile lactose on medium speed and incubated for 24 hours at 35 °C. *Salmonella* were detected by adding 25 g of product to 225 mL peptone water, further blended in a stomacher for one minute at medium speed and incubated for 16 hours at 37 °C. *Salmonella* was then isolated by adding 10 mL of culture (peptone water) to 100 mL of selenite cystine medium and incubating for 24 hours at 37 °C.

8.2.8. *Statistical Analysis*

All results were subjected to analysis of variance using the CoStat-software (CoHort Software, Monterey, CA). The mean separations were calculated by Duncan's multiple range test at $P \leq 0.05$.

8.3. Results

8.3.1. *Impact on Vegetative and Reproductive Growth*

Treating broad bean plants, which were irrigated with either reclaimed wastewater or mix, with 0.5 mM JA resulted in significantly more branching compared to the plants treated with the highest level of JA (1.5 mM) and irrigated with reclaimed wastewater only (Table 8.1). Concerning plant height, it is evident that JA treatments have a retarding effect, in particular with plants irrigated with reclaimed wastewater or mix and having received the highest level of JA (1.5 mM). Plants that received the higher JA treatments formed fewer leaves compared to plants irrigated with fresh water and no JA treatment. Concerning fruit number, no significant differences were registered, though fruit weight differs significantly upon treatment with JA and the irrigation source. Fruits from plants irrigated with mix and treated with 0.5 mM JA were significantly lighter than control fruits.

Table 8.1 Effects of Jasmonic acid and reclaimed wastewater treatments on the branching, plant height, fruit number, and fruit weight of bread bean plants*

Treatments**	Number of branches 101 days after planting (DAP)	Plant height (cm) 107 DAP	Number of leaves 101 DAP	Number of fruit (whole season)	Total fruit weight (g; whole season)
JA 0.0 mM WWS	4.6 ab	75.1 ab	51.6 ab	31.3 a	246.8 a
JA 0.5 mM WWS	6.1 b	74.4 ab	57.6 ab	47.3 a	428.9 abc
JA 1.0 mM WWS	4.6 ab	72.4 a	41.6 a	42.3 a	418.6 abc
JA 1.5 mM WWS	4.6 ab	66.0 a	45.3 a	41.3 a	418.1 abc
JA 0.0 mM WW	5.6 ab	77.1 ab	58.6 ab	44.6 a	419.8 abc
JA 0.5 mM WW	6.0 b	81.5 ab	58.5 ab	33.0 a	297.4 ab
JA 1.0 mM WW	5.0 ab	76.4 ab	52.8 ab	45.3 a	498.4 c
JA 1.5 mM WW	4.1 a	71.0 a	43.3 a	43.3 a	534.2 c
Control	5.1 ab	89.5 b	62.8 b	44.3 a	474.1 bc

* Means within each column marked with different letters are significantly different ($p \leq 0.05$, Student-Newman-Keuls range test).

** JA, Jasmonic acid; WWS, reclaimed wastewater + brackish water "mix"; WW, reclaimed wastewater.

8.3.2. Impact on Salinity Stress Symptoms

Although plants were monitored visually three times (80, 120 and 135 days after planting), the results shown in Table 8.2 are for two times, due to similarities between the last inspection times. Three stress symptoms were monitored over the entire growing season, namely yellowing, salt injuries and wilting. It is obvious that treating plants with JA resulted in significant preservation of the green leaf color. Concerning salt injuries, only the higher JA levels were sufficient to alleviate salt injury imposed by both irrigation sources. Treating plants with JA resulted in reducing wilting symptoms on plants. It is important to note that the degree of stress symptoms was larger in mix-exposed plants, regardless of JA concentration, compared to controls.

8.3.3. Impact on Mineral Composition of Leaves and Soil Characteristics

The effect of various treatments on mineral composition of leaves, both old and young, is presented on Table 8.3. Concerning nitrogen (N) levels, no significant differences were registered between most treatments. However, the promotion effect of JA treatments on N-level can be seen, in particular with the higher JA levels (1 and 1.5 mM). With potassium (K), irrigation of plants with mix resulted in lower K-levels of old leaves, in most treatments, although no clear trend can be observed with young leaves. Furthermore, it is obvious that JA treatments did not have any influence. With both phosphorus (P) and Ca, no clear trends can be found.

Table 8.2 Effects of Jasmonic acid and reclaimed wastewater treatments on the yellowing, salt injuries, and wilting of bread bean leaves*

	Yellowing		Salt injury		Wilting	
Treatments**	94 days after planting (DAP)	134 DAP	94 DAP	134 DAP	94 DAP	134 DAP
JA 0.0 mM WWS	1.0 a	5.0 c	0.3 a	5.0 c	0.6 a	5.0 d
JA 0.5 mM WWS	1.0 a	4.3 bc	2.6 b	4.0 bc	2.0 a	4.3 cd
JA 1.0 mM WWS	1.0 a	2.3 ab	0.0 a	2.0 ab	0.0 a	2.0 abc
JA 1.5 mM WWS	1.0 a	2.3 ab	1.3 ab	1.6 ab	0.6 a	2.0 abc
JA 0.0 mM WW	1.0 a	5.0 c	0.0 a	5.0 c	0.0 a	5.0 d
JA 0.5 mM WW	1.3 a	3.6 bc	0.3 a	3.0 abc	1.3 a	3.0 bcd
JA 1.0 mM WW	1.0 a	2.0 ab	0.3 a	1.3 ab	0.6 a	2.0 abc
JA 1.5 mM WW	1.0 a	1.0 a	0.0 a	0.0 a	0.0 a	0.0 a
Control	1.0 a	2.0 a	0.0 a	0.0 a	0.0 a	1.0 ab

* Means within each column marked with different letters are significantly different ($p \leq 0.05$, Student-Newman-Keuls range test).

** JA, Jasmonic acid; WWS, reclaimed wastewater + brackish water "mix"; WW, reclaimed wastewater.

Table 8.3 Effects of Jasmonic acid and reclaimed wastewater treatments on the mineral composition of broad bean leaves, and properties of the growing media*

Treatments**	% N in young leaves	% N in old leaves	% K in young leaves	% K in old leaves	% P in young leaves	% P in old leaves	% Ca in young leaves	% Ca in old leaves	pH	EC	OM
JA 0.0 mM WWS	2.2 a	3.2 ab	1.8abc	0.44a	0.59a	0.32bc	1.43ab	3.43a	7.7ab	8.31de	2.93 b
JA 0.5 mM WWS	2.8 a	3.2 ab	1.64abc	1.1c	0.53a	0.35c	1.97abc	4.13a	7.67ab	10.05e	3.09 b
JA 1.0 mM WWS	2.2 a	3.84 b	0.9a	0.52a	0.49a	0.32bc	1.31a	3.61a	7.92b	6.74cd	2.90 b
JA 1.5 mM WWS	2.2 a	3.75 b	1.07a	0.58ab	0.34a	0.21a	1.79abc	4.21a	7.9b	6.35cd	3.64 ab
JA 0.0 mM WW	2.0 a	2.70 ab	2.44c	0.91bc	0.47a	0.28abc	2.31bc	4.3a	7.55a	5.18bc	4.52 ab
JA 0.5 mM WW	1.7 a	1.80 a	1.31ab	1.15c	0.27a	0.27ab	1.09a	3.9a	7.71ab	3.32ab	4.73 ab
JA 1.0 mM WW	2.2 a	3.00 ab	1.64abc	1.07c	0.39a	0.24ab	1.38ab	4.14a	7.68ab	3.6ab	5.11 a
JA 1.5 mM WW	2.2 a	3.40 ab	2.51c	1.79d	0.30a	0.21a	2.44c	3.66a	7.69ab	2.63a	5.30 ab
Control	2.4 a	2.90 ab	2.13bc	1.27c	0.46a	0.30bc	1.85abc	4.24a	7.73ab	2.27a	3.08 b

* Means within each column marked with different letters are significantly different ($p \leq 0.05$, Student-Newman-Keuls range test).
** JA, Jasmonic acid; WWS, reclaimed wastewater + brackish water "mix"; WW, reclaimed wastewater.

Irrigating plants with both irrigation sources resulted also in significant changes in soil properties. Soil pH did not differ significantly among the treatments as compared to the control. However, the soil paste EC increased dramatically and significantly upon irrigating plants with mix or with reclaimed wastewater alone. In both cases, it is clear that treating plants with JA (1.0 and 1.5 mM) resulted in reductions in the soil paste EC. The organic matter content of soil did not differ highly between treatments, although it is obvious that irrigating with reclaimed wastewater alone tends to increase the organic matter content of the growing media.

8.3.4. Impact of Treatments on the Fruit Contamination

Fruit contamination with various bacteria was investigated, with no sign of contamination was found. Differences between all treatments, including fresh water-irrigated plants were not significant, with all parameters studied, which include total coliform, fecal coliform, *Salmonella* and *Staphylococcus aureus*.

8.4. Discussion

The main findings of this study are the reduction in vegetative growth upon treating plants with JA, the partial alleviation of salinity stress upon treatment with JA and the altered mineral composition of leaves. Furthermore, the finding that irrigating plants with reclaimed wastewater through drip irrigation resulted in the production of fruits, which are not more contaminated than fruits from plants irrigated with fresh water, is the most significant finding.

The partial alleviation of salinity stress, although coupled with the a reduction of vegetative growth upon exogenous application of JA is in agreement with Horton (1991) and Liu et al. (2002), who found that exogenously applied JA induced stomatal closure in broad bean and barley. Further, Lee et al. (1996) and Pospilisova (2003) stated that JA inhibits carbon dioxide (CO_2) fixation. The preservation of leaf greenness may be explained by the findings of Popova et al. (2003), who reported that MJ could be responsible for protection of photosynthesis against paraquat oxidative stress. Researchers hypothesized that MJ may improve the rate of the carboxylating and protection of the chlorophylls. On the other hand, the growth reduction found on plants irrigated with mix could be directly related to the increasingly negative water potential in the soil, which generally leads to lower transpiration rates and stomatal closure. Lower CO_2 uptake rates subsequently lead to lower photosynthesis rate and consequently lower growth (Rawson and Munns, 1984; Chaves, 1991; Cornic and Massacci, 1996).

Concerning the leaf mineral composition, the present study demonstrated that increasing the salinity level led to a decrease in Ca^{2+} and K^+ levels in leaves, which was reported also with Reid and Smith (2000). The lower Ca^{2+} and K^+-level levels may be attributed to the competition between Na^+ and K^+ or Ca^{2+}, since salt stress caused rapid efflux of cations, particularly K^+ (Marschner, 1995). In addition, salinity is known to reduce Ca^{2+} activity in aqueous solution (Grieve et al., 1999). Accumulation of excess Na^+ may cause also metabolic disturbances in processes where low Na^+ and high K^+ and/or Ca^{2+} are required for optimum function (Marschner, 1995). Furthermore, a decrease in nitrate reductase activity, inhibition of photosystem II (Orcutt and Nilsen, 2000) and chlorophyll breakdown (Krishnamurthy et al., 1987) are also associated with increased Na^+ concentrations. It was obvious in the present study that Ca^{2+} and K^+ levels in plant old leaves increased upon JA application, in combination with irrigation with mix. Increasing K^+ uptake may be related to the increase in Ca^{2+} level in plant tissue, which decreases the salinity effect on the K^+ uptake, as reported by Marschner (1995).

The increase in P in plant upon salinity may be due to the increased availability of P in the soil, or the synergetic effect of Na^+, which is involved in P uptake. Moreover, the present investigations indicated that as salinity increased, nitrogen content also increased, which may be related to higher N fixation and greater N uptake from the soil under the salt-stress conditions (Rao et al., 2000). Additionally, several studies suggest that some proteins produced under salinity stress, such as glycinebetaine, may play a role in osmotic adjustment subjected to salinity stress (Meloni et al., 2004). It is also documented that JA application tends to induce the synthesis certain proteins known as JIPs protein (Maslenkova et al., 1992; Muller-Uri et al., 1988).

Concerning the reproductive growth, results clearly show that increasing salinity led to a decreased fruit weight, which agreed with Rao et al. (2000) and Singleton and Bahlool (1984). However, the fruit weight increased with JA treatments, which could be attributed to inhibiting effect of JA on leaf yellowing. Under such conditions, leaves lived longer, and the filling period of pods was subsequently longer, which may be the reason for bigger fruits (pods).

8.5. Conclusion

The finding in this study that fruits irrigated with reclaimed wastewater were not more contaminated than fruits irrigated with fresh water is the major finding, which is attributed mainly to the use of drip irrigation, which prevents direct contact between water and fruits. This indicates that an irrigation source like this could be widely used in the PT, in combination with both soil-less culture and mulching. The reuse of reclaimed water is essential to meet the expanding water demands of the agricultural sector in the PT, and therefore should be part of the integrated management of the available water resources. The demonstration project showed that high-quality reclaimed water can be used efficiently for the irrigation of broad beans, which are eaten cooked. Concerning the impact of the JA, our results indicate that alleviation of salinity stress imposed on plants through using water resources of inferior quality is possible. However, additional study is required.

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References

Bennett R and Lancette G. (1998). FDA Bacteriological Analytical Manual, eighth edition, revision A. Chapter 12.

Chaves M. (1991). Effects of water deficits on carbon simulation. J Exp Bot 42:1–16.

Cornic G and Massacci A. (1996). Leaf photosynthesis under drought stress. In: Baker NR, ed. Photosynthesis and Environment. Kluwer Academic Publishers, Dordrecht, pp. 347–366.

Farmer E and Rayan C. (1992). Octadecanoid precursors of jasmonic acid activate the synthesis of wound-inducible proteinase inhibitors. Plant Cell 4:29–134.

Feigin A, Ravina I, Shalhevet J. (1991). Irrigation with Treated Sewage Effluent: Management for Environmental Protection. Springer-Verlag, Berlin.

Gygi S, Corthals GL, Zhang Y, Aebersold R. (2000). Evaluation of two-dimensional gel electrophoresis based proteome analysis technology. Proc Natl Acad Sci USA 97: 9390–9395.

Grieve C, Shannon M, Dierig D. (1999). Salinity effects on growth, shoot-ion relations and seed production of *Lesquerella Fendleri* (Gray) S. Wats. In: Janick J, ed. Perspectives on New Crops and New Uses. ASHS Press, Alexandria, VA, pp. 233–237.

Horton R. (1991). Methyl jasmonate and transpiration in barley. Plant Physiol 96:1376–1378.

Johnson W and Parnell J. (1998). Wastewater Reclamation and reuse in the city of St. Petersburg, Florida. In: Asano T, ed. pp. 1037–1104.

Krishnamurthy R, Anbazhagan M, Bhagwat K. (1987). Effect of NaCl toxicity of chlorophyll breakdown in Rice. Ind J Agric Sci 57:567–570.

Lee T, Lur H, Lin Y, Chu C. (1996). Physiological and biochemical changes related to methyl jasmonate-induced chilling tolerance of rice (*Oryza sativa* L.) seedlings. Plant Cell Environ 19:65–74.

Liu X, Zhang S, Lou C, Yu F. (2002). Effect of localized scorch on the transport and distribution of exogenous Jasmonic acid in *Vicia faba*. Acta Botanica Sinica 44:164–167.

Maslenkova L, Miteva T, Popova L. (1992). Changes in the polypeptide patterns of barley seedlings exposed to Jasmonic acid and salinity. Plant Physiol 98:700–707.

Meloni D, Gulotta M, Martínez C, Oliva M. (2004). the effects of salt stress on growth, nitrate reduction and proline and glycinebetaine accumulation in *Prosopis alba*. Braz J Plant Physiol 16:39–46.

Maathuis F and Amtamann A. (1999). K^+ nutrition and Na toxicity: the basis of cellular K^+/Na^+. Ann Bot 84:123–133.

Meerbach D. (2004). Al-Bireh demonstration project on agricultural reuse of wastewater in the West Bank. Water Line 23:1–4.

Mimi Z, Ziara M, Nigim H. (2003). Water conservation and its perception in Palestine—Case study. Water Environ Manage J 17:152–156.

Ministry of Planning and International Cooperation. (1998). Regional plan for the West Bank Governorates: water and wastewater existing situation. Ministry of Planning and International Cooperation, Rammallah.

Muller-Uri F, Parthier B, Nover L. (1988). Jasmonate-induced alteration of gene expression in barley leaf segments analyzed by *in-vivo* and *in-vitro* protein synthesis. Planta 176:241–247.

Orcutt D and Nilsen E. (2000). The Physiology of Plants Under Stress: Soil and Biotic Factors. John Wiley and Sons, Inc., New York.

Popova L, Ananieva E, Hristova V, et al. (2003). Salicylic acid and methyl jasmonate-induced protection on photosynthesis to paraquart oxidative stress. Bulg J Plant Physiol 133–152.

Pospisilova J. (2003). Participation of phytohormones in the stomatal regulation of gas exchange during water stress. Biologia Plantarum 46:491–506.

Rao H, Creelman R, Mullet J, Davis K. (2000). Jasmonic acid signaling modulates ozone-induced hypersensitive cell death. Plant Cell 12:1633–1646.

Rawson H and Munns R. (1984). Leaf expansion in sunflower as influenced by salinity and short term changes in carbon fixation. Plant Cell Environ 7:207–213.

Reid R and Smith F. (2000). The limits of sodium/calcium interaction in plant growth. Aust J Plant Physiol 27:709–715.

Ryan J, Estefan G, Rashid A. (2001). Soil and Plant Analysis Laboratory Manual. ICARDA.

Singleton P and Bohlool B. (1984). Effect of salinity on nodule formation by soybean. Plant Physiol 74:72–76.

Chapter 9
Response of Durum Wheat (*Triticum durum* Desf) Cultivar Acsad 1107 to Sewage Sludge Amendment Under a Semi-Arid Climate

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Abstract The use of sewage sludge on a large scale and at relatively low rates can contribute to the husbandry of urban wastes. This is interesting since this utilization in agriculture appeared to increase crop production. The results of the present investigation, whose objective was to study the response of a rain-fed cereal crop to organic amendment with sewage sludge showed an increase in grain yield and yield component, mainly spike fertility and straw production. 30 t/ha of sewage sludge dry matter were as efficient as 66 kg/ha of mineral nitrogen.

9.1. Introduction

Expansion of urban populations and increased coverage of domestic water supply and sewerage give rise to greater quantities of municipal wastewater. With the current emphasis on environmental health and water pollution issues, there is an increasing awareness of the need to dispose of these wastewaters safely and beneficially. Properly planned use of municipal sewage wastewater and its by-products alleviates many environmental problems and also takes advantage of its nutrient content to grow crops (Pescod, 1992).

Sewage sludge can be used to increase crop production, in those situations in which the growth conditions such as unfavorable climate associated to the high production costs do not permit the utilization of chemical fertilizers to overcome cultivated soil fertility problems (Chattha et al., 2002; Pescod, 1992; Ripert et al., 1990).

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In fact, soils treated with sewage sludge keep their relative humidity longer and their vegetation develops a deeper rooting system as compared to non-treated soils (Tester et al., 1982). Sewage sludge liberates progressively nutritive elements they contain, making them available to the plant along the crop cycle. Nitrogen (N) availability is a function of the prevailing climatic growth conditions; the amount of applied sludge and the C/N ratio (Pescod, 1992; Barbartik, 1985).

Soils treated with sewage sludge tended to have a neutral pH and a high phosphorus (P) and organic matter content (Mohammad et al., 2004; Gomez et al., 1984). However, sewage sludge is often a source of ground water pollution when their content is high in nitrates (Xanthoulis et al., 1998). They are a source of soil salinity (Tasdilas, 1997), heavy metals pollution (Mohammad et al., 2004; Bozkurt et al., 2003; Aboudrare et al., 1998) and odors nuisance (Sachon, 1995).

The present study investigates the response of durum wheat (*Triticum durum* Desf) variety Acsad 1107 to the application of sewage sludge under semi-arid climate.

9.2. Materials and Methods

The experiment was conducted on the experimental site of the Agricultural Farm of the Field Crop Institute of Setif in the northeastern part of Algeria (5° 21′ E longitude and 36° 9′ N latitude, and 1081 m altitude) during the 2002/2003 crop season. The site belongs to the semi-arid bioclimatic zone where the average monthly temperature is 24.1 °C in summer and 7 °C in winter and average annual precipitation is 397.0 mm (Agricultural Farm of the Field Crop Institute, 2003). The monthly average temperatures and precipitations at the experimental site for the period of the study are illustrated in Figure 9.1. The soil is silty clay and highly calcareous (calcisol) and its chemical characteristics are presented in Table 9.1.

The trial was laid out in a randomized complete block design with three replications. Five treatments were compared: a check without application of sludge or N fertilization, a treatment without sludge but fertilized with 33 units ha^{-1} of urea applied during the tillering stage and three treatments with 20, 30 and 40 tons dry sludge ha^{-1}, respectively. The sewage sludge used in the experiment was obtained through the activated sludge treatment process, dried in drying beds for more than six months and its characteristics are reported in Table 9.2.

The different physico-chemical analyses of the soil and the sludge were carried out at the beginning of the experiment on dry and fine samples (<2 mm). The determination of the pH and the electrical conductivity were done by Consort C535 Multiparameter Analyzer on 1:2.5 and 1:5 soil/distilled water suspension respectively, and the other analysises by standard methods (Chapman and Pratt,1982; Cottenie, 1980).

Acsad 1107, a durum (*Triticum durum* Desf) genotype, was sown on December 20, 2002 at a 300 seeds m^{-2} rate on plots whose dimensions were six rows × 5 m long × 0.20 m space between rows. Emergence was noted on December 28, 2002. Dry sludge was passed through 10 × 10 mm mech, and applied onto the experiment

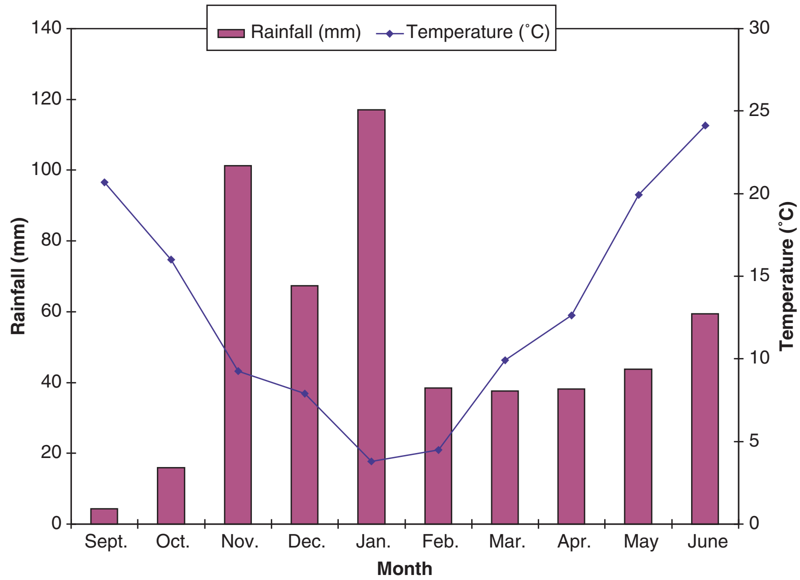

Figure 9.1 Precipitations and temperatures at the experimental site of the Agricultural Farm of the Field Crop Institute (Setif, Algeria) during the period of the study 2002/03

Table 9.1 Characteristics of the soil (0 to 20 cm) used in the experiment at the experimental site of the Agricultural Farm of the Field Crop Institute (Setif, Algeria)

Parameters	pH(H_2O)	EC	OM	D_b	H_s	H_{fc}	H_{wp}	Texture
Units	—	dS m^{-1}	%	g cm^{-3}	%	%	%	—
Mean val.	8.1	0.23	1.7	1.33	51.5	36.5	16.5	Silty clay

EC, electrical conductivity; OM, organic matter; D_b, bulk density; H_s, humidity at saturation; H, humidity at field capacity; H_{wp}, humidity at wilting point

Table 9.2 Characteristics of the sewage sludge originating from the effluents treatment plant of Ain Sfiha (Setif, Algeria)

Parameters	Humidity	pH(H2O)	EC	OM	N	P	K	C/N
Units	%	—	dS m^{-1}	%	%	%	%	—
Mean val.	80	7.3	2.61	58.0	3.30	5.7	0.5	10.15

EC, electrical conductivity; OM, organic matter; N, nitrogen; P, phosphorus; K, potassium.

at the tillering stage. Heading was noted on May 5, 2003 and the crop was harvested on June 16, 2003.

Plant height (PHT) was measured at crop maturity; the number of spikes (SN) and total dry matter (above-ground biomass measured at maturity [BIOM]) produced per m^2 of soil were estimated from vegetative samples harvest from 1 row × 1 m long area. Grain yield (GY) was measured from the combine harvested trial. Thousand kernel-weight (TKW) was estimated from the count and weight of 250 kernels per replicate. The variables number of kernels produced per m^2 (KNM^2), kernels per spike (KS), aerial biomass accumulated at heading (BIOH), vegetative growth rate (VGR), kernel filling rate (KFR), harvest index (HI) and straw yield (STR) have been deduced by calculations using the following formulae:

KNM^2 = 1000(GY/TKW), where
GY = grain yield (g m^{-2})
TKW = thousand kernels weigh (g)
KS = KNM^2 /SN, where
KS = number of kernels per spike
SN = spike number/m^{-2}
BIOH = BIOM – GY, where
BIOM = above-ground biomass measured at maturity (g m^{-2})
VGR = BIOH/DHE, where
VGR = vegetative growth rate (g m^{-2} day^{-1})
BIOH = above-ground biomass accumulated at heading stage (g m^{-2}),
DHE = number of calendars days from emergence to heading stage (days).
KFR = GY/KFP, where
KFR = rate of filling of the number of kernels produced per m^2 (g m^{-2} $days^{-1}$),
KFP = number of calendar days in the kernel filling period (days).
HI = 100 (GY/BIOM)

The collected data were subjected to an analysis of variance. Contrast was employed to test the significance of the following treatments effects: check vs N + sludge, N vs sludge, sludge linear and sludge quadratic (Steel and Torrie, 1980). The relative comparisons between treatments were done according to the following formulae:

Amendment effect N + Sludge (%) = 100 $[(X_{N+S} - X_c)/X_c]$, where
X_{N+S} = mean of N + sludge treatments
X_c = check mean
Sewage sludge effect (%) = 100 $[(X_S - X_c)/(X_N - X_c)]$, where
X_S = mean of sludge treatment
X_N = mean of N treatment
X_c = check mean

9.3. Results and Discussion

The analysis of variance showed a significant treatment effect for the whole variables measured but not for the SN (Table 9.3). The non-significant treatment effect for the SN could be explained by the fact that the amendment (sludge and N) was applied later on, at the tillering stage, when this yield component was partially expressed.

Table 9.3 Means squares of the analysis of variance of the measured variables

Source	Treatment	S+N vs C	S vs N	S lin	S qua	error
dll	4	1	1	1	1	8
GY	20,939.4**	62,489.5**	17,398.1**	3,310.7**	559.5 ns	301.2
SN	1,067.2 ns	411.2ns	458.8 ns	3,398.6*	0.00 ns	389.1
KNM²	3,201,164**	1,848,504**	9054255**	792,289**	11,198 ns	156,915.8
TKW	20.35*	72.6**	4.84 ns	3.23 ns	0.72 ns	2.92
KS	76.55**	225.2**	78.8**	1.25 ns	1.01 ns	1.96
BIOH	66,006.7**	177,055**	39,190.7**	43,146.3**	4,634.8 ns	1,967.6
VGR	4.22**	11.33**	2.51**	2.76**	0.30 ns	0.13
KFR	21.7**	64.79**	18.04**	3.43*	0.58 ns	0.31
BIOM	45,893.0**	449,916**	108,812**	70,360**	1,973.9 ns	833.9
HI	85.46**	293.7**	8.06 ns	0.43 ns	39.6 ns	8.01
STR	61,169**	196,459**	27,749**	19,728**	738.3 ns	763.6
PHT	406.9**	1316.1**	164.7**	140.2**	6.72 ns	4.9

C, check; N, nitrogen; S, sludge; lin, linear; qua, quadratic; GY, grain yield (g m^{-2}); SN, number of spikes/m²; KNM², number of kernels/m²; TKW, 1000 kernel weight (g); KS, number of kernels/spike; BIOH, above-ground biomass accumulated at heading stage (gm^{-2}); VGR, vegetative growth rate (g m^{-2} day^{-1}); KFR, filling rate of the KNM^{-2}; (g m^{-2} day^{-1}); BIOM, above-ground biomass measured at maturity (g m^{-2}); HI, harvest index (%); STR, straw yield (g m^{-2}); PHT, plant height (cm); ns, non-significant effect; *, significant at 5% probability level; ** significant at 1% probability level.

Table 9.4 Mean values of the different treatments

Variables	C	N+S	N	S	20	30	40
SN	318.9	305.8	316.5	302.3	278.5	302.3	326.1
GY	147.5	308.9	242.9	330.9	301.8	342.0	348.7
KNM²	3,159.2	5,933.7	4,769.1	6,321.9	5,879.6	6,479.6	6,606.4
TKW	46.53	52.03	50.93	52.40	51.5	52.8	52.9
KS	9.9	19.6	15.1	21.0	21.3	21.5	20.4
BIOH	223.3	494.9	395.9	527.9	459.1	495.8	628.7
VGR	1.79	3.96	3.17	4.22	3.67	3.97	5.03
KFR	4.75	9.95	7.82	10.65	9.72	11.01	11.23
BIOM	370.8	803.7	638.8	858.7	760.9	837.8	977.5
HI	54.2	43.2	41.7	43.6	44.9	40.6	45.4
STR	169.5	455.6	372.3	483.4	419.6	496.2	534.3
PHT	58.7	82.1	75.6	84.2	80.0	83.0	89.7

C, check; N, nitrogen; S, sludge; GY, grain yield (g m^{-2}), SN, number of spikes/m²; KNM², number of kernels/m²; TKW, 1000 kernel weight (g); KS, number of kernels/spike; BIOH, above-ground biomass accumulated at heading stage (gm^{-2}); VGR, vegetative growth rate (g m^{-2} day^{-1}); KFR, filling rate of the KNM^{-2}; (g m^{-2} day^{-1}); BIOM, above-ground biomass measured at maturity (g m^{-2}); HI, harvest index (%);STR, straw yield (g m^{-2}); PHT, plant height (cm).

The amount of sludge applied remains below the nutrients requirement of the plant since the quadratic effect was not significant for the measured traits. The linear effect of the applied sludge was not significant for the TKW, KS and HI (Table 9.3). The comparison between the check and amendment (N + S) means indicated that mineral and organic fertilization were beneficial to the expression of the measured variables of the crop except for the SN produced per square unit of land (Table 9.4).

Under the growth conditions of the present experiment, the relative contribution of the amendment (N + S) to the increase in the means of the measured variables ranged from 12% for the TKW to 168% for STR. The amendment effect was negative for the HI, which was reduced by 20.0% relative to the mean expressed by the check treatment. This could be explained by the fact that the N or the sludge applied had a more pronounced effect on the accumulated above-ground biomass than on gain yield (Table 9.4; Figure 9.2).

The relative increase in the mean values of the yield component was smaller compared to the increase noted in GY, which resulted from the multiplicative effects of the increase obtained in the yield components. The TKW was the yield component, which was the less sensitive to the amendment effect because this trait is formed when climatic growth conditions become less favorable.

The increase noted in the mean value of straw after application of sludge or mineral N indicated that organic or mineral amendment induced a better expression of the above-ground biomass compared to the GY, which had a negative effect on HI, as explained earlier.

The comparison between organic amendment and mineral fertilization treatments showed that the mean values of these treatments did not differ significantly for the SN, TKW and HI (Table 9.4). For these traits, the effect of sewage sludge application was similar to the effect of N mineral fertilization. Organic amendment induced a relative increase of 128.1% for PHT and 213.5% for the KS. GY showed a 192.7% increase relatively to the check mean yield (Figure 9.3).

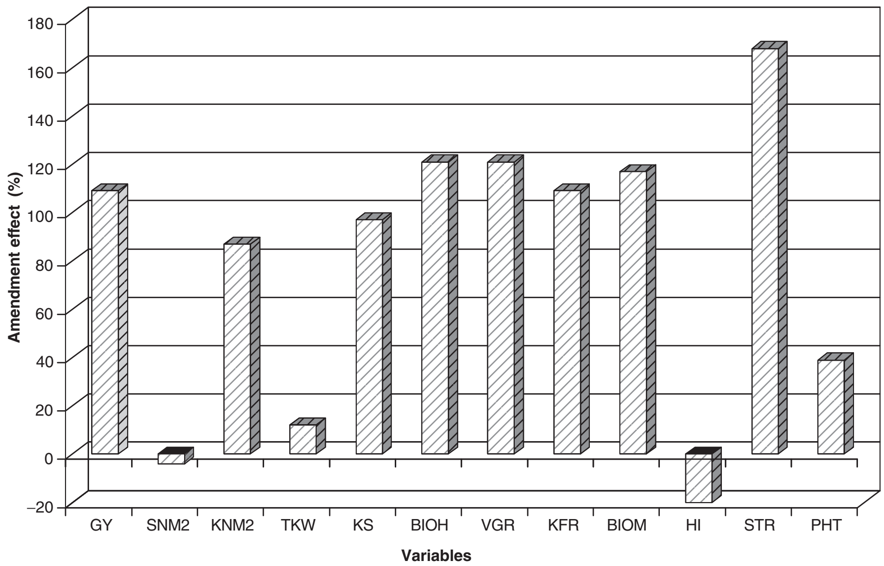

Figure 9.2 Contribution of the applied amendment (N+S) to the increase in the mean values of the measured traits relatively to the mean values of the check

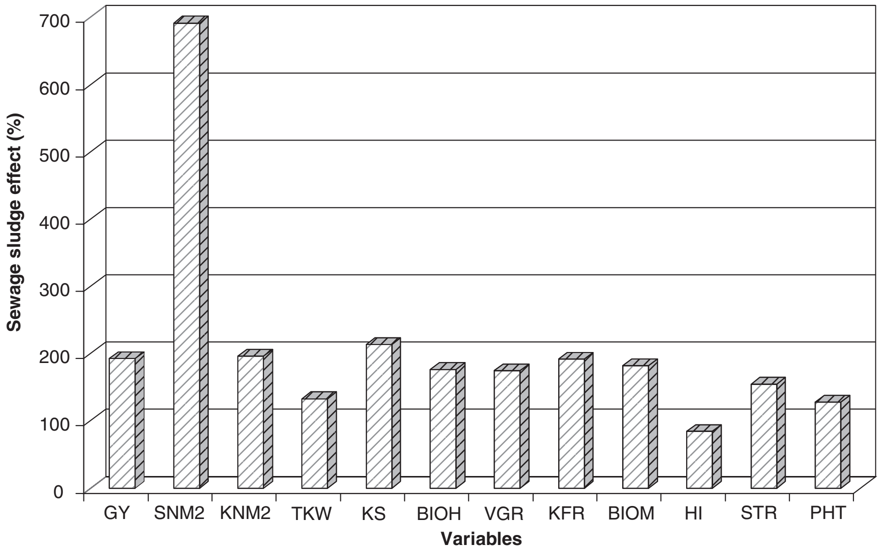

Figure 9.3 Relative increase in the mean values of the measured traits due to the effect of applied sewage sludge as percentage of the mineral nitrogen fertilization effect

On average, application of sewage sludge appeared to be more beneficial for the crop than mineral N fertilization. The effect of the applied sewage sludge was significant and more apparent on spike fertility, BIOH, BIOM, VGR and grain filling rate.

These results indicated that applying sewage sludge to cultivated soils induced an increase in crop GY and contributed to disposal of and recycling of this waste material (Ripert et al., 1990). The increases noted in GY and in the yield associated variables are due to the high concentrations of N, P and organic matter of the sewage sludge applied.

Bouzerzour et al. (2002) reported that the application of sewage sludge increased leaf dimensions, leaf area index, accumulated above-ground dry matter, tillering capacity, PHT of barley (*Hordeum vulgare* L.) and oat (*Avena sativa* L.) genotypes, evaluated in pots experiment. They noted also that the response of the measured variables to the applied sewage sludge was linear, which corroborated the results of the present study. The maximum amount of 40 t ha^{-1} of applied sewage sludge did not show any harmful effect on the expression the measured parameters.

In the present study, yield increase originated from the increase noted in the number of KNM^2 of soil ($r_{GY/KNM}{}^{-2} = 0.98^{*}$) and to the KS ($r_{GY/KS} = 0.92^{*}$) but not from the fertile tillering ability of the crop ($r_{GY/SN} = 0.21^{ns}$). During the course of the experiment, the check treatment was somewhat earlier and senesce more rapidly than the amended treatments. Application of sewage sludge acted as a seal: it reduced the soil evaporation and helped to keep soil moist because of its high organic matter content.

Sewage sludge is considered a substrate that is susceptible to contribute to maintain soil organic matter and improve soil structural stability, cationic exchange and water retention capacities (Gomez et al., 1984). Barbartik et al. (1985) noted that application of sewage sludge during four consecutive cropping seasons increased the organic matter content of the upper 15 cm soil horizon from 1.2 to 2.4%.

Tester et al. (1982) studied the response of tall fescue (*Festuca arundinacea* L.) to sewage sludge. They observed that soil amendment with sewage sludge improved tall fescue N nutrition, stimulated root growth and increased forage production comparatively to the non-amended check. With ray grass (*Lolium perenne* L.), Guiraud et al. (1977) observed an improvement of N concentration of tissue of plants grown in sewage sludge amended soils. Cherak (1999) noted an improvement of the tillering capacity of oat (*Avena sativa* L.) grown under sewage sludge-amended soil.

According to Sachon (1995), incubated sewage sludge developed aerobic and anaerobic chemical reactions that, in 6 to 7 weeks, reduced the organic matter to the form of compost similar to the humus. The mineralization of organic N is dependent on the C/N ratio: higher this ratio is, the lower the mineralization (Barbartik et al., 1985; Sachon, 1995).

9.4. Conclusion

Use of wastewater in agriculture could be an important consideration when its disposal is being planned in arid and semi-arid regions. Treated sludge can be applied to growing cereal crops without constraint. Land application of raw or treated sewage sludge can significantly reduce the sludge disposal cost component of sewage treatment and provid3 a large part of the N and P requirements of many crops. The organic matter in sludge can improve the water-retaining capacity and structure of soils, especially when applied in the form of dewatered sludge cake. Sludge application resulted in significantly increased crop yields and attributed to the beneficial effects on soil structure and nutrient content.

References

Aboudrare N, Jellal T, Benchekroun D, Jemali A. (1998). Wastewater reuse for agricultural purposes in Ouarzazate. Terre et Vie 26:7–12.

Agricultural Farm of the Field Crop Institute. (2003). Weather data report. Experimental Farm, Field Crop Institute, Setif, Algeria.

Barbartik A, Lawarabnce Jr, Sikpra J, Colacicco D. (1985). Factors affecting the mineralization of nitrogen in sewage sludge applied to soil. Am J Soil Sci. 49:1403–1406.

Bozkurt MA and Yarilgaç T. (2003). The effects of sewage sludge applications on the yield, growth, nutrition and heavy metal accumulation in apple trees growing in dry conditions. Turk J Agric For 27:285–292.

Bouzerzour H, Tamrabet L, Kribaa M. (2002). Response of barley and oat to the wastewater irrigation and to the sludge amendment. In: Proceedings of the International Seminar: Biology and Environment. University Mentouri, Constantine, Algeria, p. 71.

Chapman HD and Pratt PF. (1982). Methods for analysis of soils, plants and water. Chapman Publishers, Riverside, CA.
Chatha TH, Haya R, Latif I. (2002). Influence of sewage sludge and organic manures application on wheat yield and heavy metal availability. Asian J Plant Sci 1:79–81.
Cherak L. (1999). Incidence of wastewater reuse on the microflora and the oat behaviour. Master's thesis, University of Batna, Algeria.
Cottenie A. (1980). Soil and plant testing as a basis for fertilizer recommendations. Food and Agriculture Organization (FAO) Soils Bulletin no. 38/2. FAO, Rome.
Dursun A, Turkmen O, Turan M, Sensoy S, Cirka M. (2005). Effects of sewage sludge on the seed emergence, development and mineral contents of pepper (*Capsicum annum*) Seedling. Asian J Plant Sci 4:299–304.
Gomez A, Lineres M, Tanzin J, Solda P. (1984). The study of the effect of addition of the sludge to sandy soils on the quantitative and qualitative evolution of the organic matter. CR Acad Sc Fr pp. 516–524.
Guiraud G, Fardeau GC, Hetier JM. (1977). Evolution of the soil nitrogen in the presence of the sewage sludge. In: Proceedings of the First Symposium on Soil and Solid Wastes Research. pp. 27–33.
Mohammad MJ and Athamneh BM. (2004). Changes in soil fertility and plant uptake of nutrients and heavy metals in response to sewage sludge application to calcareous soils. J Agronomy 3:229–236.
Pescod MB. (1992). Wastewater treatment and use in agriculture. FAO irrigation and drainage paper no. 47. FAO, Rome.
Ripert C, Tiercelin JR, Navarot C, et al. (1990). Agricultural and forestry reuse of the domestic wastewater. Technical report of Cemagref. 79:18.
Sachon S. (1995). The sludge of the urban wastewater treatment plants, reuse in agriculture. BTI 21:14–29.
Tester CF, Sikora LJ, Taylor JM, Parr JF. (1982). N utilization by tall fescue from sewage sludge, compost amended soils. Agro J 74:1013–1018.
Steel GDS and Torrie JH, eds. (1980). Principles and Procedures of Statistics: a Biometrical Approach. Mc Graw Hill Book Company, New York.
Tasdilas CD. (1997). Impact of waste water reuse on some soil properties. In: International Conference on Water Management, Salinity, and Pollution Control Towards Sustainable Irrigation in the Mediterranean Region. Options méditerranéennes série B- CIHEAM. pp. 213–226.
Xanthoulis D, Kayamanidou M, Choukr-Allah R, et al. (1998). Wastewater reuse in irrigation, global approach of the effluents treatment, comparison of the different irrigation systems on divers crops and their institutional and organisational aspects. Synthesis of the multilateral research projects on wastewater. Faculty of Agronomic Sciences, Gembloux.

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Chapter 10
Waste Stabilization Ponds: A Highly Appropriate Wastewater Treatment Technology for Mediterranean Countries

Duncan Mara

Abstract This chapter describes waste stabilization pond (WSP) systems for wastewater treatment. WSP systems comprise a series of anaerobic and facultative ponds and sometimes maturation ponds. Rock filters can be used instead of maturation ponds and they can be aerated to remove ammonia and to improve biochemical oxygen demand and suspended solids removals. Effluent quality is high, and properly designed and well maintained WSP systems produce effluents that can be safely used for both restricted and unrestricted crop irrigation.

10.1. Introduction

Waste stabilization pond (WSP) systems are a high-performance, low-cost, low-energy (often zero-energy) and low-maintenance wastewater treatment process, especially suitable in warm climates.

There are three principal types of WSP systems: anaerobic, facultative and maturation (Figures 10.1 to 10.3).[1] These different types of ponds are arranged in

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[1]Other types of WSP exist; for example, high-rate or "advanced" algal ponds and macrophyte ponds, but these are not recommended for normal municipal usage (see Mara and Pearson, 1998).

I. Al Baz et al. (eds.), *Efficient Management of Wastewater.*

Figure 10.1 An anaerobic pond in Cyprus treating wastewater from a slaughterhouse

Figure 10.2 Partial view of a facultative pond in southern France treating domestic wastewater

Figure 10.3 Maturation ponds in northern France

series; at any one site there is usually more than one series, with each series comprising an anaerobic pond followed by a facultative pond and, depending on the effluent quality required, by one or more maturation ponds. Rock filters (RFs) are a land-saving alternative to maturation ponds (Section 10.2.2).

10.1.1. WSP System Usage in Mediterranean Countries

WSP systems are widely used in France where there are more than 2,500 systems, each typically comprising a facultative pond (sized at 6 m^2 per person) and two maturation ponds (each 2.5 m^2 per person; Cemagref and Agences de l'Eau, 1997; Racault and Boutin, 2005). They are also used in Portugal, Spain, Greece, Israel, Jordan, Egypt, Algeria and Morocco (i.e., in virtually every Mediterranean country; details in Mara and Pearson, 1998). In Greece, WSP systems were found to be the cheapest treatment process up to the land price of USD 300,000 per ha (Tsagarakis et al., 2003).

10.1.2. Advantages of WSP Systems

Cost is the most important advantage of WSP systems: they are almost always the cheapest form of wastewater treatment to construct and to operate (Table 10.1 gives costs in France for WSP and five other treatment processes; see also Arthur, 1983). They are also very easy to operate and maintain: there is no electromechanical machinery and only unskilled labor is required to perform very simple tasks (see Section 10.3). The oxygen required by the pond bacteria to oxidize the wastewater BOD is supplied by the micro-algae that grow naturally and profusely in facultative and maturation ponds (Figure 10.4).

10.1.3. Perceived Disadvantages of WSP Systems

WSP systems are commonly thought (especially by those selling energy-intensive electromechanical wastewater treatment systems, such as activated

Table 10.1 Capital and operation and maintenance costs of various wastewater treatment processes for a population of 1,000 in France in 1997

Treatment process	Capital costs (ECU per person)*	O&M costs (ECU per person per year)*
Activated sludge	230	11.50
Trickling filter	180	7.00
Rotating biological contactor	220	7.00
Aerated lagoon	130	6.50
Vertical-flow constructed wetland**	190	5.50
Waste stabilization ponds	120	4.50

*Average exchange rates in 1997: 1 ECU = GBP 0.69 = USD 1.17 (www.oanda.com/convert/fxhistory).
**Two-stage vertical-flow constructed wetland receiving raw wastewater.
Note: All processes designed to produce effluents complying with French regulations (see Alexandre et al., 1997; Racault and Boutin, 2005).
Source: Alexandre et al., 1997.

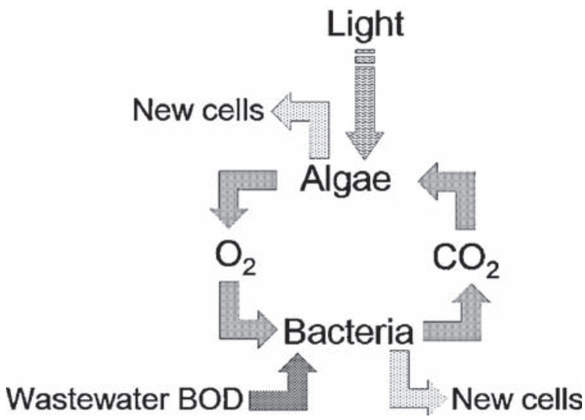

Figure 10.4 Algal-bacterial mutualism in facultative and maturation ponds

sludge) to require excessive areas of land, to be unable to produce satisfactory effluents (especially in terms of their suspended solids [SS] concentrations due to the algae present), to generate odors and to lose too much water by evaporation.

10.1.3.1. Land Area Requirements

Although it is true that WSP systems require considerably more land than energy-intensive processes such as activated sludge, this is not a disadvantage in countries with large areas of unused land (e.g., Jordan is >90% desert). Furthermore, it should be realized that land purchased for WSP systems is an investment, whereas the money spent on electricity for energy-intensive processes is money gone forever.

10.1.3.2. Effluent Quality

In the European Union, WSP system effluent requirements are 25 mg filtered BOD or less and 150 mg SS or less per liter (Council of the European Communities, 1991) This quality is achieved by a facultative pond loaded at 80 kg BOD ha^{-1} day^{-1}, which is the design loading for winter temperatures of 8 °C and below (Abis and Mara, 2003). Often, however, the local environmental regulator sets a higher standard than this and, therefore, either maturation ponds or RFs are required. Maturation ponds have lower algal biomass concentrations than facultative ponds, which decrease along a series of maturation ponds. As noted in Section 10.2.2, the effluent quality achieved by aerated RFs is very high and can be expected to satisfy even the most stringent regulator.

10.1.3.4. Odor

WSP systems, provided they are correctly designed and operated and maintained properly, do *not* cause odor. To avoid odor release from anaerobic ponds, the sulphate concentration in the raw wastewater should be less than 500 mg SO_4 L^{-1} (Gloyna and Espino, 1969); this is rarely a problem as the maximum permissible sulphate concentration in drinking water is 250 mg SO_4 L^{-1} (World Health Organization, 2003) and, although the sulphate concentration in wastewater is higher than that in the drinking water (due to sulphates being used in domestic detergents), it very seldom exceeds 500 mg SO_4 L^{-1} (however, it is always worthwhile to measure its concentration in both the local drinking water and the wastewater deriving from it).

Overloaded WSP will present odor problems, just as any overloaded wastewater treatment process does. The solution in this case is to construct an additional series of ponds to cope with the increased load.

10.1.3.5. Evaporation

WSP systems do, of course, lose water by evaporation, but commonly less than 20% of the influent raw wastewater. This is often claimed to be a serious disadvantage of WSP systems, but the real question is whether the value of the water lost is greater than the cost of the electricity that would be used for an alternative treatment process, such as activated sludge—the answer will almost always be “No.” Evaporation can be minimized by using RFs, rather than maturation ponds (see Section 10.2.2).

10.2. WSP System Design

An introduction to WSP design is given by Peña Varón and Mara (2004) and detailed in Mara and Pearson (1998) and Mara (2004). Only a brief outline of the concepts is included here. Box 10.1 gives the design equations in summary form.

Box 10.1 Pond design equations

Anaerobic ponds

The design value for the volumetric biochemical oxygen demand (BOD) loading (λ_V, g m^{-3} day^{-1}) varies with the design temperature (T, °C), taken as the mean temperature of the coldest month, as follows: at ≤10 °C $\lambda_V = 100$ g m^{-3} day^{-1}, at 15 °C $\lambda_V = 200$ g m^{-3} day^{-1}, at 20 °C $\lambda_V = 300$ g m^{-3} day^{-1}, and at 25 °C $\lambda_V = 350$ g m^{-3} day^{-1}, with linear interpolation between these values. The area (A_V, m^2) is given by:

$$A_V = \frac{L_i Q}{\lambda_V D_A}$$

where D_A is the anaerobic pond depth (m).

Facultative ponds

The surface BOD loading (λ_S, kg ha^{-1} day^{-1}) is a function of the design temperature (T, °C), taken as the mean temperature of the coldest month:

$$\lambda_S = 350(1.107 – 0.002\,T)^{T-25}$$

The area (A_F, ha) is given by:

$$A_F = \frac{10 L_i Q}{\lambda_S D_F}$$

where D_F is the facultative pond depth (m).

▸ ***Check effluent quality for restricted irrigation:***

Once the anaerobic and facultative ponds are designed, it is sensible to check if the facultative pond effluent is suitable for restricted irrigation (Chapter 1). The required log unit reduction of pathogens is taken to be achieved by the same reduction of *E. coli*, for which the equations of Marais (1974) are used. For a pond series comprising only anaerobic and facultative ponds these are:

$$N_F = \frac{N_i}{(1 + K_{B(T)}\theta_A)(1 + K_{B(T)}\theta_F)}$$

$$K_{B(T)} = 2.6(1.19)^{T-20}$$

where N_F and N_i are the numbers of *E. coli* per 100 mL of the facultative pond effluent and raw wastewater, respectively; $K_{B(T)}$ is the first-order rate constant for *E. coli* removal (day^{-1}); and θ_A and θ_F are the mean hydraulic retention times in the anaerobic and facultative ponds, respectively (days). The design temperature T is taken as the mean temperature of the coolest month in the irrigation season. An *E coli* reduction of 3 to 4 log units is required (i.e., for $N_i = 10^7 - 10^8$ per 100 mL, N_e should be no more than $10^4 - 10^5$ per 100 mL for highly mechanized agriculture or $10^3 - 10^4$ per 100 mL for labor-intensive agriculture; see Chapter 1).

For restricted irrigation, there should be no more than one intestinal nematode egg per liter of treated wastewater. For E_i eggs per liter of raw wastewater, the number of eggs per liter of facultative pond effluent (E_F) is given by the equations of Ayres et al. (1992):

$$E_F = E_i(1-r_A)(1-r_F)$$

where r_A and r_F are the fractional egg removals in the anaerobic and facultative ponds, respectively, given by:

$$r = 1-0.41[\exp(-0.49\theta + 0.0085\theta^2)]$$

where, for $r = r_A$, $\theta = \theta_A$ and, for $r = r_F$, $\theta = \theta_F$.

Maturation ponds

Maturation ponds are designed either for *E. coli* removal or for nitrogen (N) removal, or occasionally for both.

▸ ***E. coli removal:***

Marais' (1974) equations are used, as follows:

$$N_e = \frac{N_F}{[1 + K_{B(T)}\theta_{M1}][(1 + K_{B(T)}\theta_M)^n]}$$

where N_e is the number of *E. coli* per 100 mL of the final effluent, θ_{M1} and θ_M are the retention times (days) in the first and subsequent maturation ponds, respectively, and *n* is the number of maturation ponds after the first maturation pond. The value of θ_{M1} is such that the surface BOD loading on this pond is 70 percent of that on the facultative pond; it is therefore given by:

$$\theta_{M1} = \frac{10L_iD_{M1}}{0.7\lambda_F}$$

▸ ***N removal:***

For total N removal Reed's (1985) equation is used, as follows:

$$TN_e = TN_i\exp\{-[0.0064(1.039)^{T-20}][\theta + 60.6(\text{pH}-6.6)]\}$$

where TN_e and TN_i are the effluent and influent total N concentrations (mg N L^{-1}), respectively. This equation is applied to the facultative pond and then to each maturation pond in turn; it is not used for the anaerobic pond as there is no total N removal in anaerobic ponds, only partial conversion of organic N to ammonia.

(continued)

Box 10.1 (continued)

For ammonia removal one of the equations of Pano and Middlebrooks (1982) is used, as follows:

(a) for $T \leq 20\,°C$:

$$AN_e = AN_i/\{1 + [(A/Q)(0.0038 + 0.000134T)\exp((1.041 + 0.044T)(pH\text{–}6.6))]\}$$

(b) for $T > 20\,°C$:

$$AN_e = AN_i/\{1 + [5.035 \times 10^{-3}(A/Q)][\exp(1.540 \times (pH\text{–}6.6))]\}$$

where AN_e and AN_i are the effluent and influent ammonia-N concentrations (mg L^{-1}), respectively. These equations are applied to the facultative pond and then to each maturation pond in turn. The ammonia-N concentration in the influent to the facultative pond may be taken as about 75% of the total N concentration in the raw wastewater.

Rock Filters (RFs)

The RF area (A_{RF}, m^2) is given by:

$$A_{RF} = \frac{Q}{HLR \times D_{RF}}$$

where Q is the wastewater flow (m^3 day^{-1}), HLR the hydraulic loading rate (day^{-1}; range: 0.6–1 day^{-1}), and D_{RF} the wastewater depth in the RF (0.5–1 m). This equation is valid for both aerated and unaerated filters. Currently, no design equations are available for BOD, SS, N and *E. coli* removals in aerated RFs, only the effluent quality data given in the main text.

10.2.1. Anaerobic, Facultative, and Maturation Ponds

Anaerobic and facultative ponds are designed on the basis of volumetric and surface BOD loadings (in g BOD m^{-3} day^{-1} and kg BOD ha^{-1} day^{-1}), respectively, to achieve high BOD removals, with concomitant high SS removal in anaerobic ponds. (SS removals in facultative ponds are not as high due to the growth of green algae, the cells of which are measured as SS.) Design values for these loadings depend on the design temperature, which is taken as the mean temperature of the coldest month (see Box 10.1). Depths are typically 3 m in anaerobic ponds (range 2 to 5 m) and 1.5 m in facultative ponds (1 to 2 m).

Maturation ponds are designed for the removal of excreted pathogens and nutrients such as nitrogen (N) and phosphorus (P; Box 10.1). Pathogen removal is extremely important when the effluent is to be used for crop irrigation (Chapter 1). BOD and SS removals are much lower than in anaerobic and facultative ponds. Depths are typically 1 to 1.5 m.

10.2.2. Rock Filters

RFs are subsurface horizontal-flow filters with a rock size of 75 to 200 mm. They have been used for more than 30 years in the United States to remove algal BOD and SS in maturation pond effluents (O'Brien et al., 1973; Swanson and Williamson, 1980; Middlebrooks 1995; U.S. Environmental Protection Agency, 2002; Figure 10.5). Work in Jordan on RFs composed of "wadi gravel" with a size of 30 to 230 mm has confirmed their efficiency: SS removal was about 60% at a loading of 32 to 44 g SS m^{-3} day^{-1} (Saidam et al., 1995). However, these RFs were unaerated and thus unable to remove ammonia through nitrification as they were anoxic. Recent work in England has investigated the use of aerated RFs to treat facultative (rather than maturation) pond effluents; it was found that aerated RFs effectively enabled ammonia removal by nitrification and achieved higher BOD, SS and fecal coliform removals than unaerated RF (Mara and Johnson, 2006). Mean effluent quality from an aerated RFs receiving a hydraulic loading rate of 0.6 m^3 of facultative pond effluent per m^3 of RF volume per day was about 9 mg BOD, about 7 mg SS, about 3 mg ammonia-N per liter and 10 to 1,000 fecal coliforms per 100 mL. This is a very good quality effluent indeed, which is suitable for both restricted and unrestricted crop irrigation. In fact, RFs should be considered an integral part of WSP systems, in exactly the same way that secondary sedimentation tanks are considered an integral part of activated sludge systems, since they both serve the same purpose, namely the removal of excess biomass produced in the preceding biological treatment—bacteria in the case of activated sludge and algae in the case of WSP. The area required for RFs is very much less than that for maturation ponds: about 0.4 m^2 per person, compared with 5 m^2 per person for maturation ponds in France. RF design is detailed in Box 10.1.

10.3. WSP Maintenance Requirements

The maintenance requirements of WSP are listed in Table 10.2. It is essential that these simple tasks are done regularly to avoid operational problems. Therefore, while only unskilled labor is required, it is very important that all maintenance work is adequately supervised.

10.4. WSP Systems: A Highly Sustainable Solution

WSP systems are a high-efficiency, low-maintenance and low-cost wastewater treatment process. Land area requirements can be minimized by good design, and also by using RFs instead of maturation ponds. High-quality effluents can be produced that are suitable for crop irrigation, thereby ensuring that the valuable nutrients in domestic wastewater are not wasted.

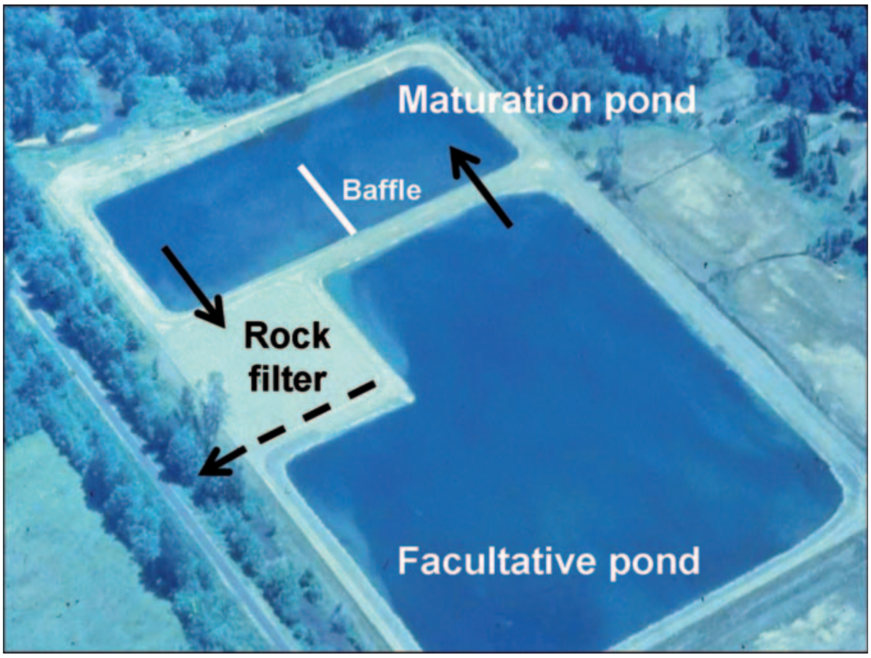

Figure 10.5 Rock filter treating maturation pond effluent at Veneta, Oregon, in the United States

Table 10.2 WSP maintenance requirements

Maintenance task	Frequency
Removal of screenings and grit from preliminary treatment processes	Daily
Cutting the grass on the embankments and removing it so that it does not fall into the pond (necessary to prevent the formation of mosquito-breeding habitats)	Monthly
Removal of floating scum and floating macrophytes, such as *Lemna* (duckweed), from the surface of facultative and maturation ponds (required to maximize photosynthesis and surface reaeration and to prevent fly and mosquito breeding)	Weekly
Spray the scum on anaerobic ponds (which should not be removed as it aids the treatment process), as necessary with clean water, pond effluent or a suitable biodegradable larvicide to prevent fly breeding	Monthly
Remove the sludge from anaerobic ponds	Annually*
Remove any solids blocking the inlets and outlets	Whenever observed
Repair any damage to embankments caused by rodents, rabbits or other animals	Whenever observed
Repair any damage to the external fences and gates	Whenever observed

*Usually done when the pond is one-third full of sludge, which takes about 2 to 4 years. However, it is better to desludge partially every year as a task that has to be done every April, for example, is more likely to be done than if scheduled for every so many years.

References

Abis K and Mara DD. (2003). Research on waste stabilization ponds in the United Kingdom—Initial results from pilot-scale facultative ponds. Water Sci Technol 48:1–8.
Alexandre O, Boutin C, Duchène P, et al. (1997). Filières d'Epuration Adaptées aux Petites Collectivités. FNDAE technical document no. 22. Ministère de l'Agriculture et de la Pêche, Paris.
Arthur JP. (1983). Notes on the Design and Operation of Waste Stabilization Ponds in Warm Climates of Developing Countries. Technical paper no. 7). The World Bank, Washington, DC.
Ayres RM, Alabaster, GP, Mara DD, Lee DL. (1992). A design equation for human intestinal nematode egg removal in waste stabilization ponds. Water Res 26:863–865.
Cemagref and Agences de l'Eau. (1997). Le Lagunage Naturel: Les Leçons Tirées de 15 Ans de Pratique en France. Centre National du Machinisme Agricole, du Génie Rural, des Eaux et des Forêts, Lyon.
Council of the European Communities. (1991). Council directive 91/271/EEC of 21 May 1991 concerning urban waste water treatment. Official Journal of the European Communities L135:40–52.
Gloyna EF and Espino E. (1969). Sulfide production in waste stabilization ponds. Journal of the Sanitary Engineering Division, American Society of Civil Engineers 95:607–628.
Mara DD. (2004). Domestic Wastewater Treatment in Developing Countries. Earthscan Publications, London.
Mara DD and Johnson MJ. (2006). Aerated rock filters for enhanced ammonia and fecal coliform removal from facultative pond effluents. Journal of Environmental Engineering, American Society of Civil Engineers 132:574–577.
Mara DD and Pearson HW. (1998). Design Manual for Waste Stabilization Ponds in Mediterranean Countries. Lagoon Technology International, Leeds.
Marais GvR. (1974). Faecal bacterial kinetics in waste stabilization ponds. Journal of the Environmental Engineering Division, American Society of Civil Engineers 100:119–139.
Middlebrooks EJ. (1995). Upgrading pond effluents: an overview. Water Sci Technol 31:353–368.
O'Brien WJ, McKinney RE, Turvey MD, Martin DM. (1973). Two methods for algae removal from wastewater stabilization ponds. Water Sewage Works J 120:66–73.
Pano A and Middlebrooks EJ. (1982). Ammonia nitrogen removal in facultative waste stabilization ponds. J Water Pollut Control Fed 54:344–351.
Peña Varón MR and Mara DD. (2004). Waste Stabilization Ponds—Thematic Overview Paper. IRC International Water and Sanitation Centre, Delft.
Racault Y and Boutin C. (2005). Waste stabilization ponds in France: state of the art and recent trends. Water Sci Technol 51:1–9.
Reed SC. (1985). Nitrogen removal in wastewater stabilization ponds. J Water Pollut Control Fed 57:39–45.
Saidam MY, Ramadan SA, Butler D. (1995). Upgrading waste stabilization pond effluent by rock filters. Water Sci Technol 31:369–378.
Swanson GR and Williamson KJ. (1980). Upgrading lagoon effluents with rock filters. Journal of the Environmental Engineering Division, American Society of Civil Engineers 106:1111–1129.
Tsagarakis KP, Mara DD, Angelakis AN. (2003). Application of cost criteria for selection of municipal wastewater treatment systems. Water Air Soil Pollut 142:187–210.
U.S. Environmental Protection Agency. (2002). Rock Media Polishing Filter for Lagoons. Wastewater technology fact sheet no. EPA 832-F-02-023. U.S. Environmental Protection Agency, Washington, DC.
World Health Organization (WHO). (2003). Guidelines for Drinking Water Quality, third ed. WHO, Geneva.

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Chapter 11
Sustainable Sanitation by Constructed Wetlands in the Mediterranean Countries: Experiences in Small/Medium-Size Communities and Tourism Facilities

Fabio Masi(✉), Giulio Conte, and Nicola Martinuzzi

Abstract Constructed wetlands (CWs) are efficient treatments for various types of wastewater. A strict relationship between the CWs treatment performances and the local climatic conditions have been widely demonstrated. This chapter focuses on the treatment strategies, adopted designs and obtained performances in different CW facilities currently operating around the Mediterranean basin. Several CW treatment systems operating in Italy for small- or medium-size communities are described in deeper detail and compared to the foreign experiences. In general, the Mediterranean CWs systems seem to obtain better results, probably due to the more constant and warmer climatic conditions, in comparison to most of the other European experiences. The operating experiences generally show a high rate of efficiency in the removal of organic content (BOD, COD), nitrogen (Ntot, NH_4^+, NO_3^-), total suspended solids (TSS) and pathogens (EC, FC, TC), both in secondary and tertiary treatment plants.

11.1. Introduction

Constructed wetlands (CWs) have been adopted by several small-/medium-size communities as a cost-effective means of secondary and tertiary wastewater treatment, in order to meet more stringent standards and to lower operating costs. Some small systems have now been in existence for nearly 15 years, while wetland treatment

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systems for larger towns and small cities have become a more recent trend. As tertiary treatment, CWs have been realized even for several thousand population equivalent (PE; some experiences in the order of hundred thousand).

The operating CWs can be subdivided into two main categories: surface-flow or subsurface-flow design. In surface-flow wetlands (FWS) the wastewater flows through a shallow basin planted with emergent and submerged macrophytes. These kinds of systems are mainly exploited for tertiary treatment or polishing stage and also in several cases of diffuse pollution. In subsurface flow or "reed-bed" treatment systems (RBTS), the wetland is filled with gravel or sand or similar substrates, and the plants, most commonly reeds *(Phragmites australis* or *communis*), grow rooted in the filling medium. This category, in which the horizontal flow (HF) beds and the vertical flow (VF) beds can be used alone or in combination, are particularly recommended for a complete wastewater treatment, in most cases after an efficient primary treatment. Ponds are sometimes inserted in the treatment train, both as primary stage—with the aim of a light anaerobic digestion pretreatment and equalization basin—or as final storage (this case often joined with a reuse practice).

Just to talk about the Mediterranean basin, very successful experiences with CWs have been reported for France (Lesavre and Iwema, 2002; Liénard et al., 1995; Molle et al., 2004; Paing and Voisin, 2004), Spain (Garcia et al., 2004), Portugal (Matos et al., 2002), Morocco (Mandi, 1996), Italy (Conte et al., 2001; Masi et al., 2000), Egypt (Awad and Saleh, 2001; Higgins et al., 2001), Israel (Brenner and Messalem, 2002), Slovenia (Bulc, 2002; Bulc et al., 2003), Croatia (Shalabi, 2004), Greece (Papadopoulos, 2002), Turkey (Yildiz et al., 2004).

Since 1999, CWs in Italy have been "officially" recognized as a treatment technology. The newest national law concerning wastewater, D.L.152/99, officially recognizes the use of CWs for urban centers with populations in the range of 10 to 2,000 PE discharging into freshwater, in the range of 10 to 10,000 PE discharging in sea water, and for tourist facilities (TFs) and other point sources with high rates of fluctuation of organic and/or hydraulic loads. Since the 1980s, more than a thousand CWs, both free water and subsurface HF and VF systems, have been realized. Most of the facilities are located in the northern and central parts of the country.

The main applications throughout the Mediterranean countries are:

11.1.1. Point-source pollution

- Municipal and domestic wastewater treatment, both as secondary and tertiary stage; only in France a particular configuration of VF beds has shown optimal performances in raw wastewater treatment (without any primary sedimentation stage).
- Black water treatment; mainly the HF and VF reed beds have been used, due to their good capacity in treating high organic content wastewater; due to the high inlet ammonia concentration, VF beds seem to be a necessary stage for this kind of wastewater treatment.

- Grey water treatment; HF or VF systems have demonstrated their expected good performances for this “easy” wastewater; they are in most cases joined with a light pretreatment, like a degreaser or a septic tank.
- Rain water disinfection and filtration; all CWs typologies have been used for this purpose.
- Landfill leachate treatment; for this “difficult” wastewater HF and/or VF beds are obtaining satisfactory results; they need a strong pretreatment, like an oxidation pond (equalization and oxidation are needed).
- Sludge dewatering; it’s a particular kind of modified VF beds, a promising solution for the sludge management (both primary or secondary sludge).

11.1.2. Diffuse pollution

- Agricultural and urban runoff; CWs systems, mainly FWS kind, can act efficiently also in nutrients removal, like fertilizers solved by the runoff, and also in buffering and partially treating the combined sewer overflows.
- Highway runoff; HF systems, and also FWS, are showing interesting removal rates for persistent organics, like polycyclic aromatic hydrocarbons) and for some heavy metals.

Particular attention has been given to CW application for remote TFs, hosting up to a few hundred PE, which face several severe problems in the treatment of their wastewater, especially during spring and summer periods: lack of connection to a sewer system or on-site treatment plant; high variations in their seasonal and/or weekly wastewater hydraulic and organic loads; clean water supply deficiencies; shortage of receiving water bodies; insufficient owner/operator availability for systems maintenance; they are often situated in a environmentally sensitive natural areas needing special protection and preservation. Given this situation, if used in a synergy with the other available approaches for sustainable water management (SWM), such as water saving, grey and black water segregation, water and nutrients recycling and reuse, CWs have the potential to become one of the most appropriate practices for TFs wastewater treatment and management. There is an apparent need for lowering the impact and contribution of TFs to water pollution and an increase in carrying-capacity of their often quite exploited territories. The adoption of CWs for wastewater treatment could represent an essential strategic step in protection and preservation of Mediterranean natural resources. Over the past 13 years, the engineering company IRIDRA S.r.l. (Florence, Italy) has designed more than 100 CWs for wastewater treatment that are currently in operation in Italy (Conte et al. 2001; Masi et al. 1999). The performances of four different CWs designed by IRIDRA to treat TFs wastewater have been monitored for about two years by IRIDRA and Agenzia Regionale per la Protezione dell’Ambiente della Toscana, the regional public authority responsible for environmental water quality monitoring. These CW treatment systems are located near four TFs, each representing some of the most common types in the country:

1. Abetina Reale Shelter (Reggio Emilia) is a mountain shelter with a restaurant, open to the public mainly during weekends and the summer period (July/August). The shelter is located inside a natural park (Parco del Gigante) in the Apennines, between Tuscany and Emilia Romagna. Prior to the CW establishment by IRIDRA, this facility's wastewater was discharged into a high-quality stream with a small Imhoff tank ($2\,m^3$) as the only primary treatment.
2. Baggiolino Farmhouse (Florence), located in the hills about 25 km from Florence, Italy. This tourism complex is considered a small settlement (three separate houses) inhabited only by the owners (5 PE) during the winter months (November until March). However, from early March until October, this complex is available for weekly rentals. This form of countryside tourism has been promoted in Italy over the last 10 years, resulting in several thousand tourist farms (commonly located in remote areas or city outskirts) presently in operation.
3. Relais Certosa Hotel (Florence): a five-star hotel in Florence with the typical tourist fluxes of the city (about 35,000 guests per year), frequently occupied at its maximum capacity. There are no sewer systems in the proximity of this facility (about 1500 m distance), so it has been necessary to adopt an on-site treatment system. The hotel was using drinking water for irrigation. A reuse of treated wastewater has been requested by the owners to decrease water costs. Some-cost saving measures, such as the use of double-choice toilet flushing in the hotel rooms, were already present at the time of CW installation.
4. "La Cava" Camping Site (Arezzo): a small camping site recently established and designed according to the SWM principles (water saving, reuse, recycling). The black and grey waters are segregated and treated by a CW; the treated grey water is recycled for toilet flushing while the treated black water is reused for landscaping. The camping complex covers a surface area of about $20{,}000\,m^2$ with wood, green terraces and parking places for a total of 25 cars. The CW area occupies only 3.5% ($700\,m^2$) of the camp surface area.

11.2. Results and Discussion

As reported in all the cited literature, there are several full-scale and pilot plants currently running in the Mediterranean countries. The different treatment schemes by CWs are nowadays going to be well known in terms of the different performances that can be obtained in the specific climatic conditions in countries like France, Spain, Italy and Greece. The few experiences available in the scientific literature for the other countries are actually promising because the tendency, going north to south, seems to be the reduction of the needed area for obtaining the same removal, at least for certain parameters.

The main general conclusion produced by the study of the literature related to the Mediterranean area is that CWs are surely an efficient wastewater treatment method in this climate and their application for any kind of water pollution problem has to be strictly linked to the treatment scheme choice and the sizing process. The

Table 11.1 General performances of constructed wetlands systems in the Mediterranean countries (range of removal percentages)

Constructed wasteland type	Organic content	Nitrogen	Ammonia	Total solids	Pathogens
Horizontal flow bed	73–99	23–67	18–76	59–96	94–99,999
Vertical flow bed	52–95	—	78–99	48–98	96–99,9
Surface-flow wetlands	11–63	21–76	15–82	36–67	90–99,999
Hybrid systems	86–99	43–89	85–96	72–84	98–99,9995
Vertical flow bed (raw wastewater)	82–99,7	66–98	85	95–99,8	—

operating experiences generally show a high rate of efficiency in the removal of organic content (BOD, COD), nitrogen (Ntot, NH_4^+, NO_3^-), total suspended solids (TSS) and pathogens (EC, FC, TC), both in secondary and tertiary treatment plants (Table 11.1).

Designs are often adapted to take account of different site characteristics, treatment goals and secondary benefits, such as the reuse of the treated wastewater or the provision of wildlife habitat. FWS are increasingly being favored as tertiary treatment, because of their cheaper investment costs and their higher wildlife habitat values. Subsurface-flow wetlands, however, tend to be more widely applied, due to their effectiveness at filtering out solids and removing BOD per unit land area. In general, the Mediterranean CWs systems seem to obtain better results, probably due to the more constant and warmer climatic conditions, in comparison to most of the other European experiences.

As an example of this assumption, the CW Haran-Al- Awamied, in Syria, can be cited. The system, running for a 7000 PE community, is composed by parallel HF reed beds, with a surface area equal to 0.43 m^2/PE and an hydraulic retention time (HRT) of less than one day: the organic matter removal obtained in this system is 84 to 85%, on average, both for BOD_5 and COD (Mohamed A, 2004).

A second example is represented by the pilot system realized in Marrakech in which an HF CW with a mean HRT of four hours and a mean hydraulic loading rate (HLR) of 60 lt/m^2/d has obtained, on average, 62% removal for COD and 97% removal for helmint eggs. These figures show that even with a very reduced retention time, comparing with the northern European experiences, interesting performances can still be obtained, which could be considered even more of interest in case of sanitary emergencies and lack of economical resources for facing the situation. The high performances reached in this research (Mandi, 1996) by the floating macrophytes systems are advising their usage in such climatic conditions (Figure 11.1), even though the biomass management can represent a negative aspect for ensuring a continued good performance of the treatment with a very low profile in maintenance.

In Figure 11.2, the mean removal percentages obtained over some years of monitoring for four HF reed beds in central Italy are shown (Conte et al., 2001) The two cited experiences in warmer climate countries highlight an increase of more than 10% in the mean efficiency yields in comparison to the Italian systems, which

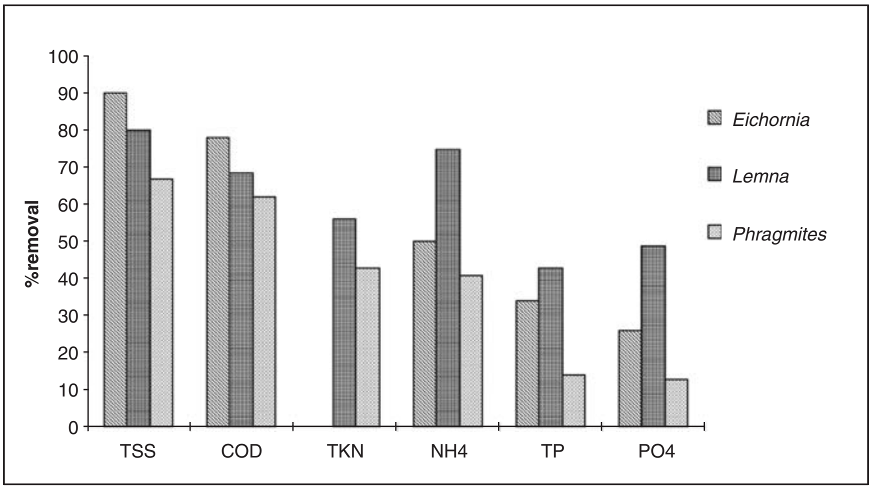

Figure 11.1 Performances in percentage removals between three parallel lines in Morocco (Mandi et al., 1996). The first two lines are floating macrophytes systems (*Eichornia* and *Lemna*) while the third one is a subsurface horizontal flow reed bed (*Phragmites*)

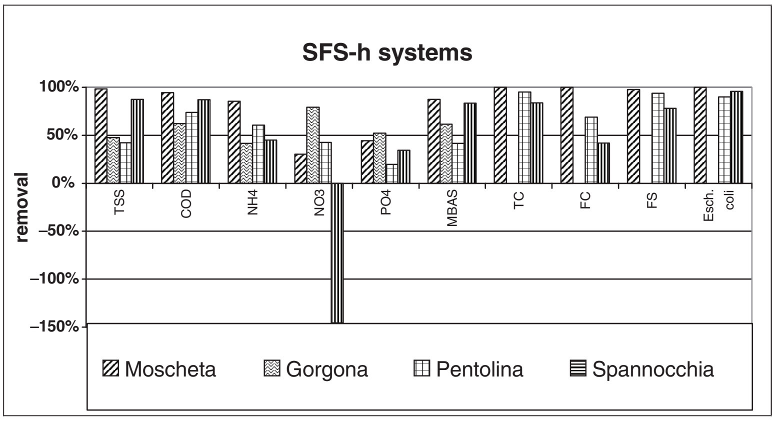

Figure 11.2 Performances in percentage removals for four horizontal flow reed beds located in central Italy and treating domestic or municipal wastewater (Conte et al., 2001)

seem themselves a bit more efficient in comparison to the northern countries like the United Kingdom, Germany and Denmark.

The results from the four TFs wastewater treatment systems demonstrate the potential of CWs as a suitable technology for treating wastewater from such facilities in remote areas. Very efficient COD (83 to 95%), TSS (68 to 93%), NH_4 (78 to 98%) reductions and pathogen elimination (3 to 5 logs) have been achieved in all four different sceneries (low and high tourist seasons). The performance of each

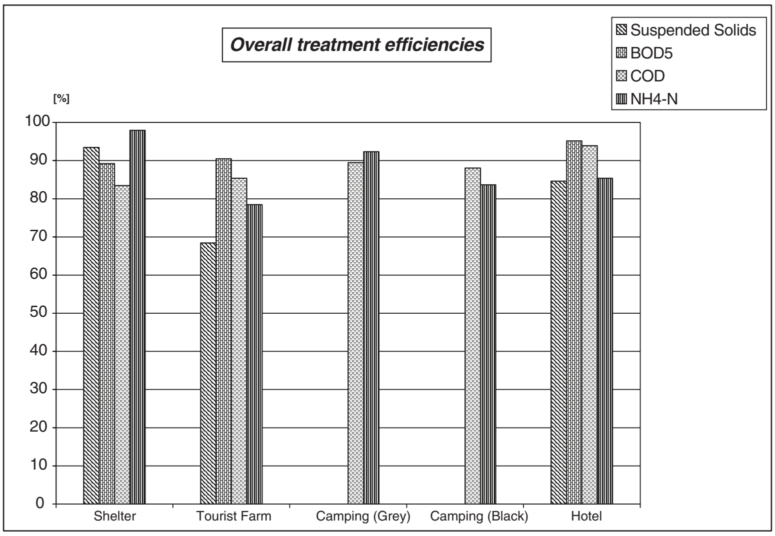

Figure 11.3 Overall performances of the four tourism facilities wastewater treatment plants expressed as removal percentages for TSS, BOD_5, COD and NH_4^+

scenario met the water quality requirements established at each individual trial location (Figure 11.3). When compared against other municipal or domestic wastewater treatment plants in Italy, the wetland treatment system influents have generally shown a more diluted chemical composition. A particular case is the inlet chemical composition of the segregated wastewater of the camping site. Here the concentrations are quite high in comparison to other reported values due to the concentrating effect obtained with the various water saving measures in operation. Given that due the additional water saving measures La Cava camping site had an average influent concentrations three-fold higher than measured at other sites, and the treatment efficiency of this wetland system was rather impressive. Moreover, the treatment performances of both HF CW for grey water treatment at the La Cava camping site and the hybrid CW established at the Certosa Hotel fulfilled the stringent Italian water quality limits for treated wastewater reuse (Figure 11.3). The La Cava camping site case study provides important information about the high efficiency of CWs when used for grey water treatment. The extremely high efficiency in COD removal (approximately 90%) obtained in this study, confirms good biodegradability of grey water observed by other researchers (Olson et al., 1968).

All four TFs have their operational peak in the hot season. Therefore, despite the small area per PE employed at all four wetland treatment sites (ranging between 1.2 and 3.6 m^2/PE), the COD removal rates achieved in our study were similar to those reported for larger area/PE ratio, operating in other northern European countries (Axler et al. 2000; Cooper et al. 1996; Vymazal et al. 1998). Design specifications of each of the four CW treatment systems, and their HLR and organic loading rates are presented in Table 11.2.

Table 11.2 Main features of treatment facilities

	Abetina Reale shelter	Baggiolino tourism farm	Relais Certosa Hotel	La Cava camping site
Load (PE)	140	100	30	80
Area horizontal flow bed (m^2)	160	—	108	116 (grey) 126 (black)
Area vertical flow bed (m^2)	180	126 (63 +63)	—	—
Horizontal flow bed depth (m)	0.7	—	0.7	0.7
Vertical flow bed depth (m)	0.9	0.9	—	—
Horizontal flow gravel size (mm)	5–10		5–10	5–10
VF filling media (sand + gravel)	Top 10 cm gravel Ø 6–12 mm Middle 60 cm sand Ø 0/4 Bottom 20 cm gravel Ø 30–40 mm	Top 10 cm gravel Ø 4–8 mm 30 cm sand Ø 0,06/4 Bottom 20 cm gravel Ø 4–8 mm	—	—
HRT (theoretical) (d)	3	1.5	3	2,9 (grey) 4,5 (black)
HLR ($m^3\ m^{-2}\ d^{-1}$) measured	HF: mean 0.15 min 0.11 max 0.23 VF: mean 0.15 min 0.10 max 0.18	mean 0.018 min 0.01 max 0.023	Mean 0.019 min 0.004 max 0.062	mean 0.015 (black) mean 0.073 (grey)
Organic loading rate (g COD $m^{-2}\ d^{-1}$) measured	HF: mean 17.5 min 1.5 max 50.3 VF: mean 2.0 min 0.5 max 5.9	mean 2.9 min 0.4 max 3.2	mean 4.2 min 0.14 max 23.2	mean 11 (black) mean 36 (grey)
Flow (m^3/d)	17–33	2–8	0.4–7	0.9–2.4 (black) 3–10 (grey)
Primary treatment	Imhoff 70 m^3	Imhoff + Buffer Tank	Imhoff + Degreasers	Imhoff
Operating since	January 2003	September 2003	June 2002	July 2004

Mean influent and effluent concentrations of the four treatment facilities are presented in Table 11.3. Despite the high variability in HRL (1.5 to 7.3 cm/d), all four facilities showed excellent COD removal efficiencies (average 90%). Two of the wetland treatment systems (Certosa Hotel and Abetina Reale Shelter) show a high percentage of ammonium removal, and are comparable to data presented in previous literature (Rustige and Platzer, 2001). Regarding the microbiological parameters, the three plants where data was available (except the camp site) showed very high performances, often more than 99.9% removal. These figures are comparable with values obtained in plants with lower HLR in Europe (Ottovà et al., 1996) and with results from experiments using similar HLR but performed in warmer and more constant climates (Conte et al. 2001; Khatiwada and Polprasert, 1999; Masi et al. 1999).

The relation between the drastic flow variations (experienced in all four treatment systems) and wetland treatment performances was most apparent in the second stage VF CW at the Certosa Hotel facility (Figures 11.4 and 11.5).

These results provide the most representative picture of the wetland systems treatment performances observed in all four systems. Despite the flow variations, the outlet concentrations were stable at low levels throughout the entire monitoring period.

These results demonstrate that using specific design treatment techniques, such as high volumes and HRTs, resulted in minimizing the problems related to flow fluctuations typical of TFs. Self-consistent bacterial communities that developed in CWs, indicated fast adaptation to the higher HLR employed.

11.3. Conclusions

The use of septic tanks and secondary treatment subsurface CWs for small populations is set to increase sharply in the Mediterranean countries. The use of hybrid designs incorporating both surface and subsurface-flow sections is now becoming more common, as well as the powerful combination of vertical and horizontal subsurface-flow systems. In addition, the treatment of raw wastewater by CWs, without a primary treatment, as suggested by the French experiences, looks very promising for the future. In general, the Mediterranean CWs systems seem to obtain better results, probably due to the more constant and warmer climatic conditions, in comparison to most of the other European experiences.

The larger number of tourists during warmer months and the fixed limits for the discharge of ammonia in open water bodies (due to the lower number of treated PE), represent the two recurrent conditions that occur in tourism facilities in remote areas of the Mediterranean basin.

The showed results demonstrate that apart from being cost effective, CWs provide excellent treatment for wastewaters with variable peak flows. In particular, considering the relatively small surface area employed (2 m^2/PE), the hybrid system at Certosa Hotel has shown the best treatment performance among the four systems

Table 11.3 Mean concentration of inputs and outputs of the four treatment facilities

		Shelter		Tourist farm		Hotel		Camping (black ww)		Camping (grey ww)	
		In	Out	In	Out	In	Out	In	Out	In	Out
# of samples	[-] min-max	3	3	16	16	17	17	7	7	7	7
Temperature	[°C] min-max	2–10	2–11	5–25	5–27	10–27	8–28	26–28	26–28	26–29	26–29
TSS	mg/L	76	4	57	18	26	4	—	—	—	—
Electrical conductivity	[µS/cm]	940	975	2744	1856	1115	1015	—	—	—	—
Alkalinity	[mmol/l]	4,2	3.0	11.6	9.3	7.0	6.2	—	—	—	—
COD	mg/L O_2	163	27	226	33	115	7	745	89	502	53
BOD_5	mg/L O_2	65	4	84	8	41	2	—	—		
TKN	mg/L N	51.0	23.00	79.0	31.00	53.0	21.00	190.0	31.0	2.5	1.10
Ammonium	mg/L NH_4^+	48.0	0.7	51.0	11.0	15.0	2.2	51.0	23.00	1.7	0.13
Nitrates	mg/L N–NO_3^+	2.20	13.7	0.86	1.0	2.00	15.1	1.60	42.1	0.32	0.46
Nitrites	mg/L N	—	—	—	0.05	—	0.09	—	0.03	—	<0.01
TP	mg/L P	8.1	0.7	6.6	1.2	5.1	0.3	2.4	0,02	6.6	0.3
Fec. coli	CFU/100 ml	2.3E + 06	9.9E + 01	2.9E + 06	3.1E + 02	4.2E + 06	1.3E + 03	—	—	—	—
Esch. coli	CFU/100 ml	1.4E + 06	4.4E + 01	6.6E + 05	1.1E + 02	3.4E + 06	1.8E + 02	—	—	—	—
Total coliforms	CFU/100 ml	2.9E + 06	1.4E + 02	4.9E + 07	3.2E + 04	8.0E + 06	3.8E + 03	—	—	—	—

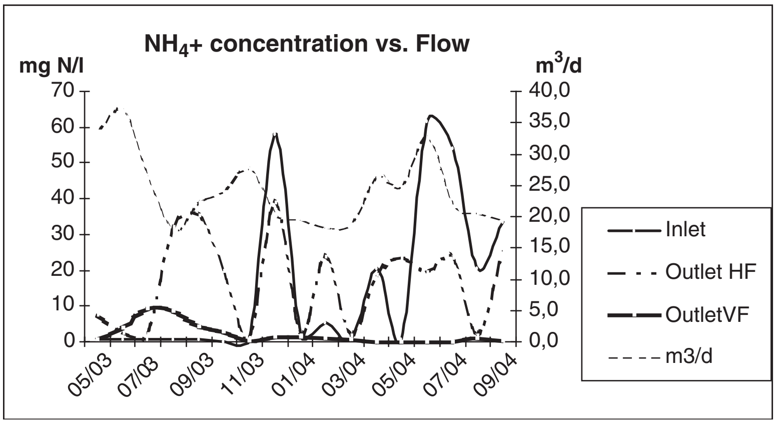

Figure 11.4 Inlet (after the primary treatment), outlet of horizontal flow constructed wasteland and outlet of the second stage vertical flow constructed wasteland at the Certosa Hotel facility versus the measured daily flow; ammonia ion concentrations

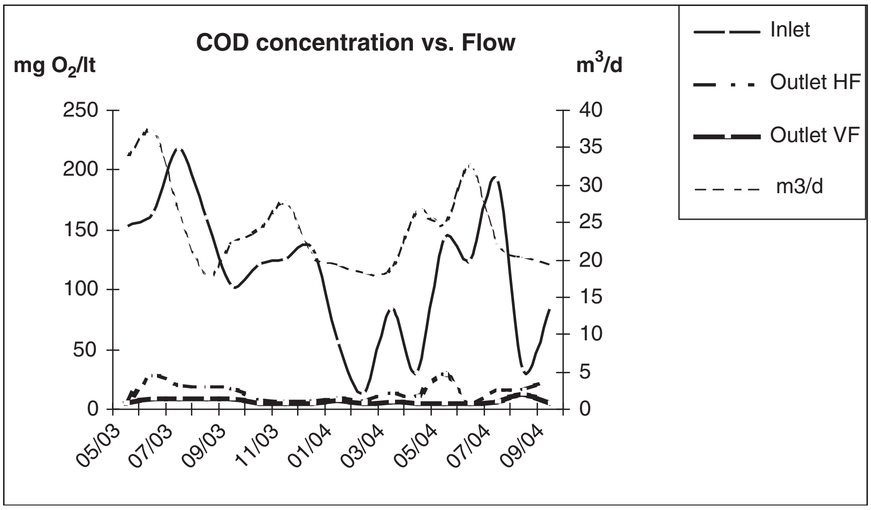

Figure 11.5 Inlet (after the primary treatment), outlet of horizontal flow constructed wasteland and outlet of the second stage vertical flow constructed wasteland at the Certosa Hotel facility, versus the measured daily flow; COD concentrations

presented in this study as TFs applications. Therefore, this configuration is recommended as a benchmark design for other warm climate TFs needing to improve and preserve the quality of open water bodies.

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References

Axler R, Henneck J, McCarthy B. (2000). Residential subsurface flow treatment wetlands in northern Minnesota. The 7th International Conference on Wetland Systems for Water Pollution Control.

Awad AM and Saleh HI. (2001). Evaluating contaminants removal rates in sub-surface flow constructed wetland in Egypt. The 2001 Wetlands Engineering & River Restoration Conference, August 27–31, Reno, NV.

Brenner A and Messalem R. (2002). Wastewater treatment and reuse in Israel: policy and applications. Small Wastewater Technologies and Management for the Mediterranean Area, Seville, Spain.

Bulc T. (2002). Development of CW in Slovenia. Small Wastewater Technologies and Management for the Mediterranean Area, Seville, Spain.

Bulc T, Zupancic M, Vrhovsek D. (2003). CW experiences in Slovenia: development and application. Constructed Wetlands: Applications and Future possibilities, Volterra, Italy.

Conte G, Martinuzzi N, Giovannelli L, Pucci B, Masi F. (2001). Constructed wetlands for wastewater treatment in central Italy. Water Sci Technol 44:339–343.

Cooper PF, Job GD, Green MB, Shutes RBE. (1996). Reed beds and constructed wetlands for wastewater treatment. WRc Seven Trent Water.

Garcia J, Morato J, Bayona JM, Aguirre P. (2004). Performance of horizontal sub-surface flow CW with different depth. Ninth International Conference on Wetland Systems for Water Pollution Control, September 26–30, Avignon, France.

Higgins JM, El-Qousey D, Abul-Azm AG, Abdelghaffar M. (2001). Lake Manzala Engineered Wetland, Egypt. 2001 Wetlands Engineering & River Restoration Conference, August 27–31, Reno, NV.

Khatiwada NR and Polprasert C. (1999). Kinetics of fecal coliforms removal in constructed wetlands. Water Sci Technol 40:109–116.

Lesavre J and Iwema A. (2002). Dewatering of sludge coming from domestic wastewater treatment plant by planted sludge beds. French situation. Eighth International Conference on Wetland Systems for Water Pollution Control, September 16–19, Arusha, Tanzania.

Liénard A, Duchène PH, Gorini D. (1995). A study of activated sludge dewatering in experimental reed-planted or unplanted sludge drying beds. Water Sci Technol 32:251–261.

Mandi L. (1996). The use of aquatic macrophytes in the treatment of wastewater under arid climate: Marrakech experiment. Fifth International Conference on Wetland Systems for Water Pollution Control, Vienna, Austria.

Masi F, Martinuzzi N, Loiselle S, Peruzzi P, Bacci M. (1999). The tertiary treatment pilot plant of PubliSer Spa (Florence, Tuscany): a multistage experience. Water Sci Technol 40:195–202.

Masi F, Bendoricchio G, Conte G, et al. (2000). Constructed wetlands for wastewater treatment in Italy: State-of-the-art and obtained results. IWA Seventh International Conference on Wetland Systems for Water Pollution Control, Orlando, FL.

Matos J, Santos S, Dias S. (2002). Small wastewater systems in Portugal: challenges, strategies and trends for the future. Small Wastewater Technologies and Management for the Mediterranean Area, Seville, Spain.

Mohamed A. (2004). Constructed wetland Haran-Al-Awamied, Syria. Data sheets for ecosan projects. Available at: http://www2.gtz.de/ecosan/download/ecosan-pds-015-Syria-HaranAlAwamied.pdf.

Molle P, Lienard A, Boutin C, Merlin G, Iwema A. (2004). How to treat raw sewage with CW: an overview of the French systems. Ninth International Conference on Wetland Systems for Water Pollution Control, September 26–30, Avignon, France.

Olson E, et al. (1968). Residential Wastewater. The Swedish Nation Institute for Building Research. Box 27S-102 52 Stockholm 27, Sweden.

Ottovà V, Balcarovà J, Vymazal J. (1996). Microbiological characteristics of constructed wetlands. Fifth International Conference on Wetland Systems for Water Pollution Control, I/13, IAWQ, Vienna, Austria.

Paing J and Voisin J. (2004). Vertical Flow CW for municipal wastewater and septage treatment in French systems. Ninth International Conference on Wetland Systems for Water Pollution Control, September 26–30, Avignon, France.

Papadopoulos A. (2002). The Nagref experimental station in the Thessaloniki Greece. Small Wastewater Technologies and Management for the Mediterranean Area, Seville, Spain.

Rustige H and Platzer C. (2001). Nutrient removal in subsurface flow constructed wetlands for application in sensitive regions. Water Sci Technol 44:149–155.

Shalabi M. (2004). CW in Croatian Adriatic Area. Ninth International Conference on Wetland Systems for Water Pollution Control, September 26–30, Avignon, France.

Yildiz C, Korkusuz AE, Arikan Y, Demirer GN. (2004). CW for municipal wastewater treatment: a study from Turkey. Ninth International Conference on Wetland Systems for Water Pollution Control, September 26–30, Avignon, France.

Vymazal J, Brix H, Cooper PF, Green MB, Haberl R. (1998). Constructed wetlands for wastewater treatment in Europe. Backhuys publ. Leiden. The Netherlands.

Chapter 12
Effect of Depth on the Performance of Algae-Based Wastewater Treatment Ponds

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Abstract A pilot-scale treatment plant consisted of a UASB-septic tank followed by three parallel pond systems each consisting of three stabilization ponds of equal depth and with the same hydraulic retention time (HRT) of 28 days. The setting was intended to investigate the effect of pond depth on the performance of algae-based ponds (ABPs). The depth of the ponds in the first, second and third systems were 90, 60 and 30 cm, respectively. The average ambient temperature throughout the experimental period was 24.5 °C. The influent chemical oxygen demand (COD) to the UASB was 1275 ± 84 mg/L and the influent COD concentration to each pond system was 331 mg/L ± 69 mg/L. The results reveal that the performance of the ABPs was inversely proportional to the depth. COD removal efficiencies for the shallowest and deepest ponds were 54.0 ± 1.1% and 51.6 ± 3.2%, respectively. Higher ammonium (NH_4^+) removal efficiencies were achieved in the shallowest pond instead of the deepest pond. The removal efficiencies of shallowest and deepest ponds 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 ponds were 45.4 ± 3.1% and 61.1 ± 4.5%, respectively. Negative removal efficiencies for total suspended solids (TSS) were observed in ABPs.

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12.1. Introduction

The Palestinian territories face serious environmental problems resulting from inadequate sanitation and wastewater treatment, originating from a lack of finance for construction and operation of conventional wastewater systems. The high percentage of non-connectors to the sewer system (76%) necessitates a great challenge to develop and introduce sustainable sewage collection and treatment systems (Zimmo, 2003). Algae-based ponds (ABPs) are low cost and efficient in producing high-quality effluent that enables wastewater reuse in irrigation (Mara et al., 1992). However, there are high algal concentrations in the effluent (Middlebrooks. 1995), causing severe clogging problems in advanced drip irrigation systems (Pearson et al., 1995). Moreover, ABPs systems are known for their high land requirement (LR), which is estimated at about 5 to 7 m^2/capita depending on the influent strength of the wastewater and the effluent quality requirements (Zimmo et al., 2000). Nevertheless, increasing the hydraulic retention time (HRT) by increasing the pond depth is expected to reduce LR (Soler et al., 2000).

In waste stabilization ponds with aquatic plant cover, a shallower pond depth (high surface area to depth ratio) reported to have higher performance than that with a greater depth (Smith and Moelyowati, 1999; Steen et al. 1998), which could be attributed to the enhancement of surface- and/or volume-related processes, such as sedimentation, ammonia volatilization and denitrification. In shallow ABPs, the amount of light available per pond volume is higher than that of deeper ponds. This would result in higher algal growth and, consequently, an increase in oxygen produced via photosynthesis, which favors different treatment processes such as nitrification (Reddy and DeBusk, 1987).

Silva (1995) obtained an ammonium removal efficiency of 81% in the pond system with a depth of 1.0 m. Regarding the effect of depth on chemical oxygen demand (COD) removal, no clear relation was found in the literature. Increasing the depth of waste stabilization ponds and the issue of anaerobic pretreatment is expected to play a significant role in decreasing the LR per capita to approach the guidelines. The effect of depth of stabilization ponds with anaerobic pretreatment on the performance of the ponds was not studied for the issue of nitrogen (N) removal efficiency. The main objectives of this study are to determine the effect of depth on N transformation and on the removal of major constituents of pollutants in algae-based waste stabilization ponds.

12.2. Material and Methods

To perform this study, a pilot-scale treatment plant was constructed within the site of the Al-Bireh Wastewater Treatment Plant (AWWTP) at Al Bireh city, 15 km northeast of Jerusalem in West Bank, Palestine. This study was performed during May to November 2004. The study was intended to test the effect of depth on N removal in duckweed-based ponds (DBPs). However, after the death of duckweed (Isayed, 2005), the system was altered to ABPs. The experimental period lasted for more than 24 weeks during May to October 2004. The climate was mostly typical of summer: the temperature range was 20 to 30 °C, humidity 60 to 70%, radiation 150 to 260 W/m^2, and wind speed 2.3 to 2.6 m/s.

The pilot-scale treatment plant (Figure 12.1) consisted of a UASB-septic tank of four days HRT (250 cm height, 64 cm diameter and 0.8 m^3 volume). This was followed by three parallel lines of aluminum circular ponds, each line consisting of three ponds of equal depth connected in series. The depth of the three ponds in the first, second and third lines were 90, 60 and 30 cm, respectively. The total HRT in each line was 28 days. The UASB-septic tank and the three lines of ponds series were interconnected by a 160-L distribution tank. Wastewater in the distribution tank was pumped to the lines via two peristaltic pumps. To maintain an HRT of 28 days in each treatment line, two pumps were used. One pump was pumping to the 90-cm deep ponds at a flow rate of 59.0 L/d. The second pump was calibrated to pump flow rates of 39.0 and 20.0 L/day into the 60 and 30 cm depth ponds, respectively. All of the ponds were made in a similar diameter (88 cm). Baffles at the outlet of each pond were installed to reduce short-circuiting and prevent the transfer of floating materials to the consecutive ponds. The surface organic loading rates in the first three ponds in each treatment line of the 90, 60 and 30 cm depth were calculated to be 118, 71 and 66 g COD/m^2.d, respectively.

12.2.1. Wastewater Sampling and Analyses

Grab samples of 300 mL were collected weekly from the influent (i.e., the effluent of the UASB-septic tank) and the effluents of the nine ponds. The samples were collected for conducting chemical analyses according to *Standard Methods*

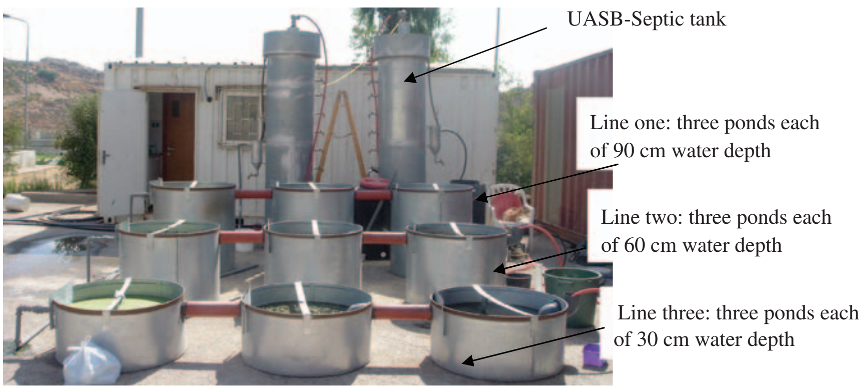

Figure 12.1 Photograph of the experimental pilot plant; the three lines of stabilization ponds and the UASB-septic tanks

(American Public Health Association, 1992). Temperature, pH and dissolved oxygen (DO) were directly measured *in situ* in the ponds and the influent. All tests were performed on the same day of sampling.

12.3. Results and Discussion

12.3.1. Physicochemical Properties of the System

The average wastewater temperatures in the ponds of the first, second, and third lines were similar throughout the experimental period. The average temperature was 22.1 °C. The DO concentration in the influent remained zero throughout the experimental period. DO concentrations in the effluent of the ABPs (90, 60 and 30 cm) were 3.7, 3.8 and 4.0 mg/L, respectively. The average values of pH at the effluent of the three ABPs lines (90, 60 and 30 cm, respectively) were 8.0 ± 0.03, 8.1 ± 0.03 and 8.3 ± 0.03, respectively.

12.3.2. Removal Efficiencies of TSS and COD

No effect for depth was detected on TSS concentration at the effluent in the pond system. This could be attributed to the small difference in depth among the three lines. Negative removal efficiencies were found in most of ABPs ponds. The average TSS concentrations in the effluent of the three ABPs lines (90, 60 and 30 cm) were 133.8, 88.5 and 153.1 mg/L, respectively. The negative removal efficiencies that were found in ABPs could be attributed to the intensive growth of algae in the ABPs due to the high availability of nutrients and light. Similar results have been reported in northeast Brazil (Pearson et al., 1995) and Palestine (Zimmo, 2003).

There was an inverse relationship between COD removal and depth. The average influent COD concentration was 331 mg/L. COD concentrations in the effluents of the three ABPs lines (90, 60 and 30 cm) were 159, 163 and 152 mg/L, respectively. The difference in removal efficiencies (from 51.6% to 55.9%) was insignificant. This fact encourages deep pond usage, as they are more economically feasible than shallower ponds. In our system, the effluent COD concentrations in the three ABPs lines complied with the Palestinian guidelines (PSI, 2003) for restricted irrigation.

12.3.3. Nutrient Removal

12.3.3.1. Total Phosphorus (TP) Removal

Average TP concentrations at the influent of ABPs was 12.8 mg/L. An inverse proportionality was found between depth and TP removal efficiency. TP concentrations at the effluent of the three lines (90, 60 and 30 cm) were 8.0, 6.8, and 5.4 mg/L, respectively. The corresponding removal efficiencies were from 37.6 to 57.6%, respectively. 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. TP removal by sedimentation of particulate phosphorus in decayed algae could have taken place in our system.

12.3.3.2. NH_4^+ Removal

Average NH_4^+ concentration in the influent to ABPs was 63 mg/L. Inverse proportionality was observed between NH_4^+ removal efficiency and depth. The average NH_4^+ concentration at the effluent of the three lines (90, 60 and 30 cm) were 31, 27 and 22 mg/L, respectively, with average removal efficiencies from 51 and to 65%. The higher surface/volume ratio in the system could favor denitrification (Al-Jabari 2003; Zimmo 2003). Moreover, the raised pH shifted the equilibrium between NH_4^+ and NH_3 toward the latter, which may have played a slight role in ammonia volatilization (Steen et al., 1998; Zimmo, 2003).

12.3.3.3. Total Kjeldahl Nitrogen (TKN) Removal

Similar to ammonium removal, inverse proportionality was observed between TKN removal and depth. The average TKN concentration in the influent of ABPs was 88.8 mg/L. Average TKN concentrations in the effluent of the three lines (90, 60 and 30 cm) were 47.2, 43.8 and 33.7 mg/L, respectively, with corresponding removal efficiency from 45.4 to 61.1%. Similar results were reported by (Reed 1995; Zimmo, 2003). Most of these studies attributed this result to the higher quantity of sediment occurred in the shallow ponds and higher rates of denitrification.

According to literature, the amount of sediment, nitrification/denitrification and ammonia volatilization are dependent on pH value (Ferrara and Avci, 1982). High N removal efficiencies in ABPs systems were reported despite the fact that algae (organic N) remain in the effluent. High pH values in ABPs are expected to play a considerable role in increasing the capacity of these ponds in N removal in our system. Moreover, nitrification could take place in all ponds where DO concentration is higher than the bottleneck value (0.5 mg/L; Taylor and Bishop, 1989).

pH and temperature in both periods were optimal for occurrence of nitrification process, according to Metcalf and Eddy (1991). However, nitrifiers are known to prefer attachment to solid surfaces (Focht and Verstraete 1977; Verhagen and Laanbroek, 1991). But the smooth ponds' surface made of aluminium is not expected to be favorable for attachment of nitrifiers. However, heterotrophic nitrification could take place in the anoxic zones, where organic N is decomposed by anoxic bacteria into ammonium and *ortho*-phosphate (Metcalf and Eddy, 1991; Zimmo, 2003). Nevertheless, a complete N mass balance is required to investigate these predictions in our system.

12.3.4. Sedimentation

Sedimentation was considered the main pathway for N removal in several studies (Avci, 1982; Zimmo, 2003), while it is found to be minor removal mechanisms in other studies (King, 1978; Pano and Middlebrooks, 1982; Silva et al., 1995; and Soares et al., 1996), the latter studies discussed ammonia volatilization as the main N removal mechanism. N removal as well as TP removal via sedimentation was not dependent on depth. This could be attributed to the low difference in depth among the three lines of ponds and long HRT. The color of the sediment was olive-green, which indicates the presence of decaying algae.

12.3.5. N Mass Balance

The mass balance equation (Equation 12.1) was used to calculate N budget in the three lines during the experimental period. The results for N mass balance are depicted in Table 12.1. N removal via ammonia volatilization, denitrification and uptake by algae were not measured in our system and were lumped as unaccounted for N in Equation 12.1.

$$N_{inf} = N_{eff} + N_{sed} + \text{unaccounted for nitrogen} \tag{12.1}$$

where

N_{Inf} and N_{Eff}: Influent and effluent N concentrations (mg/L)
N_{Sed}: N accumulation in the sediment (mg/L).

Unaccounted for N:N loss due to ammonia volatilization and denitrification and N uptake by algae (mg/L).

Most of N removed in our system could not be accounted for and referred to N leaving the system via ammonia volatilization, denitrification and uptake by algae. In the light of literature, a study proved experimentally that denitrification is one of the important mechanism for N removal (Zimmo, 2003). However, Camargo Valero and Mara (2006) proved that N uptake by algae was ranked as the major N removal mechanism.

In other studies, ammonia removal in waste stabilization ponds is mainly attributed to ammonia volatilization during periods of high temperatures and pH (Pano and Middlebrooks, 1982; Silva et al., 1995; Soares et al., 1996). According to Ferrara and Avci (1982) and Zimmo (2003), ammonia volatilization played a negligible role in N removal for the conditions prevailing in our system. However, the environmental conditions were optimum for the occurrence of denitrification (Metcalf and Eddy, 1991; Van Luijn, 1997; Verstraete and Alexander, 1973). Therefore, and according to Table 12.1, it is difficult to conclude which mechanism is the most important one responsible for unaccounted N.

12.3.6. Land Requirement

The results of N removal were projected to check for LR for a large-scale treatment plant operated as ABP. The removal curves for N, COD_{tot} and TP were used to determine the number of ponds required to achieve the desired value for each pollutant to achieve World Health Organization (WHO) standards for restricted irrigation.

If the pilot plant system was applied on a large scale, LR by the ponds to comply with WHO guidelines for restricted irrigation in (90, 60 and 30 cm depths) than the required area per capita to achieve the aforementioned guidelines for these depths are 3.9, 5.5 and 7.8 m^2/capita, respectively. Therefore, the use of the 90 cm-ponds for achievement of these guidelines is more feasible in terms of LR (3.9 m^2/capita). Better results were reported by Zimmo (2003) who cited an LR of less than 2 to 4 m^2/capita in a 90-cm pond depth. Zimmo (2003) reported that ABPs require 2.2 to 4.4 m^2/capita (depending on influent strength of wastewater and effluent guideline requirements).

12.4. Conclusions

- No significant differences in both DO concentrations and pH values were found among variable depths throughout the experimental period.
- No significant difference in COD removal efficiency was observed among the three ABPs lines.
- No relation was found between TSS removal and the depth. Negative TSS removal efficiency was observed in most ABPs.
- Inverse proportionality was found between TP removal efficiency and depth. TP removal efficiencies were 37.6 to 57.6% in ABPs lines.
- Higher TKN and NH_4^+ removal efficiencies were observed at ponds with lower depth. NH_4^+ removal efficiencies were from 51.2 to 64.5 %. TKN removal efficiencies were from 45.4 to 61.1 %.
- Deepest ABPs (90 cm) proved to be economically feasible compared to shallower ponds (60 and 30 cm) when considering COD removal efficiencies.

12.5. Recommendations

- Based on results presented in this chapter, deep ponds (90 cm) are highly recommended for large-scale application rather than shallower ponds (60 and 30 cm).
- 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 decreasing environmental problems.

- According to the achieved results in this research, ABPs are promising as a post-treatment unit after the anaerobic pretreatment unit in terms of capital cost land requirement.
- For achieving concrete results, another range of depths has to be tested, mainly depths of more than 90 cm.
- The performance of UASB-septic tank followed by pond system should be investigated during winter period at lower ambient temperature.

References

American Public Health Association (APHA). (1992). Standard Methods for Examination of Water and Wastewater, 18th edition. APHA, Washington, DC.

Camargo Valero MA and Mara DD. (2006). Nitrogen Removal in Maturation Ponds: Tracer Experiments with ^{15}N-Labelled Ammonia. IWA Specialist Conference on Waste Stabilization Ponds, Bangkok, Thailand.

Ferrara R and Avci C. (1982). Nitrogen dynamics in waste stabilisation ponds. J WPCF, 54:361–369.

Focht D and Verstraete W. (1977). Biochemical ecology of nitrification and denitrification. Adv Microb Ecol 1:135–198.

Isayed A. (2005). Effect of depth on nitrogen transformation in duckweed and algae-based ponds as a post-treatment stage for a UASB-septic tank. MSc thesis, Birzeit University, Palestine.

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

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

Metcalf and Eddy. (1991). Wastewater Engineering: Treatment, Disposal and Reuse, third edition. McGraw Hill, New York.

Middlebrooks E. (1995). Upgrading pond effluents: an overview. Water Sci Technol 31:353–368.

Pano A and Middlebrooks EJ. (1982). Ammonia nitrogen removal in facultative wastewater stabilisation ponds. J WPCF, 54: 344–351.

Pearson H, Mara D, Arridge H. (1995). The influence of pond geometry and configuration on facultative and maturation waste stabilisation pond performance and efficiency. Water Sci Technol 31:129–139.

Reddy KR and DeBusk TA. (1987). State of the art utilization of aquatic plants in water pollution control. Water Sci Tech 19:61–79.

Reed S, Crites R, Middlebrooks E. (1995). Natural Systems for Management and Treatment. McGraw-Hill, New York.

PSI. (2003). Palestinian Standards for Wastewater Treatment and Reuse.

Author: Please spell out PSI and provide publisher and location.

Silva S, de Oliveira R, Soares J, Mara D, Pearson HW. (1995). Nitrogen removal in pond systems with different configurations and geometries. Water Sci Technol 31:321–330.

Smith M 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 SA, de Oliveira R, Araujo ALC, Mara DD, Pearson HW. (1996). Ammonia removal in a pilot-scale WSP complex in Northeast Brazil. Water Sci Technol 33:165–171.

Soler A, Moreno M, Saez J, Moreno J. (2000). Kinetic model for deep waste stabilization ponds operating in batch mode.

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.

Van der Steen P, Brenner A, Oron G. (1998). An integrated duckweed and algae pond system for nitrogen removal and renovation. Water Sci Technol 38:335–343.

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

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

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

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

Zimmo O, Al-Sa'ed R, van der Steen P, Gijzen H. (2000). Comparison between algae-based and duckweed-based wastewater treatment: differences in environmental conditions and nitrogen transformations. Water Sci Technol 42: 215–222.

Zimmo O. (2003). Nitrogen transformations and removal mechanisms in algal and duckweed waste stabilization ponds. PhD thesis, University of Wageningen, Wageningen, the Netherlands.

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Chapter 13
Adapting High-rate Anaerobic Treatment to Middle East Conditions

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Abstract High-rate anaerobic technologies offer cost-effective solutions for sewage treatment in the Middle East and Palestine in particular. The sewage characteristics in Palestine are quite different from the values elsewhere and show solids contents of more than 1000 mg chemical oxygen demand (COD)ss/L and total COD values exceeding 2000 mg/L. While summer temperatures exceed 25 °C, temperatures may drop to below 15 °C in winter. Simple model calculations indicate that conventional upflow anaerobic sludge bed (UASB) reactor should be dimensioned on hydraulic retention times (HRTs) approaching 1 day to ensure methanogenic conditions in all seasons. Consequently, reactor volumes are three times the size of similar reactors in the tropics while the feasibility of the hydraulic flow patterns at such reduced rate are questionable. In an alternative approach, the UASB reactor was amended by incorporating a parallel digester unit for enhanced sludge stabilization and generation of active methanogenic sludge to be recirculated to the UASB reactor. The

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digester operational conditions were assessed by operating eight completely stirred tank reactors (CSTRs) fed with primary sludge. The results showed a high degree of sludge stabilization in the parallel digesters at a solids retention time of ≥10 and 15 days at process temperatures of 35 and 25 °C, respectively. The technical feasibility of the UASB-Digester combination was demonstrated by continuous flow pilot-scale experiments. A pilot UASB reactor was operated for 81 days at six hours HRT and 15 °C and was fed with raw domestic sewage. This period was subsequently followed by an 83 days operation period incorporating a parallel digester unit, which was operated at 35 °C. The UASB-Digester combination achieved removal efficiencies of total, suspended, colloidal and dissolved CODs of 66, 87, 44 and 30%, respectively. Preliminary model calculations indicated that a total reactor volume (UASB-Digester) corresponding to 8.6 hours hydraulic retention time (HRT) might suffice for sewage treatment in Palestine.

13.1. Introduction

Anaerobic digestion has been broadly recognized as the core of sustainable waste management. The upflow anaerobic sludge bed (UASB) reactor is the most widely and successfully applied high-rate anaerobic system for sewage treatment (Monroy et al. 2000). It has been greatly used for sewage treatment in tropical countries like India and Brazil where the ambient temperature is rather high and constant all year, ranging between 20 and 30 °C (Von Sperling and Chernicharo, 2005). The current challenge in anaerobic technology development is to amend the system to temperate climate conditions and/or to regions that are periodically subjected to low temperatures, like the hilly areas in Palestine and Jordan. Previous research has demonstrated that the performance of single-stage UASB systems at low temperatures (5 to 20 °C) is severely limited by the slow hydrolysis (H) of entrapped solids that accumulate in the sludge bed (de Man, 1990; Zeeman and Lettinga, 1999). This phenomenon is of particular concern when the reactor is fed with high concentrated raw sewage at low temperatures. Consequently, the excess sludge will accumulate, leading to a low solids retention time (SRT) and a concomitantly less stabilized sludge bed with a low specific methanogenic activity. The latter will result in a poor soluble chemical oxygen demand (COD) removal and an overall deterioration of the digestion process.

Recently, various researchers anticipated this limiting condition by removing the high suspended solids loads from the wastewater before soluble COD removal in an anaerobic upflow reactor. Zeeman et al. (1997) and Elmitwalli et al. (2002) proposed two-stage systems consisting of a high loaded first stage entrapping the suspended solids, followed by a methanogenic stage for soluble COD removal. However, in this set-up, the produced sludge from the high loaded first stage is, by definition, not stabilized and needs further stabilization in a separate digester. Based on the earlier theoretical considerations, Lettinga and Hulshoff Pol (1991) proposed an integration of a high loaded UASB with a digester for sewage treatment under low temperature conditions for treating both the water and the sludge.

This chapter describes the potentials of anaerobic sewage treatment in Palestine/the Middle East region and the feasibility, including experimental results, of the proposed UASB-Digester system in particular. Experiments are performed to optimize the digester design for sludge stabilization. After that, the technical viability of the UASB-Digester system for sewage treatment at low temperature is demonstrated.

13.2. Materials and Methods

13.2.1. Sewage Characteristics

Sewage characterization was conducted on time-interval composite samples collected from three different locations in Ramallah/Al-Bireh district in the West Bank. The samples were collected every two hours from 10 a.m. to 4 p.m. on Saturday, Tuesday and Thursday for three weeks in September. Temperatures of other sewage samples were measured four times during January, February and March.

13.2.2. Digester Optimization

For the digester optimization study, the UASB-digester system was simplified by eliminating the UASB reactor and by using primary sludge for feeding the completely stirred tank reactors (CSTR) digesters. Two sets of eight CSTRs, with 15-L working volumes, were operated to maintain SRTs of 10, 15, 20 and 30 days at temperatures of 25 and 35 ± 1 °C. The reactors were intermittently mixed at 30 rpm for 10 sec/min. The CSTRs were inoculated with primary digested sludge and fed with primary sludge collected once from the sewage treatment plant of Ede, the Netherlands, diluted to 20 g total solids per liter (g TS/l) and stored at 4 °C. The biogas, pH, Volatile Fatty Acids (VFA) and TS were monitored. After a period of at least three SRTs, the monitoring parameters were stable and the reactors were considered at "steady state." Then, within three weeks, four samples were analyzed in duplicate.

13.2.3. UASB Reactor and UASB-Digester System Pilot Plants

The pilot-scale experiments are divided in two experimental periods, I and II. In period I, a pilot-scale UASB reactor with a working volume of 140 L and a height, diameter of 325 cm and 23.5 cm, respectively, was operated for 81 days at 15 °C. Hereafter, the set-up was modified to a UASB-Digester system (Figure 13.1) by incorporating a CSTR type digester with a 1061 working volume, operated at 35 °C and mixed at 8 rpm. In period II, the UASB-Digester system was operated for 83 days.

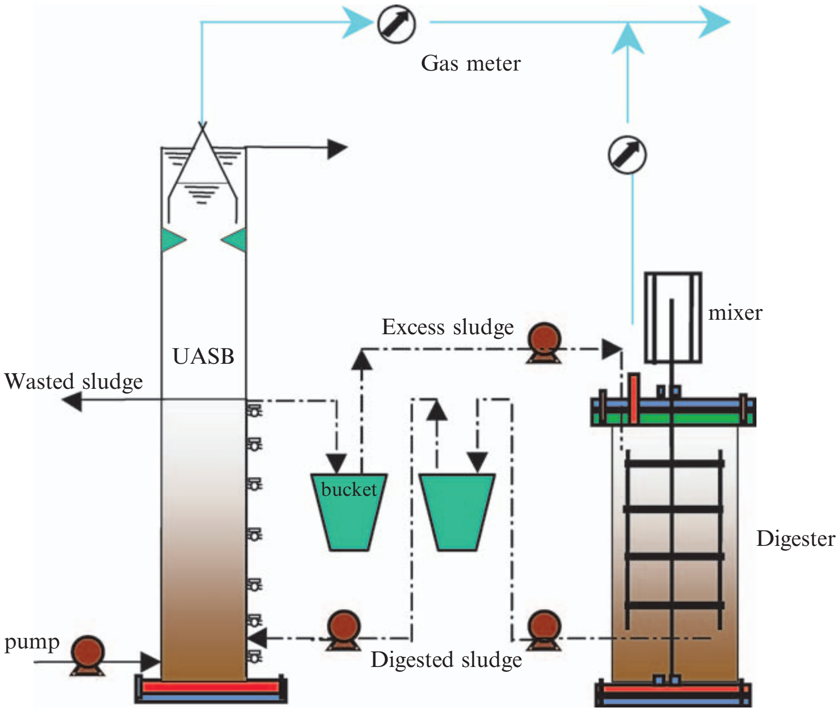

Figure 13.1 Schematic diagram of the UASB-Digester pilot plant

The UASB reactor was inoculated with sludge from a UASB reactor treating sewage from the town Bennekom, the Netherlands and the digester with digested primary sludge. The sludge bed height was kept at 190 cm by daily discharging the sludge that accumulated above this level. The discharged sludge was fed to the digester and the digester effluent was recirculated to the bottom of the sludge bed. Sludge was wasted from the upper part of the sludge bed 30 minutes after ending the recirculation process. The first 35 and 33 days of periods I and II, respectively, are considered start-up periods. The UASB reactor was fed with sewage from Bennekom that was collected in a combined sewer system. Composite samples were collected for 48 and 72 hours from the influent and effluent of the UASB reactor and stored at 4 °C.

13.2.4. Analytical Methods

Sewage COD was assessed using the micro-method described by Jirka and Carter (1975). Sewage samples were fractionated to suspended, colloidal and dissolved parts and VFA were determined in membrane-filtered samples by a gas chromatograph, as described by Mahmoud et al. (2003). The biogas content of CH_4 was determined using a gas chromatograph, as described by Mahmoud (2000). The capillary suction time (CST), total solids (TS), volatile solids (VS), total suspended solids (TSS), volatile suspended solids (VSS) and SO_4^{2-} were measured according to *Standard Methods* (American Public Health Association, American Water Works Association, Water Environment Federation, 1995). The total lipids (neutral and long chain fatty acids (LCFA)) and eluted LCFA were determined as described by Mahmoud (2002). Methane in the headspace was determined by the gas displacement method using 5% NaOH solution and methane from the CSTRs was determined using a 16% NaOH solution. Sludge total COD was determined according to *Dutch Standard Normalized Methods* (1969).

13.2.5. Calculations

The percentage of H, acidification (A) and methanogenesis (M) in each reactor were calculated according to the following equations.
CSTR experiments

$$H(\%) = 100\left(\frac{COD_{dis(eff)} + COD_{CH4} + COD_{LCFA(eff)}}{COD_{t(inf)}}\right) \quad (13.1)$$

$$A(\%) = 100\left(\frac{COD_{VFA(eff)} + COD_{CH4}}{COD_{t(inf)}}\right) \quad (13.2)$$

$$M(\%) = 100\left(\frac{COD_{CH4}}{COD_{t(inf)}}\right) \quad (13.3)$$

UASB reactor and UASB-Digester system

$$H(\%) = 100\left(\frac{COD_{CH4} + COD_{dis(eff)} - COD_{dis(inf)}}{COD_{t(inf)} - COD_{dis(inf)}}\right) \quad (13.4)$$

$$A(\%) = 100\left(\frac{COD_{CH4} + COD_{VFA(eff)} - COD_{VFA(inf)}}{COD_{t(inf)} - COD_{VFA(inf)}}\right) \quad (13.5)$$

$$M(\%) = 100\left(\frac{COD_{CH4}}{COD_{t(inf)}}\right) \quad (13.6)$$

Where: *inf*, influent; *eff*, effluent; $COD_{CH4} = CH_4$ as $COD_{(liquid\ form)} + CH_4$ as $COD_{(gas\ form)}$

13.3. Results and Discussion

13.3.1. Sewage Characteristics in Palestine

The results presented in Table 13.1 reveal that the sewage in Palestine is of high strength, since the mean values of COD, SO_4^{2-}, TSS and VSS are very high as compared to sewage strength classification in Metcalf and Eddy (2003). This high sewage

Table 13.1 Sewage characteristics of Ramallah City, Al-Bireh City and Al-Jalazoon Palestinian refugee camp. Standard deviations are presented between brackets

Parameters	No. of samples	Ramallah	Al-Bireh	Al-Jalazoon
COD Total	8	2,180 (663)	1,586 (125)	1,489 (251)
Suspended	8	1,096 (456)	919 (157)	725 (153)
Colloidal	8	323 (101)	274 (52.4)	327 (71.3)
Dissolved	8	761 (297)	393 (62.3)	438 (113)
COD_{VFA}	2	187 (12)	160 (3.1)	123 (25.4)
SO_4^{2-} as SO_4^{2-}	4	975 (742)	138 (9.9)	213 (57)
TSS	5	729 (197)	736 (67)	630 (234)
VSS	5	584 (209)	617 (66.1)	480 (148)
T-sew. Summer	8	30.9 (3.19)	25.8 (0.67)	23.4 (1.52)
Winter	4		13.13 (0.63)	
T-amb. Summer	8		27.1 (3.17)	
Winter	4		13.8(2.75)	

All parameters have been measured in duplicate and their units are in mg/L except sewage temperatures (T-sew.) and ambient temperature (T-amb.; °C).
COD, chemical oxygen demand; COD_{VFA}, COD for volatile fatty acids; TSS, total suspended solids; VSS, volatile suspended solids.

strength can be attributed to the low water consumption, industrial discharges and people's habits (Mahmoud et al., 2003). The results also show that the sewage temperature fluctuates between summer and winter from 13 to 26 °C. The results were used for model calculations (Zeeman and Lettinga, 1999) to elucidate the applicability of a single-stage UASB reactor in Palestine. The model calculations revealed that the application of a single-stage UASB reactor is only possible when a design Hydraulic Retention Time (HRT) exceeding 22 hours is applied, a value nearly three times the value of similar reactor types applied in Latin America and India. The required long HRT is fully determined by the sewage high solids content and low temperatures during wintertime (Mahmoud et al., 2003). Application of HRTs less than 22 hours would result in nonstable sludge beds and deterioration of the anaerobic treatment process. The theoretical considerations are experimentally confirmed by Hallalsheh and Wendland in Chapter 12 of this book. This required relatively long HRT has spurred the investigation of the UASB-Digester as an alternative technology.

13.3.2. Anaerobic Stabilization of Primary Sludge

The results presented in Tables 13.2 and 13.3 reveal that major sludge stabilization occurred at an SRT ≥ 10 and 15 days at process temperatures of 35 and 25 °C, respectively. H was the rate limiting-step of the overall digestion process, for the reactors operated at 35 and 25 °C, except for the reactor operated at an SRT of 10 days and 25 °C. At the latter conditions, M apparently is the rate-limiting step for the overall process. The dewaterability results of the digested sludge reveal the

Table 13.2 Percentage of hydrolysis (H), acidification (A) and methanogenesis (M) of the influent COD for primary sludge digested at 25 °C

SRT (day)	0	10	15	20	30
H_{COD}	10.0 (0.3)	23.9 (1.3)	40.7 (0.5)	41.4 (0.4)	42.1 (0.4)
A_{COD}	6.0 (0.46)	22.4 (1.2)	39.0 (0.5)	41.0 (0.4)	41.6 (0.4)
M_{COD}	0.0 (0.0)	10.9 (1.1)	39.0 (0.5)	41.0 (0.4)	41.6 (0.4)

SRT = 0 stands for the influent. Standard deviations are presented between brackets.

Table 13.3 Percentage hydrolysis (H), acidification (A) and methanogenesis (M) of the influent COD for primary sludge digested at 35 °C

SRT (day)	0	10	15	20	30
HCOD	10.0 (0.3)	41.1 (1.4)	45.4 (1.2)	47.2 (1.0)	49.2 (0.3)
ACOD	6.0 (0.4)	40.5 (1.4)	45.1 (1.2)	46.8 (1.0)	48.8 (0.3)
MCOD	0.0 (0.0)	40.5 (1.4)	45.1 (1.2)	46.8 (1.0)	48.8 (0.3)

SRT = 0 stands for the influent. Standard deviations are presented between brackets.

existence of an optimal SRT for dewaterability at 20 days and 15 days for 25 and 35 °C, respectively. The measured filterability constants are: 63.4 × 10^6 $kg^2.m^4.s^{-2}$ for 25 °C, 20 days SRT and 115 × 10^6 $kg^2.m^4.s^{-2}$ for 35 °C, 15 days SRT.

13.3.3. Anaerobic Sewage Treatment in a UASB-Digester System

13.3.3.1. Removal of COD Fractions

The main characteristics of the sewage used in this research are presented in Table 13.4. The low sewage concentrations in period II can be attributed to dilution with rainwater.

The mean influent and effluent CODt and the removal efficiencies including the COD fractions are presented in Table 13.5. It must be noted that a direct comparison between both systems is not possible as they were not operated in parallel in the same period, resulting in differences in sludge and influent characteristics. Nonetheless, the results indicate that the UASB-Digester system provides better removal efficiencies for CODt, CODss, CODcol, compared to the single-stage UASB reactor. The same is true for COD_{VFA} (results not shown). The effluent CODdis of the UASB-Digester system reached about 50 mg/L, a value that likely cannot be lowered further since this fraction is considered to be inert for anaerobic biodegradation. Therefore, despite of its good performance, the CODdis removal shows the relatively low value of 30%. The improvement of CODss and CODcol removal in the UASB-Digester system compared to the single-stage UASB reactor is likely due to prevailing improved digestion conditions. Elmitwalli (2000) showed that colloids in the sewage are highly biodegradable and that colloidal particles may be generated from the CODss during

Table 13.4 Characteristics of the influent sewage (Bennekom, The Netherlands)

+Parameter	Period I, UASB Avg (STD) <no samples>	Period II, UASB-Digester Avg (STD) <no samples>
CODt	721 (171) <16>	460 (122) <20>
CODss	398 (167) <16>	251 (100) <20>
CODcol	151 (46) <16>	124 (24) <20>
CODdis	172 (42) <16>	86 (37) <20>
COD_{VFA}	76 (38) <16>	34 (25) <20>

+ All parameters are in mg/l, CODt = COD total of raw sewage, CODss = COD of the suspended solids, CODcol = COD of the colloidal matter, CODdis = COD of dissolved matter, COD_{VFA} = COD of the volatile fatty acids.

Table 13.5 Influent and effluent CODt concentration and removal efficiencies (%) of CODt and fractions and organic loading rates (OLR) during anaerobic sewage treatment at 15 °C in a UASB reactor and a UASB-Digester system. Standard deviations are presented between brackets

		CODt		Removal			
	+OLR	Influent	Effluent	CODt	CODss	CODcol	CODdis
Reactor	gCOD/l.d	(mg/l)	(mg/l)	(%)	(%)	(%)	(%)
UASB-Digester	1.9	460	151	66	87	44	30
UASB	2.9	721	390	44	73	3	5

+UASB reactors

digestion. He also showed that it is very difficult to physically retain the CODcol fraction. Therefore, poor CODcol removal is generally observed during anaerobic sewage treatment, particularly at low temperatures.

13.3.3.2. Conversion

Results indicate that in the single-stage UASB reactor (period I) M was limiting the overall conversion of organic matter to methane. The effluent contained a high amount of soluble COD of 162(47) mg COD/L of which 80(40) mg COD/L was in the form of VFA. In contrast, in period II, H appeared to be the limiting step in the UASB reactor of the UASB-Digester system. Interestingly, the overall conversion in the UASB reactor of the UASB-Digester system is substantially higher than the single-stage UASB reactor, which very likely must be attributed to the better sludge characteristics and higher degree of sludge stabilization in the former. The percentages of methanized COD of the influent COD in the single-stage UASB reactor, the UASB unit of the UASB-Digester system and the overall UASB-Digester system were 21, 44 and 47%, respectively.

13.3.3.3. Sludge Characteristics

Quantative data of the daily wasted sludge reveal a significantly lower sludge production in the UASB-Digester system as compared to the single-stage UASB reactor, viz. 0.21 and 0.77 g COD/g COD removed, respectively. The VS and TS profiles of the sludge bed (Figure 13.2A) of the UASB unit of the UASB-Digester system show a declining trend in concentration from bottom to top of the sludge bed with clear stratification at the 40% height from the bottom of the reactor. The VS/TS ratio over the sludge bed height indicates that the sludge is equally stabilized over the bed (Figure 13.2B).

13.3.3.4. Methane Gas Recovery

The results of the gas composition analysis revealed a high methane content in the biogas released from the single-stage UASB reactor and the UASB unit of the UASB-Digester system, amounting to 73.4 (1.2)% and 82.9 (0.7)%, respectively. These values are similar to those reported in literature (Elmitwalli, 2000; Vieira and Souza, 1986). The recovered methane in the gas form from the single-stage UASB reactor, the UASB unit of the UASB-digester system and the overall UASB-digester system amounted to 0.09, 0.15 and 0.17 Nm^3/kg COD removed. About 50 and 55%

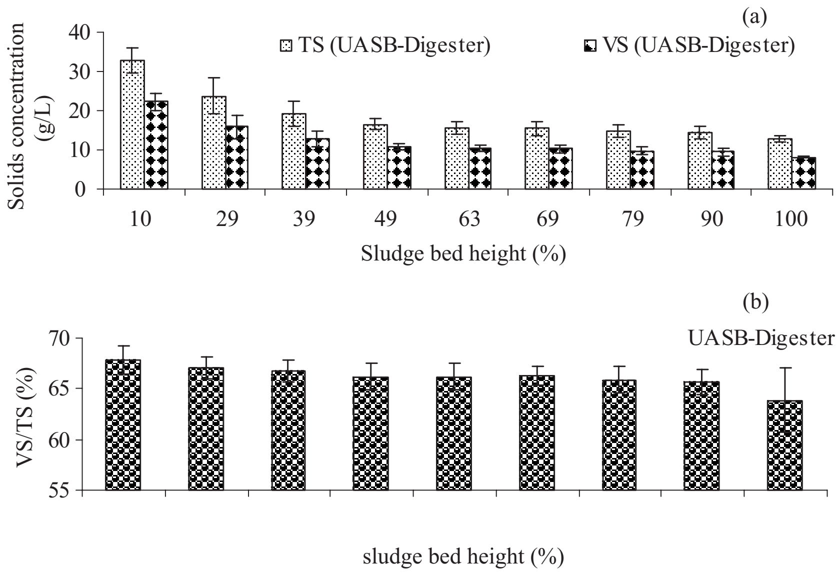

Figure 13.2 Sludge profile in a UASB reactor of a UASB-Digester system as VS and TS (a) and VS/TS ratio (b) along the sludge bed height expressed as percentages of the total bed height. The 0-sludge height stands for the bottom of the reactors

of the total produced methane gas in the single-stage UASB reactor and the UASB unit of the UASB-Digester system was dissolved in the effluent following Henry's law, respectively. The gas production values of the UASB-Digester system are similar to those reported by Lettinga et al. (1993) and by Uemura and Harada (2000), viz. 0.19 and 0.16 Nm^3/kgCOD removed, respectively.

13.3.4. Overall Research Discussion

The results of the research presented in this chapter show that the incorporation of a sludge digester in an anaerobic sewage treatment under conditions of low temperatures in a UASB reactor will substantially improve both the physical removal and biological conversion of the different COD fractions. Moreover, the integrated UASB-Digester system can be further optimized, especially with respect to size of the digester. The results of the CSTR experiments show that the optimum conditions for anaerobic conversions and improving the dewaterability of the sludge in the digester are a digester temperature of 35 °C and a sludge retention time of 15 days. In the experimental set-up of the UASB-Digester system, sludge from the top of the sludge bed in the UASB reactor, which is relatively low in solids concentration, was recirculated over the sludge bed. As the sludge profile in terms of VS/TS ratios does not indicate a clear change in stability, recirculation of sludge with the highest concentration (from the bottom of the UASB reactor) would result in a considerable decrease in required digester volume, viz. by approximately 50%. Based on the sewage characteristics in this research, this would mean that at an HRT in the UASB reactor of six hours, and thus a required UASB reactor volume of 0.25 m^3/m^3 sewage per day, the required additional sludge digester volume would be 0.071 m^3/m^3 sewage per day (i.e., about 30% of the installed UASB reactor volume).

The UASB-Digester system shows interesting perspectives for practical application. Considering the Middle Eastern conditions regarding climate and sewage characteristics, some further optimization is required. The reactor volume (consequently the applicable HRT) of the UASB reactor is mainly determined by the amount of active methanogenic sludge that can be retained in the system. This so-called methanogenic conversion capacity should suffice to convert the dissolved COD, which is either present in the influent or is produced in the reactor, into methane gas. Our present results show the feasibility of the UASB-Digester for treatment of high concentrated wastewaters that are (periodically) subjected to low temperatures. Technical optimization includes the fine-tuning of sludge circulated to and from the digester unit. Results of preliminary model calculations indicate that under Palestinian conditions, a digester volume of 0.107 m^3/m^3 sewage per day suffices, provided the sludge concentration in the recirculation flow to the digester is increased from 10 to 22 g VS/L, while operating the UASB reactor unit at six hours HRT. By increasing the sludge recirculation ratio by a factor 3 to 4, a reduction in the total required reactor volume of a factor 2.5 can be achieved.

13.3.5. *Potentials of Anaerobic Treatment in the Region*

The research presented in this chapter has been conducted to assess the feasibility of anaerobic pretreatment of domestic sewage under Middle East conditions, with special emphasis to the conditions in Palestine. Sewage characterization in Palestine shows extreme values for strength and suspended solids concentration compared to other regions in the world. Therefore, anaerobic pre-treatment can be regarded as ideal to reduce the organic pollution load before post-treating the sewage for reaching effluent discharge and/or reuse standards. The projected population for Ramallah and Al Bireh Governorate are estimated to reach 300.000 in 2007. At assumed average water consumptions of 70 L/cap.day, the total flow may reach about 20,000 m^3/day. Applying activated sludge would require an operational aeration energy demand of about 1.3 MW, assuming a specific aeration requirement of 1.2 KWh/kg COD removed, an average CODt of 2 g/L, and a pre-settling efficiency of 40%. In contrast, the proposed anaerobic pretreatment would produce a useful electric energy potential of 0.8 MW based on 65% CODt removal efficiency and a specific CH_4 production of 0.17 Nm^3/kg CODremoved, and an electric conversion efficiency of 40% using a modern combined heat power (CHP) gas motor. The difference is 2 MW, a considerable amount for a country that is deprived from any energy source. Furthermore, the generated energy could be used to upgrade the pretreated waste water up to irrigation standards, providing an alternative water resource for agricultural production. Obviously, the energy aspect is of interest for the entire region. Implementing anaerobic treatment for pretreating the sewage for Amman-Zarqa in Jordan yields a useful electric energy potential of 4-4.5 MW assuming a flow of 200.000 m^3/day, 1.5 kg COD/m^3 raw sewage, CODt efficiency of 65%, a very modest 0.12 Nm^3/kg CODremoved, and a 40% electric power CHP conversion efficiency. Completely non-comprehendible is the recent choice of the Water Authorities of Jordan to implement a conventional activated sludge system to treat the huge flow. Instead of the enormous energy surplus when implementing anaerobic treatment, the energy demand for aeration will be 6 to 7 MW, a difference of 10 to 11 MW!

The current and predicted increase in fossil fuel energy prices will be an important incentive to gear environmental engineering solutions to more sustainable alternatives. Cost-efficient technologies such as anaerobic pretreatment do not yet receive the attention they deserve. Particularly regarding the potential for reuse of treated water in agriculture, a wide-scale implementation of adequate treatment systems is of considerable interest. While the financial limitations obstruct large-scale implementation of the conventional approach, the anaerobic alternative offers such adequate solution to bridge this gap. Other advantages of anaerobic treatment coupled to agricultural reuse are the effective removal of heavy metals that are precipitated under the reducing conditions, the preservation of nutrients for ferti-irrigation, the potential to effectively filter pathogenic Helminth ova from the wastewater and the compactness, which prevents evaporation of valuable clean water to the atmosphere also resulting in a concomitant salinity increase. It must be noted that calculated with the design values of the Khirbet As Samra pond system in Amman (80.000 m^3/day), about 30 to 40% of the flow is evaporated.

13.4. Conclusions and Recommendations

- Anaerobic pretreatment of domestic sewage is considered an adequate solution for protecting surface waters from severe pollution and for being the first step in the reclamation of treated waters for agricultural production.
- Considering the high strength of the local sewage, anaerobic pretreatment is an energy source of interest, particularly when applied at a large scale.
- Because of the high sewage strength a single-stage UASB reactor requires an HRT approaching one day.
- The UASB-Digester system is a technically feasible alternative for the pretreatment of high-strength sewage at 15 °C.
- The UASB-Digester system offers all advantages of anaerobic pretreatment while the total required reactor volumes is similar to tropical UASB applications, working at HRTs of about eight hours.
- Based on our current results, assessing the long-term performance of a UASB-Digester system under Middle Eastern field conditions is recommended.

References

American Public Health Association, American Water Works Association, Water Environment Federation. (1995). Standard Methods for the Examination of Water and Wastewater, 19th edition. Washington, DC.

de Man AWA. (1990). Anaerobe Zuivering van ruw rioolwater met behulp van korrelslib UASB-reaktoren. Vakgroep Milieutechnologie, Landbouwuniversiteit, Wageningen, The Netherlands.

Dutch Standard Normalized Methods. (1969). The Netherlands Normalisation Institute, Delft, The Netherlands.

Elmitwalli TA. (2000). Anaerobic treatment of domestic sewage at low temperature. PhD thesis, Wageningen University, Wageningen, The Netherlands.

Elmitwalli TA, Oanh KLT, Zeeman G, Lettinga G. (2002). Treatment of domestic sewage in a two-step anaerobic filter/anaerobic hybrid system at low temperature. Water Res 36:2225–2232.

Jirka A and Carter MJ. (1975). Micro semi-automated analysis of surface and waste waters for chemical oxygen demand. Anal Chem 47:1397–1401.

Lettinga G and Hulshoff Pol LW. (1991). UASB process design for various types of wastewater. Water Sci Technol 24:87–107.

Lettinga G, Man AWA de, Last ARM van der, et al. (1993). Anaerobic treatment of domestic sewage and wastewater. Water Sci Technol 27:67–73.

Lettinga G, Rebac S, Zeeman G. (2001). Challenge of psychrophilic anaerobic wasteweater treatment. Trends in Biotechnol 19:363–370.

Mahmoud N. (2002). Anaerobic pre-treatment of sewage under low temperature (15 °C) conditions in an integrated UASB-digester system. PhD thesis, Wageningen University, Wageningen, The Netherlands.

Mahmoud N, Amarneh MN, Al-Sa'ed R, Zeeman G, Gijzen H, Lettinga G. (2003). Sewage characterisation as a tool for the application of anaerobic treatment in Palestine. Environm Pollut 126:115–122.

Metcalf and Eddy, Inc. (2003). Wastewater Engineering: treatment and reuse, 4th ed./revised by George Tchobanoglous, Franklin L. Burton and H. David Stensel, McGraw Hill, USA.

Monroy O, Fama G, Meraz M, Montoya L, Macarie H. (2000). Anaerobic digestion for wastewater treatment in Mexico: State of the technology. Water Res 34:1803–1816.

Uemura S and Harada H. (2000). Treatment of sewage by a UASB reactor under moderate to low temperature conditions. Biores Technol 72:275–283.

Vieira SMM and Souza ME. (1986). Development of technology for the use of the UASB reactor in domestic sewage treatment. Water Sci Technol 18:109–121.

Von Sperling M and CA de Lemos Chernicharo. (2005). Biological Wastewater Treatment in Warm Climate Regions. IWA Publishing, London.

Zeeman G and Lettinga G. (1999). The role of anaerobic digestion in closing the water and nutrient cycle at community level. Water Sci Technol 39:187–194.

Zeeman G, Sanders WTM, Wang KY, Lettinga G. (1997). Anaerobic treatment of complex wastewater and waste activated sludge—application of an upflow anaerobic solid removal (UASR) reactor for the removal and pre-hydrolysis of suspended COD. Water Sci Technol 35:121–128.

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Chapter 14
Options for Improving the Effectiveness and Potentials for a Sustainable Resource Recovery in Constructed Wetlands

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Abstract This chapter is divided into two parts, one presenting the options to improve the effectiveness of constructed wetlands (CWs) by focusing into their associated problems and one investigating the potentials of sustainable resource recovery. To deal with the problematic septic tank, one particular system initiated in France aims to treat raw household wastewater solely by CWs. It has been proved to function efficiently and hence there is no need to install a septic tank. Clogging is among the major operational concerns of CWs, which its likelihood of occurrence could be reduced by incorporating earthworms into the CWs. Earthworms were also found within real-scale CWs. The treatment efficiency could even be increased if the synergy of earthworms and the biological communities can be established. On the resource side, one should make use of the plants more effectively so that they will not be wasted. Instead of using the conventional plants, which are usually burnt after harvest, an alternative plant could be applied. Several plants such as bamboo or even the system of so-called wastewater garden can be used. Further, one might think about changing the paradigm of how one perceives wastewater by applying

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the ecological sanitation concept. CWs can provide a key to treat grey water. These options exhibit high potentials and can be adapted to the Mediterranean region.

14.1. Introduction

Among several technologies applied to treat household wastewater, subsurface-flow constructed wetlands (SFCWs) are one of the popular options due to its nature-orientated concept as well as its low cost requirement. Wastewater flowing to CWs normally requires some form of preliminary treatment, usually septic tank, in order to reduce its strength and potential of clogging inside the system (Reed et. al., 1995). SFCWs can be classified according to the type of feed pattern as horizontal SFCWs (HSFCWs) and vertical SFCWs (VSFCWs), in which both types have been applied for a long time in the Mediterranean region under various treatment steps and various configurations.

Based on the development of this technology, several problems have been presented such as the internal one concerning clogging, or the external one associated with related treatment components such as septic tank. This paper aims to present the options to mitigate such problems, which have already been established and widely applied, as well as ones that are expected to emerge. All would lead to one particular mean—to improve the effectiveness of CWs. Apart from the problems stated previously, one should also aim to make use of VSFCWs in a more sustainable way, such as increasing their biodiversity, or implementing the resource recovery aspect. The resources from the latter term can be either the plants or water itself. This can guarantee that they would not be wasted, but appropriately used. This would positively contribute to both ecological and economical aspects. As a result, the discussion will be based on two aspects, on both internal and external problems, and on further enhancement to recover the resources efficiently.

14.2. Improving the Effectiveness Based on the Problems Associated With Constructed Wetlands and Their Corresponding Treatment Component

14.2.1. Problems With Septic Tanks

As stated previously, one of the components in wastewater treatment that poses several problems is the septic tank, as in many cases the sludge accumulated is not disposed of properly, or there are leakages of the tank leading to groundwater contamination. In France, a particular VSFCWs design was initiated by SINT (La Société d'Ingénierie Nature et Techniques) with the backup provided by CEMAGREF (Institut de recherche pour l'ingénierie de l'agriculture et de l'environnement), which aimed to avoid installing this component (Boutin et al., 1997). The idea behind this system is that sludge management can be simpler by managing within the CWs in comparison with the conventional Imhoff or digesting tank. Recently, there should be more than 500 plants in France (Molle et al., 2005).

It is recommended to divide the system into two stages, in which the first stage consists of three alternately fed beds and the second stages consists of two alternately fed beds. The feeding phase for each receiving bed generally lasts for three to four days, after that the bed is needed to rest for twice this time to maintain unsaturated condition within the wetland bodies as well as to mineralize the organic suspended solid (SS) accumulated. The feed is in most cases regulated by siphons and the flows that depend on the wastewater production. The plant uses a specially designed siphon to maintain the hydraulic condition without an external energy source, provided the appropriate topography (Molle et al., 2005). Schematic of the first stage CW can be seen from Figure 14.1.

Concerning the area requirement, totally 2 m^2/population equivalent (PE) is required, in which 1.2 m^2/PE is attributed to the first stage, and 0.8 m^2/PE is attributed to the second stage. Gravel is used as the main layer for the first stage, whereas sand is used for the second stage. The substrate configuration of each stage is depicted in Figure 14.2.

According to its performance, the system is very efficient in terms of chemical oxygen demand (COD), total SS (TSS) and nitrification (Boutin et al., 1997). The sludge withdrawal should be performed approximately once every 10 to 15 years, and this has no subsequent effect to the regrowth of reeds from the rhizomes. One particular plant in Roussillon, in which the design was estimated for 1,250 person equivalent resulting in a total area of 1,550 m^2, has exhibited very good treatment performance. The average outlet concentration of BOD_5 for three years period is only 6 mg/L, as well as only 5 and 2 mg/L of Kjeldahl nitrogen and ammonium nitrogen, respectively (Liénard and Boutin, 2003). Nevertheless, it should be noted that the effluent nitrate concentration should be focused, as the lacks of denitrification processes are common in VSFCWs. Another system based on this principle in

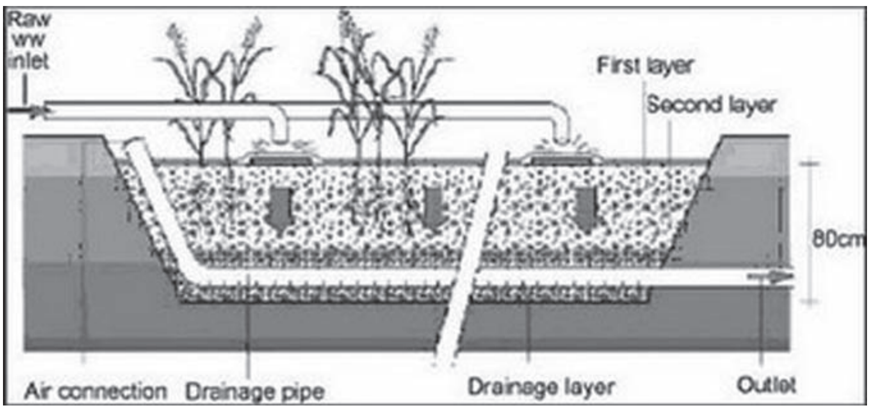

Figure 14.1 Schematic of the first stage French system (Molle et al., 2005)

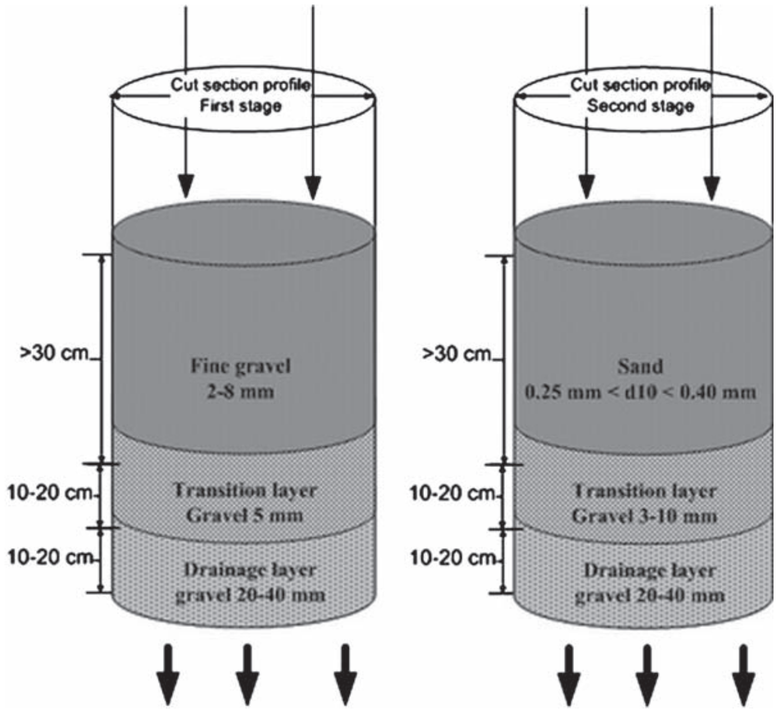

Figure 14.2 Schematic of the substrate profile in each treatment stage

France showed a considerable amount of nitrate in the final effluent from 20 treatment plants varying between 14 and 84 mg/L with a mean value of 43 mg/L (Paing and Voisin, 2005), which should be comparable to the system stated previously. A photo from part of the plants in France is shown in Figure 14.3.

With this system, no septic tank is required. As a result, the construction cost of septic tank can be neglected and the potential health risk for human as well as groundwater contamination can be avoided. Further, it should also be noted that the French regulation requires the treatment plant having a minimum distance of 300 m from the closest house to ensure that groundwater contamination will not be the case. The performance of this system would be more efficient in warm and dry climates without freezing periods, which is the case in the Mediterranean countries.

14.2.2. Problems With Clogging

One major operational problem with CWs, especially the VSFCWs, is concerning clogging, which has been pointed out by several studies (Blazejewski and Murat-Blazejewska 1995; Crites and Tchobanoglous, 1998). Clogging of wetland media will decrease the hydraulic conductivity, resulting in the surface flow of wastewater.

Figure 14.3 The first stage unit from the Phragmifiltre® of Saint Thomé (SINT, 2004)

This would negatively affect the overall treatment performance as well as its operation lifetime. Usually this situation occurs due to the accumulation of solid content in wastewater on the surface or within the substrate layer of VSFCWs and at the inlet of HSFCWs. Generally, the problem can be dealt with by increasing the rest periods between each feeding cycle in VSFCWs (Breen, 1997) and/or imposing the limitation of hydraulic loading rate as well as organic loading rate according to the guideline, which in the case of VSFCWs are 8 cm/d and 20 g COD/m^{2*}d respectively (ATV-DVWK, 2004). However, it will inevitably affect the design-loading rate of an influent as both options mean lowering the loading rate at constant wastewater quantity, which implies that more area is needed. Alternatively, earthworms could be a promising solution, as they by nature can ingest the organic matter and will then deposit their casts on or near to the surface. One study in Australia reported the presence of earthworms at the inlet of several non-clogged HSFCWs receiving grey water, and the lab-scale experiment reported that earthworms could move the sludge within the saturated bed to the surface (Davison et al., 2005).

The treatment system using earthworms has been widely applied to treat kitchen and animal wastes (Edwards, 2004), as well as municipal sludge (Vigueros and Camperos, 2002) and human feces (Shalabi, 2006). Earthworms fragment the waste substrate, enhance microbial activity and the rates of decomposition of the material, leading to composting or humification effect in which the unstable organic matter is oxidized and stabilized. This could ensure the digestion of clogging-related solids within the wetland. Moreover, earthworms might even enhance the process because they and aerobic microbes can act symbiotically to accelerate and enhance the decomposition of the organic matter (Loehr et al., 1988).

Figure 14.4 Earthworms found during the investigation

Although it seems possible theoretically, to decide whether it would be scientifically and technically sound to introduce earthworms into CWs, one should look further into the wetland itself whether earthworms do already reside there. An investigation was conducted and numbers of earthworms were found in a VSFCW in Luebeck, Germany. The detail about this unit is presented in Section 14.3.2. Earthworms were found to thrive even in a hot summer month, with ambient temperature more than 30 °C (Chiarawatchai and Otterpohl, 2006) as well as in a late autumn where the ambient temperature is as low as 5 °C. This is probably because the habitat provided by the reed and gravel helps alleviating the extreme condition. Hence, this means that they are among the biological communities inside the wetland and the use of earthworms as a further enhancement could be possible. Numbers of earthworms that have been found within the CW during the investigation can be seen in Figure 14.4.

The preliminary experiment in Germany indicated that earthworms should thrive with domestic wastewater. The survival rate of earthworms with respect to the mass after applying domestic wastewater for two weeks in lab-scale/mesocosms was found to be approximately 50%. This might be due to the fact that there was an excessive inoculants of earthworms compared to the food available for them, as one study concerning the vermicomposting of dry sludge predicted that 1 g of earthworms are able to consume approximately 200 mg of organic matter per day (Prince et al., 1981). In this case, 5 g of earthworms were put into the mesocosms, their numbers were undergone self-adjustment relative to the organic in wastewater, and hence resulted in this final earthworm biomass (Chiarawatchai and Otterpohl, 2006). Nonetheless, this concept is currently under further investigation.

14.3. Further Enhancement Based on the Resource Recovery Aspect

14.3.1. Alternative Plant Use for a Post-harvest Utilization

Reed (*Phragmites australis*) is among the most popular plants used in the CWs because of high tolerance and its abundance in several area of the world (Kadlec and Knight, 1996). Nevertheless, the harvest of reed, which in general is conducted at the end of the growing season, has been less focused. Burning of plants after the harvest is a common practice in several wetlands, in which this method represents wasting of the resources. In addition, there is no harvest at all in several cases, such as in the Czech Republic (Vymazal, 1996). Hence, it would be more ecologically sound if plants that possess more utilization options, are used rather than the conventional ones. One concern is that they have to be proved for exhibiting a similar impact on oxygen access to the soil at their root zones as other conventional species, except in VSFCWs where the oxygen contribution by plants is not important due to its naturally aerobic condition by intermittent feeding.

14.3.1.1. Vetiver

Vetiver is the plant that needs more water than other conventional wetland plants, such as *Phragmites australis* (Truong and Hart, 2001). It has long been used for the disposal of leachate in Australia due to its highly absorbent characteristics. There have been several units serving this purpose in Australia, China and Thailand (Truong, 2000). This plant represents several utilization options, namely as a compost, animal feed, pesticide, or mushroom cultivating medium concerning agricultural-based practices, handicraft products, traditional medicine, fragrance, as well as construction materials (Chomchalow and Chapman, 2003).

Moreover, it is even possible to utilize vetiver as an energy source, particularly when it is converted to ethanol. The dry vetiver leaves can be processed to produce ethanol (Kuhirun and Punnapayak, 2000). Its treatment performance is comparable to other conventional plants. In Thailand, this ubiquitous species has been promoted by H.M. the King of Thailand for treating wastewater and can efficiently treat swine wastewater (Kantawanichkul, 1999). As it has very high range of tolerance, it could be possible to implement the CW system utilizing this species in the Mediterranean countries, probably as a pilot project in case an opportunity arises. Although vetiver is not an autochthonous species in this region, it is considered as noninvasive species due to its sterility outside its natural habitat. Hence, there would be no problem concerning the invasion.

14.3.1.2. Bamboo

Certain bamboo species are considered to possess similar characteristics as reeds, but with some advantages, in which they can be transformed to a valuable product. Moreover, they can maintain green foliage all year round (De Vos, 2004). Among Mediterranean countries, this idea is presently applied in Portugal and France.

According to the International Network for Bamboo and Rattan (INBAR), bamboos are perennial plants of the grass family (*Poaceae/Gramineae*) and include more than 1,200 species worldwide in more than 100 genera. They are widespread throughout the subtropical and tropical regions worldwide, particularly in South, Southeast, and East Asia/as well as in tropical Africa and South America (Brazil). They tolerate warm/temperate climates such as in the Mediterranean region

(De Vos, 2000), are usually fast growing and highly productive species, and are one of the most widely utilized natural resources in the world. There are several utilization options for bamboo (De Vos, 2000; Whish-Wilson, 2002), which are biomass fuel (renewable source of energy), high-quality timber (wood for furniture or construction material for housing), high strength fiber, pulp and paper production and livestock forage.

There was a CWs study comparing the treatment efficiency of several conventional species with the unplanted unit as well as one bamboo species *(Bambusa multiplex)* under greenhouse conditions (Wolverton, 1983). Bamboo performed the poorest in BOD removal compared to all other systems including the unplanted bed. However, the bamboo filter was more effective in reduction of TKN and ammonia nitrogen than the unvegetated system, but less than three other species. Altogether, it was still/concluded that bamboos be suitable for use in CWs.

A further investigation was conducted within the scope of the research project “Bamboo for Europe” supported by the European Commission. Its main objective was to define and to overcome major problems and limitations to large-scale introduction of bamboo in the European Community. One option among them was to utilize two bamboo species in a CW and to compare their treatment efficiency with a standard wetland species (*Phragmites australis*). As a result, two bamboo species (*Phyllostachis nidularia* and *Phyllostachis heteroclada*) were planted in a vertical flow system treating primary effluent from septic tank or Imhoff tank. The system, constructed in Portugal, was in operation and complied fully with the regulations imposed by European standards. The study concluded that there exists a high potential for further developments in Europe and in other areas. It also suggested that bamboo stands could be irrigated with secondary effluents that are already treated to a sufficient extent in order to prevent groundwater from contamination and to produce valuable biomass yields suitable for industrial purposes (De Vos, 2000, 2004). Although Portugal is not considered as one of the Mediterranean countries, its climate is classified as the Mediterranean climate according to Köppen climate classification. As a result, there could be high potential for a successful adaptation of this system.

14.3.1.3 Wastewater Garden

If biodiversity is considered as one of the valuable resources, one should pay attention to the concept of a Wastewater Garden. In general, its aim is to enhance landscape beauty and to pave a way of sustainable greening urban areas by utilization of the renewable natural resources. Moreover, this approach ensures the diversity of biological communities as well as the landscape beauty. Some of this system in Mexico did provide comparable biodiversity indexes to the tropical forest systems and even surpass the natural wetlands (Nelson, 1998). The system utilizes locally available plants adaptable to the wetland condition and operates according to the HSFCWs concept. It requires an area of 3 to 4 m^2/PE. The length per width ratio is at least 4:1. The treated wastewater is used for subsoil irrigation. According to the results for one year from the pilot system in northern West Australia, the unit achieved 95% reduction of BOD_5 to 12.5 mg/L, as well as 95% reduction of TSS to 19 mg/L. Nonetheless, as it should be expected for a one-stage unit, the efficiency of nitrogen removal is decent, which its efficiency of total nitrogen removal is around 42% to 25.6 mg/L (Nelson, 2002). As nitrogen is a valuable resource, the way this system can effectively utilize nitrogen should receive more attention instead of focusing mainly on its removal. In this case, nitrogen is partly utilized to increase the diversity of CWs. One of the systems in West Australia is shown in Figure 14.5.

The wastewater garden shown in Figure 14.5 is located at Birdwood Downs' homestead, West Australia. It has been operated effectively for more than six years, in which its treatment performance is comparable to the pilot unit mentioned above. This garden supports heliconia, canna lilies, pandanus palm, coconut palm, plantain, elephant ear, papyrus and oleander. The effluent is used for the irrigation of trees within the area (Nelson, 2006). Similar systems have been built and operated in several countries, such as Mexico, Indonesia, Poland, the Bahamas, as well as in the Mediterranean country. The unit is located near Aix-en-Provence in France receiving domestic wastewater.

14.3.1.4. Fruit

Sugar cane, bananas, canna lilies, comfrey and sweet potatoes may be used to absorb nutrients and water and could be applied alternatively or purposely in CW. However, a scientific research concerning this matter is rare. Particularly the issue of heavy metal and other toxic substances has to be investigated.

Figure 14.5 Wastewater garden at West Australia (Nelson, 2006)

14.3.2. The Paradigm Shift Concerning Wastewater Perception

Several problems stated previously might not need to be solved, in case ones would change the way wastewater is perceived. To treat wastewater effectively, it should instead be separated into grey water, urine and feces and treated according to the characteristics of each stream (Otterpohl, 2001). Such a concept is called ecological sanitation. In this case, the benign grey water can be treated easily by CWs, yielding exceptional effluent quality as Table 14.1 shows the effluent parameters of the VSFCW at Flintenbreite in Luebeck, Germany. In this settlement, a source separation system of wastewater for the housing estate with 350 PE has been installed. The wetland built in this case makes more sense considering the fact that the characteristics of grey water in term of organic and nitrogen is more benign than the combined stream wastewater, therefore there is significantly less concern over operational problems. The required area is less than 2 m^2/PE. This system uses gravel as a substrate due to the idea that particle size should not be too fine in order to prevent clogging. The system, planted with reeds, is very effective in reducing organic and nitrogen, albeit the low level of phosphorus reduction that is comparable to others CWs. Figure 14.6 presents a photograph of this VSFCW.

14.4. Conclusion

There are several options and potentials to improve the effectiveness of CWs. The countries in the Mediterranean area have high possibilities to adapt these options, in which some have even already established such as the VSFCW with septic tank wastewater garden in France. However, it is worth or/stating that an obstacle or drawback is to be expected among each option. Further research has to be carried on. For the French system, effluent recirculation or third stage HSFCW could be applied to deal with the nitrate, if there is a need to remove it. The integration of earthworm into CW needs further optimization of its operation and demonstration of its effectiveness. The aspect concerning resource recovery requires significant effort from several stakeholders, for example how to and which parties will make

Table 14.1 Concentration of grey water before and after treatment in constructed wetland (Werner et al., 2005)

Parameter (mg/L)	Influent	Effluent
COD	502	59
BOD	194	14
Total N	12	2.7
NH_4–N	4.5	0.9
Total P	8	5.7
PO_4–P	7.6	4.8

Figure 14.6 Vertical flow constructed wetland for grey water in Flintenbreite, Luebeck

use of the plants after harvesting. An ecological sanitation concept can be applied to new settlements or demonstration villages in any Mediterranean countries aimed toward closing the loop of wastewater, in which CW can play a major role for treating grey water. Concerning all of the options stated previously, if their limitations have been overcome, there stands a high potential to be successfully applied within the Mediterranean countries.

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References

ATV-DVWK. (2004). Principles for the design, construction, and operation of constructed wetland for the treatment of municipal wastewater. Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall e.V., Guideline A 262 draft.

Blazejewski R and Murat-Blazejewska S. (1995). Soil clogging phenomena in constructed wetlands with subsurface flow. Water Sci Technol 35:183–188.

Boutin C, Liénard A, Esser D. (1997). Development of a new generation of reed-bed filters in France: First results. Water Sci Technol 35:315–322.

Breen PF. (1997). The performance of vertical flow experimental wetland under a range of operational formats and environmental conditions. Water Sci Technol 35:167–174.

Chiarawatchai N and Otterpohl R. (2006). Potentials of earthworm-assisted constructed wetlands: principals and preliminary results. 10th International Conference on Wetland Systems for Water Pollution Control, September 23–29, Lisbon, Portugal.

Chomchalow N and Chapman K. (2003). Other uses and utilization of vetiver. Third International Vetiver Conference, October 6–9, Guanzhou, China.

Crites R and Tchobanoglous G. (1998). Small and Decentralized Wastewater Management Systems. International Editions, McGraw Hill, Singapore.

Davison L, Headley T, Pratt K. (2005). Aspects of design, structure, performance and operation of reed beds—eight years' experience in northeastern New South Wales, Australia. Water Sci Technol 51:129–138.

De Vos J. (2000). Bamboo for Europe—final report EEC Brussels, part 2. European Economic Community, report no. FAIR-CT96-1747, pp. 1–35.

De Vos J. (2004). Potential of bamboo in phytoremediation—the Portugese technology. VII World Bamboo Congress, New Delhi.

Edwards CA. (2004). Earthworm Ecology, second edition. CRC Press LLC, CITY, FL.

Author: Please provide city for Edwards 2004.

Kadlec RH and Knight RL. (1996). Treatment Wetlands. CRC Press, Lewis Publishers, Boca Raton, FL.

Kantawanichkul S, Pilaila S, Tanapiyawanich W, Tikampornpittaya W, Kamkrua S. (1999). Wastewater treatment by tropical plants in vertical-flow constructed wetlands. Water Sci Technol 40:173–178.

Kuhiran M and Punnapayak H. (2000). Leaves of vetiver grass as a source of ethanol. An abstract of a poster presentation at ICV-2, January 18–22, Phetchaburi, Thailand.

Liénard A and Boutin C. (2003). Constructed wetlands for wastewater treatment and sludge dewatering: the French experience. Conference on Constructed Wetlands: Application and Prospects, June 17–19, Voltera, Italy.

Loehr TC, Lee YC, Liang JB, Tan D. (1988). Stabilization of liquid municipal sludge using earthworms. In: Edwards CA, Neuhauser EF, eds. Earthworms in Waste and Environmental Management. SPB, Academic Publishing, The Hague, pp. 95–110.

Molle P, Liénard A, Boutin C, Merlin G, Iwema A. (2005). How to treat raw sewage with constructed wetlands: an overview of the French systems. Water Sci Technol 51:11–21.

Nelson M. (1998). Wetland systems for bioregenerative reclamation of wastewater—from closed systems to developing countries. J Life Supp Biosphere Sci 5:357–369.

Nelson M and Tredwell R. (2002). New paradigms: Wastewater Gardens, creating urban oases and greenbelts by productive use of the nutrients and water in domestic sewage. UNEP Conference on Cities as Sustainable Ecosystems, Perth, Australia.

Nelson M, Tredwell R, Czech A, Depuy G, Suraja M, Cattin F. (2006). Worldwide applications of wastewater gardens and ecoscaping: decentralized systems which transform sewage from problem to productive, sustainable resource. International Conference on Decentralised Water and Wastewater Systems, Murdoch University, Australia.

Otterpohl R. (2001). Design of highly efficient source control sanitation and practical experiences. In: Lens P, Zeeman G, Lettinga G, ed. Decentralised Sanitation and Reuse, IWA Publishing, CITY, pp. 165–180.

Author: Please provide publisher's city for Otterpohl 2001.

Paing J and Voisin J. (2005). Vertical flow constructed wetlands for municipal wastewater and septage treatment in French rural area. Water Sci Technol 51:145–155.

Prince AB, Donovan JF, Bates JE. (1981). Vermicomposting of municipal solid waste and municipal wastewater sludges. In: Apppelhof MA, ed. Workshop on the Role of Earthworms in the Stabilization of Organic Residues, Vol. I Proc. Beech Leaf Press, Kalamazoo, MI, pp. 207–219.

Reed SC, Crites RW, Middlebrooks EJ. (1995). Natural Systems for Waste Management and Treatment, second edition. McGraw Hill, New York.

Shalabi M. (2006). Vermicomposting of faecal matter as a component of source control sanitation. PhD thesis, Institute of Wastewater Management and Water Protection, Hamburg University of Technology, Hamburg, Germany.

SINT. (2004). Phragmifiltre® plant of Saint Thomé. Ninth International Conference on Wetland Systems for Water Pollution Control, September 29, France.

Truong P. (2000). The global impact of vetiver grass technology on the environment. Second International Vetiver Conference, Thailand.

Truong P and Hart B. (2001). Vetiver system for wastewater treatment. Pacific Rim Vetiver Network Technical Bulletin No. 2001/2, pp. 1–26.

Vigueros LC and Camperos ER. (2002). Vermicomposting of sewage sludge: a new technology for Mexico. Water Sci Technol 46:153–158.

Vymazal J. (1996). Constructed wetlands for wastewater treatment in the Czech Republic the first 5 years experience. Water Sci Technol 34:159–164.

Werner C, Klingel F, Bracken P, Schlick J, Oldenburg M. (2005). Ecological housing estate Luebeck Flintenbreite. Data sheets for ecosan projects, pp. 1–3.

Whish-Wilson P and Maley S. (2002). Preliminary assessment of product & market opportunities for the bamboo industry in far north Queensland. Bamtek and Kleinhardt FGI Pty Ltd, pp. 1–66.

Wolverton BC, McDonal RC, Duffer WR. (1983). Microorganisms and higher plants for wastewater treatment. J Environ Qual 12:236–242.

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Chapter 15
Integrated Anaerobic–Aerobic Treatment of Concentrated Sewage

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Abstract The limited per capita share of fresh water in many Mediterranean countries has resulted in the production of concentrated sewage with average total chemical oxygen demand around 1,500 mg/L. The concentrated sewage exerts high energy demand for aeration on conventional activated sludge treatment systems. It can be calculated that aeration requirements needed for treating certain flow of concentrated sewage are three times higher than those needed for treating the same flow produced in most other countries in the world. Moreover, higher excess sludge amounts are produced, which results in extra operational costs. Therefore, concentrated sewage requires special attention and non-traditional management considerations. Integrated anaerobic–aerobic treatment of sewage is recognized as a sustainable and cost effective option. Anaerobic pretreatment can be used for upgrading existing overloaded centralized conventional treatment plants. It can also be applied when compact decentralized treatment systems are required. The present article shows that anaerobic pretreatment of concentrated sewage will reduce energy costs needed for operating wastewater treatment plants. Energy produced by anaerobic processes is surplus of aeration needs for activated sludge post treatment system. Moreover, 40% reduction in the total amount of excess sludge production

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can be obtained. The article also discusses the potential of applying constructed wetlands as a low-cost option for post-treatment of anaerobically treated effluent that can be applied (e.g., in rural areas).

15.1. Introduction

In Mediterranean countries with serious water shortage, the reuse of treated waste water is increasingly demanded for different purposes like agriculture and tourism. Jordan for example, is reusing more than 90% of its collected and treated wastewater for agricultural production. However, only 57% of the currently produced wastewater is collected and properly treated (Water Authority of Jordan, 2005) due to the unaffordable costs needed for infrastructure, especially in rural and remote areas. The existing applied wastewater management practices that depend on centralized collection and treatment systems form a real barrier against the wide spread of sanitation services and consequently the optimization of wastewater utilization as an additional water resource. For such conditions, there is an urgent call for the development of appropriate decentralized treatment options that require minimum construction, operational and maintenance costs. In fact, the existing wastewater management practices need critical assessment even when considering conventional centralized activated sludge treatment plants applied for urban areas in the region. The concentrated sewage (average chemical oxygen demand [COD] = 1,500 mg/L) produced in many countries as a result of very low per capita share of water, exerts three times higher energy consumption in activated sludge treatment plants compared with energy needed for operating such systems in other countries of the world. For example, energy consumption in activated sludge systems in Jordan range between 2.1 and 2.3 $kWhr/m^3$ of treated wastewater (Water Authority of Jordan, 2000) compared with 0.77 $kWhr/m^3$ of wastewater treated in Japan (Hu et al., 2000) for the same size of treatment plants. Energy requirement increases linearly with increasing organic loads of wastewater assuming efficiently operated activated sludge plant. Moreover, the amount of sludge produced using solely aerobic treatment systems is considerably higher compared with amounts produced in treating the same flow with low (300 mgCOD/L) or medium (500 mgCOD/L) strength wastewater. Taking this into account and knowing that actual costs associated with the operation of sludge treatment and disposal facilities constitute a considerable portion of the total operational costs, there exists a necessity for searching more efficient treatment concepts. This is of special importance when treating concentrated sewage, but also holds for low and medium strength sewage when sustainability is to be considered.

A foreseen flexible approach that could be used in order to overcome the previous limitations in sanitation practices is to consider integrated anaerobic–aerobic treatment options. Anaerobic treatment can be applied for up grading existing units or in the planned future decentralized and centralized treatment plants. The recent developments in high rate anaerobic sewage treatment technologies render their implementation feasible for most kinds of wastewaters and at different scales starting from house onsite sanitation up to centralized treatment options. High rate anaerobic treatment systems have the ability to separate hydraulic retention times (HRTs) and sludge retention times (SRTs). Consequently, these systems can be operated at very long SRT while keeping volumes needed for wastewater purification at a minimum. Among these systems, the upflow anaerobic sludge blanket (UASB) reactor is the most widely and successfully applied technology (Singh et al., 1996). The system was introduced as full-scale application for the pretreatment of low strength domestic sewage in some tropical regions (Wiegant, 2001). Currently, it is the system of choice used for sewage treatment in India and recently is being applied in some Middle East countries like United Arab Emirates and in many rural areas in Egypt (Figure 15.1). The UASB reactor separates solids from wastewater, but also acts as a sludge digester where the entrapped solids are converted into biogas. In this case, minimum excess sludge is produced and the resulting biogas can be utilized as an energy source. Integrating the system with the appropriate post-treatment technology represents the most sustainable sewage treatment option.

Different post treatment of anaerobically treated effluents can be used including activated sludge systems (Jordão and Volschan, 2004), stabilization ponds (Tessele et al., 2005), rotating biological contactors (Tawfik et al., 2005), trickling filters (Pontes et al., 2003) and constructed wetlands (CWs; El-Khateeb and El-Gohary, 2003; Sousa et al., 2001). Integrating the UASB reactor with CWs represents an attractive option for decentralized sewage treatment due to the low costs needed for construction, operation and maintenance of the system (Okurut et al., 1999). CWs are also suitable for isolated and remote areas because of their easy and low-cost operation and are being in use for post treatment of domestic sewage in many countries (Denny, 1997). Integrating the UASB reactor with activated sludge or trickling filters can be the systems of choice in case compact treatment systems are to be implemented.

Figure 15.1 Upgraded Fayyoom treatment plant in Egypt. The existing trickling filter is preceded by UASB reactor

This chapter discusses some research results of anaerobic concentrated sewage pretreatment using UASB reactor and presents the potentials seen for integrating the reactor with the appropriate aerobic post treatment system. Two integrated treatment options, namely UASB–activated sludge system and UASB–CWs system are specifically chosen to represent compact treatment options and voluminous low cost sewage treatment options respectively.

15.2. Anaerobic Pretreatment of Concentrated Sewage Using UASB Reactor

15.2.1. Treatment of Raw Sewage

Sewage in many Mediterranean countries is characterized by high concentrations of suspended solids and by fluctuating temperatures between summer and wintertime. Low temperatures prevailing during winter necessitates operating UASB reactors at SRTs higher than 100 days (Halalsheh et al., 2005) in order to achieve sufficient digestion of the entrapped suspended solids. Moreover, organic loading rate (OLR) rather than hydraulic loading rate is the main design parameter of the UASB reactor treating sewage with average COD concentration of 1,500 mg/L (Halalsheh et al., 2004). It was found that operating a conventional UASB reactor at 1.5 kg COD/m^3.d would result in average total COD removal efficiencies of 62 and 51% for summer and winter, respectively. It should be noted that 80% of the effluent COD is present in the form of stabilized sludge with VS/TS ratio of 0.5 all over the year. This means that a consequent removal of the suspended solids in a physical treatment step would significantly improve the total COD removal efficiency. The performance of a conventional 60 m^3 UASB reactor operated in Khirbit As-Samra in Jordan in comparison with other reactors is shown in Table 15.1, which shows that the performance of the UASB reactor operated in Jordan was comparable with results obtained from literature for reactors operated at the same OLR. With respect to the digestion process, Table 15.2 shows the hydrolysis, acidification and methanogenesis achieved compared with literature. Obviously, the UASB reactor operated in Jordan was superior in terms of conversion of organic material even at the lower temperature prevailed during wintertime. The results show the potential of applying the system for pretreatment of concentrated sewage as is discussed further in a later section.

15.2.2. Treatment of Pre-settled Concentrated Sewage

As part of the activities in the EMWater project, the UASB reactor was operated for the treatment of pre-settled wastewater at Hamburg University of Technology

Table 15.1 Performance of the 60 m^3 conventional UASB reactor of Khirbit As-Samra in comparison with results obtained from literature for full-scale reactors

Reference	Vol. (m^3)	Temp. (°C)	HRT (hours)	OLR Kg/m^3.d	COD_{totin} (mg/L)	COD_{ssin}* (mg/L)	% rem. COD_{tot}	%rem.* COD_{ss}
Halalsheh et al., 2005	60	25	24	1.5	1612	1184	62	55
Halalsheh et al., 2005	60	15	24	1.5	1419	1008	51	50
Lettinga, 2001	64	25	6	1.1	267	155	50–75	—
Vieira and Garcia, 1992	120	18–30	5–15	—	113–593	(44–512)	60	(70)
Van Starkenburg et al., 2001	4,660	20–31	8	1.10	400–450	(360)	49–65	(50–76)
Schellinkhout, 1993	3,350	24	5.2	1.9–2.0	330–450	(210–300)	45–50	—
Wiegant et al., 2001	11,200	26–29	6	—	—	—	61	(51)

*Values between parentheses represent SS.

Table 15.2 Percentage hydrolysis and methanogenesis (based on total influent COD) calculated for Khirbit As-Samra UASB reactor in comparison with values obtained from literature

Reference	Temp. (°C)	SRT (d)	Hydrolysis (%)	Methanogenesis (%)
Halalsheh et al., 2004	25	186	76	71
Halalsheh et al., 2004	18	137	46	42
Vieira and Garcia, 1992	18–30	—	—	36
Schellinkhout, 1993	24	—	—	45*
Lettinga, 2001	25	—	—	33–50

*Calculated based on the available data.

(Wendland et al., 2007). Sewage produced in Hamburg is characterized by high concentration of organic matter (average COD = 1,005 mg/L) due to limited drinking water consumption with an average value of 113 L/cap.d (HWW, 2005). The reactor was operated at ambient temperature ranging between 18 and 25 °C. A sedimentation tank was introduced ahead of the UASB reactor and the system was operated at different HRTs ranging between three and 30 hours. Results showed that the total COD removal efficiencies were 27±24% when the UASB reactor was operated at HRT of three to eight hours indicating an overloaded reactor (Table 15.3). Sludge washout was reported during this operational period. The performance of the system improved significantly at HRT of 24 to 30 hours. The average total COD removal efficiency was 60±11%. Obviously, at the prevailing operational temperatures, a pre-sedimentation step did not improve the performance of the UASB reactor compared with results obtained at Khirbit As-Samra in Jordan. In the later case, the average total COD removal efficiency achieved was 62% during summer time (Table 15.1). However, adequate solids removal from raw sewage is expected to improve the performance of the system at lower temperatures. Hydrolysis of

Table 15.3 COD and TOC removal with standard deviation (from Wendland et al., 2007)

	HRT of UASB reactor in hours (h)			
	3–8 h	8–16 h	16–24 h	24–30 h
COD total	27% ± 24%	46% ± 19%	47% ± 19%	60% ± 11%
COD diss	40% ± 12%	43% ± 21%	40% ± 25%	40% ± 20%
COD coll	61% ± 24%	62% ± 22%	40% ± 29%	66% ± 15%
COD ss	sludge washout		55% ± 36%	64% ± 39%
TOC	29% ± 30%	42% ± 18%	39% ± 24%	39% ± 15%
Expected COD total after additional settler	53% ± 24%	60% ± 16%	61% ± 13%	70% ± 6%

suspended solids is limited at low temperatures, which results in accumulation of solids in the UASB reactor and a following deterioration in the effluent quality. High COD_{ss} removal efficiency can be achieved using anaerobic filter (AF) reactor—rather than a sedimentation tank—where most of solids are removed and partly hydrolysed (Elmitwalli, 2000; Halalsheh et al., 2004). The HRT applied for the combined UASB–AF reactor is four plus eight hours, which is half of the HRT needed for a conventional UASB reactor (Halalsheh et al., 2004). The main disadvantage of such concept is that removed solids are not stabilized and needs separate digester.

15.3. Pretreatment of Concentrated Sewage in a Two-Stage Anaerobic Reactor

Based on results obtained for the conventional UASB reactor treating concentrated sewage and presented in Section 15.1, Halalsheh et al. (2004) operated a pilot scale anaerobic system consisting of UASB reactor followed by a filtration step using polyurethane sheets as the filtering media at the experimental station in Abu-Nusier treatment plant in Amman. The second step filter was operated at four hours HRT. The results showed average COD_{tot} and COD_{ss} removal efficiencies of 71 and 90%, respectively. Although, the anaerobic system was still in the start up period, the combination showed a high potential for concentrated sewage pretreatment. Excess sludge produced by the second step AF is stabilized with the characteristics shown in Table 15.4. The small amount of sludge discharged indicates the potential of the system in reducing the total amount of sludge produced compared with a solely

Table 15.4 Characteristics of the discharged sludge from the second stage anaerobic filter reactor

Parameter	Discharged sludge Kg VS/m^3 of $WW_{treated}$	TS g/L	SVI mL/gTS
Value	0.12–0.21	8–10	13.6–40

activated sludge system as is shown later. It could also be possible that excess sludge produced by the AF is recirculated back to the UASB reactor, and in this case the system can be kept closed with respect to sludge. However, this option is still under investigation.

15.4. Integrated Anaerobic–Aerobic Concentrated Sewage Treatment

15.4.1. Example on Integrated Compact Systems

To show the potential of UASB–AF system for concentrated sewage pretreatment, a comparison can be made between a conventional activated sludge system, which is adopted in many existing treatment plants in the Mediterranean region, and a system UASB-anaerobic filter step (UASB–AF) followed by aerobic activated sludge system. All assumptions needed for calculations are shown in Table 15.5. Results show that introducing anaerobic pretreatment a head of aerobic reactor would produce at least 40% less sludge than a conventional activated sludge system (Table 15.6).

Table 15.5 Assumptions made for comparison example

Parameter	Value
Flow	1000 m^3/d
Influent COD_{tot}	1500 mg/L
Influent BOD_5	900 mg/L
Primary clarifier removes 30% of the BOD_5	
OLR in aeration tank	0.5 kgBOD/m^3.d
Required power for aeration*	1.6 KWhr/kg$COD_{removed}$
Concentration of discharged sludge from aeration tank	8 gTS/L (practically it is even lower)
Concentration of excess sludge from primary clarifier	30 gTS/L

*Calculations are based on MetCalf and Eddy, 2003.

Table 15.6 Comparison between conventional activated sludge system and anaerobic–aerobic treatment system

	Conventional		Anaerobic–Aerobic			
	Primary clarifier	Aeration tank	UASB/AF*	Aeration tank	UASB/AF*	Constructed wetlands
HRT (hours)	2	30	24	12	24	>24 hours
Power for aeration KWh		−1832	+1593	−776	+1593	Only pumping
Sludge production m3/d	9	86	23	36	23	**

*AF: anaerobic filter described by Halalsheh et al., 2004.

** will be analyzed on the base of the projected pilot plant in Turkey within the EMWater project.

Moreover, degradation of the entrapped organic material under anaerobic conditions will produce 1,593 KWh for each 1,000 m^3/day of sewage, which is surplus of the energy needed for aeration in the aeration tank. However, the total volume needed for treatment is comparable in the two systems.

The presented figures clearly show that combining anaerobic–aerobic processes is a strategic choice not only for the future planned treatment plants in the region, but also for upgrading heavily loaded existing activated sludge treatment systems.

15.4.2. Example on Integrated Low-Cost Treatment Options: Two-Stage Anaerobic Reactors Followed by CWs

CWs act as biofilters combining physical, chemical and biological treatment and are especially suitable for low diluted water flows (Masi, 2005; U.S. Environmental Protection Agency, 2000) as many plants prove (SWAMP project funded by the European Union in 2002). Sousa et al. (2001) and El-Khateeb and El-Gohary (2003) showed that submerged as well as free water CW are suitable for the treatment of anaerobic effluents to total COD of 60 and 70 mg/L, respectively, when operated at OLR in the range of 5.5 to 10 gCOD/m^2.d. CWs were also efficient in removing fecal coliform with average removal of 99.99% achieved by Sousa et al., (2001). It can be assumed that the vertical flow and sub surface CWs are most efficient as they provide aerobic conditions. Due to high applied HRT, the operation of the system is reliable and can cope with fluctuating influent flows and temperature. CWs need a start up phase of about three months to achieve a reliable effluent standard in terms of COD and TSS reduction. For the case of concentrated sewage with average flow of 1000 m3/d, average total COD 1,500 mg/L and average removal efficiency of 70% achieved by a two-stage UASB–AF reactor, a total area ranging 45 to 82 ha is needed for sufficient treatment. When space is available, the combined system becomes an attractive low cost treatment option that needs minimum operational and maintenance costs.

15.5. Conclusion

Introducing anaerobic pretreatment for concentrated sewage using UASB–AF system ahead of conventional activated sludge system would result in a total reduction of 40% of excess sludge production compared with solely aerobic treatment system. Moreover, energy produced by anaerobic wastewater digestion was shown to be surplus of aeration requirements in a following activated sludge aeration tank. Integrating anaerobic–aerobic concentrated sewage treatment is superior when considering operational and maintenance costs of the treatment plant and in cases where compact systems are to be implemented.

Anaerobic concentrated sewage pretreatment should also be considered as a low cost treatment option for rural and remote areas. As the UASB–AF system can be applied at any scale starting from house on-site treatment, its application will ensure the acceleration of sanitation coverage for remote areas where sewerage systems are not affordable. When area is available, integrating the UASB–AF system with CW could be an attractive option due to the minimum operational and maintenance costs associated with these systems.

Acknowledgment Part of this research conducted in TUHH was funded by the European Community under the MEDA WATER PROGRAMME initiative ME8/AIDCO/2001/0515/59641-P044.

References

Denny P. (1997). Implementation of constructed wetlands in developing countries. Water Sci Technol 35:27–34.

Elmitwalli TA, Dun M, van Bruning H, Zeeman G, Lettinga G. (2000). The role of filter media in removing suspended and colloidal particles in anaerobic reactor treating domestic sewage. Biores Technol 72:235–242.

Halalsheh M, Sawajneh Z, Salhi A, Omari A, Fayyad M. (2004). AF/UASB and UASB/AF systems for strong sewage treatment. 10th anaerobic digestion conference, Montreal, Canada.

Halalsheh M, Sawajneh Z, Zu'bi M, et al. (2005). Treatment of strong domestic sewage in a 96 m^3 UASB reactor operated at ambient temperatures. Two stage versus one stage UASB reactors. Biores Technol 96:577–585.

Hamburg Water Works. (2005). Annual report of Hamburg Water Works 2004. Available at: www.hww-hamburg.de/hww_prod_engine.shtml?id=1110.

Hu HY, Goto N, Fujie, K. (2000). Statistical analysis of operating conditions and power consumption characteristics in small scale conventional activated sludge plants for sewage treatment. Environ Technol 21:1167–1172.

Jordão EP and Volschan I. (2004). Cost effective solutions for sewage treatment in developing countries- the case of Brazil. Water Sci Technol 50:237–242.

Lettinga G. (2001). Potentials of anaerobic pretreatment (AnWT) of domestic sewage under tropical conditions. Decentralized sanitation and reuse. Concepts, systems and implementation. IWA publications.

Okurut TO, Rijs GBJ, van Bruggen JJA. (1999). Design and performance of experimental constructed wetlands in Uganda, planted with *cyperus papyrus and phragmites mauritianus*. Water Sci Technol 40:265–271.

Pontes PP, Chernicharo CAL, Frade EC, Porto MTR. (2003). Performance evaluation of an UASB reactor used for combined treatment of domestic wastewater and excess aerobic sludge from a trickling filter. Water Sci Technol 48:227–234.

Schellinkhout A. (1993). Anaerobic treatment of domestic wastewater. The Bucaramanga case. International course on anaerobic wastewater treatment. Delft/Wageningen, the Netherlands.

Singh KS, Harada H, Viraraghavan T. (1996). Low-strength wastewater treatment by a UASB reactor. Biores Technol 55:22–29.

Tawfik A, Klapwijk A, El-Gohary F, Lettinga G. (2005). Potentials of using a rotating biological contactor (RBC) for post treatment of anerobically pretreated domestic wastewater. Biochem Engineer J 25: 89–98.

Tessele F, Fronza A, Heglert A, Monteggia LA. (2005). Low strength domestic wastewater treatment by the use of anaerobic (UASB) integrated to stabilization pond in subtropical climate. VIII Latin American workshop and symposium on anaerobic digestion. October 2–5, Punta del Este, Uruguay.

Water Authority of Jordan. (2000). Annual report of Water Authority of Jordan.

Water Authority of Jordan. (2005). Annual report of Water Authority of Jordan.

Wendland C, Al Baz I, Akcin G, Kanat G, Otterpohl R.(2007). Wastewater Treatment and Reuse in Mediterranean Countries, in Wastewater reuse—Risk Assessment, Decision-Making and Environmental Security. Springer, CITY.

Wiegant WM. (2001). Experiences and potentials of anaerobic wastewater treatment in topical regions. Anaerobic digestion for sustainable development. Water Sci Technol 44:107–113.

Chapter 16
Aerobic and Anaerobic Biotreatment of Olive Oil Mill Wastewater in Lebanon

Faud Hashwa(✉) and Elias Mhanna

Abstract Olive oil mill wastewater (OMW), known in Lebanon as Zibar, is one of the two by-products obtained during olive oil extraction. OMW represents a serious environmental pollution problem especially for underground and surface water. Aerobic and anaerobic OMW biotreatment processes were developed and improved and showed promising success. A bacterial mixture of 10 strains (*Aquaspirillum dispar, Bacillus cereus/thuringiensis, Brevibacterium otitidis, Klebsiella pneumoniae, Proteus penneri/vulgaris, Pseudomonas fluorescence* biotype F, *Pseudomonas marginalis, Pseudomonas mendonica, Pseudomonas sp. and Pseudomonas viridilivida*) and five yeast cultures (*Candida boidini, Candida memodendra, Candida mogii, Pichia haplophia* and *Sacharomyces ludwigii*) were isolated from OMW, purified and reused in OMW aerobic biotreatment. Pilot- (5,000 L) and industrial-scale (25,000 L) biotreatments were performed. After 14 days of pilot-scale biotreatment, a 69.6% in biochemical oxygen demand (BOD) and a 68.3% reductions in chemical oxygen demand (COD) values were achieved, while a 71.0% BOD and a 66.9% COD reduction were scored after 31 days of

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industrial-scale biotreatment. Anaerobic OMW biotreatment done at the experimental laboratory scale using omasomal juice as microbial starter achieved a reduction of 67.4% BOD and 65.9% COD with 37.2 L of biogas production per liter of Zibar after six weeks of incubation. The employed aerobic and anaerobic OMW biotreatment processes, developed at the LAU Biotechnology Labs, that achieved acceptable BOD and COD reduction rates and produced biogas, are low cost technologies and suitable for possible application in small rural olive mills in Lebanon and in the Middle East.

16.1. Introduction

In the Mediterranean basin and since early civilizations, olive trees were valued as symbols of wisdom, peace, abundance and glory.

The olive tree is most adapted to the Mediterranean climate and is one of its characteristic species (Figure 16.1). The Mediterranean region being poor in water resources, rendered the olive tree to become an economically beneficial target. Olive trees depend only on rainfall and don't need irrigation.

On the other hand, olive oil extraction is one of the most polluting agro-industrial sectors worldwide due to the formation of the olive oil mill wastewater (OMW). It is one of the two by-products obtained from olive oil extraction (Papadimitriou et al., 1997), known in Lebanon as Zibar.

16.2. Objectives of Study

The aim of this study is to improve Zibar treatment and bioremediation employing low cost technologies at laboratory and semi-industrial scales, the objectives are:

- Isolation, purification, identification and development of microbial cultures used in OMW biotreatment.

Figure 16.1 Hundreds of years old olive trees in Hasbaya, South Lebanon.

- Reduction of OMW organic load and toxic effects, thus minimizing its environmental threats.
- Development and operation of pilot aerobic and anaerobic treatment facilities at LAU Byblos campus and an aerobic industrial low-cost facility in Hasbaya, South Lebanon.
- Production of non-harmful products from the OMW biotreatment such as biogas and agricultural fertilizer.

16.3. OMW

OMW accounts for up to 50% (v/v) of the total olive oil mill output, while the olive oil accounts for 20% (v/v) and the remaining 30% (v/v) are the solid residue (Baccari et al., 1996; Hamdi et al., 1991; Pérez et al., 1998) known in Lebanon as Jift. More than 7.3 million tons per year (t /y) of OMW is generated worldwide, coming from the extraction of around 1.8 million tons per year of olive oil from 9.7 million tons of olives (Feria, 2000). Many studies report that OMW is a major pollutant to surface and ground water resources in the Mediterranean basin. It is one of the least biodegradable natural compounds due to its high phytotoxic phenols content, colored organic substances and high organic matter concentration (Al Khudari, et al, 2004; Paredes et al., 1986; Saez et al., 1992). OMW is a strongly smelling phytotoxic waste known for its antimicrobial activity (Pérez et al., 1992; Tuncel and Nergiz, 1993). General approximated constituents of the OMW could be established (Table 16.1) where each constituent alone has a negative environmental impact if disposed without treatment with the phenolic OMW being the most polluting constituents (Ragazzi and Veronese, 1989). OMW biological oxygen demand (BOD) could reach values as high as 100 g/L while the chemical oxygen demand

Table 16.1 Characteristics and composition of olive oil mill wastewater (Lopez, 1992; Skerratt and Ammar, 1999)

Olive oil mill wastewater characteristic	Value
Color	Intensive violet–dark brown up to black
Odor	Strong specific olive oil smell, foetid smell
pH	4.5–6.0
Water content	83–92%
Organic and volatile material	7–15%
Mineral solids	1–2%
Residual oil	0.3–10.0%
Total sugars	2–8%
Reducing sugars	1–8%
Polyalcohols	1.0–1.5%
Protein	0.5–7.5%
Pectins and tannins	1.0–1.5%
Phenols	17%
Suspended solids	5–35 g/L
BOD_5	65–70 g/L
COD	40–200 g/L

(COD) around 200 g/L (Ubay and Ozturk, 1997). The environmental problems linked to OMW are not confined to water, but some phytotoxic effects were also observed especially on plants germination and premature fall of the fruit and vegetables senescence (Feria, 2000).

New measures were implemented with many chemical, biological and physical methods being suggested due to the great failure of the land spreading and lagooning methods (Hoyos et al., 2002). With yearly increases in olive oil production and thus more OMW release into the environment no practical solution to the Zibar problem exists yet. Each olive mill may adopt a different system of waste treatment. The most adopted solutions include: adsorption, aerobic treatment, anaerobic treatment, composting, decolorization, drying or evaporation, electrolysis, filtration, membrane filtration, ultrafiltration, precipitation or flocculation, lagooning, thermal treatment, treatment by fungi and wet oxidation; and these solutions could be used separately or applied in combination (Improlive project A2, 2000).

16.3.1. Distribution of Olive Oil Production

Olive oil production is almost concentrated in the Mediterranean basin (Ettayebi et al., 2003). Regions dominated by a Mediterranean climate such as California, South Africa (Cape) and some regions in Mexico, Argentina and Australia are also involved in olive oil production. According to the United Nations Convention on Trade and Development records based on data from Food and Agriculture Organization Statistics (FAOSTAT), eight countries were found to produce 96% of the world olive oil in 2003 with the rest of the world contributing to the remaining 4% as shown in Figure 16.2 (FAOSTAT, 2003).

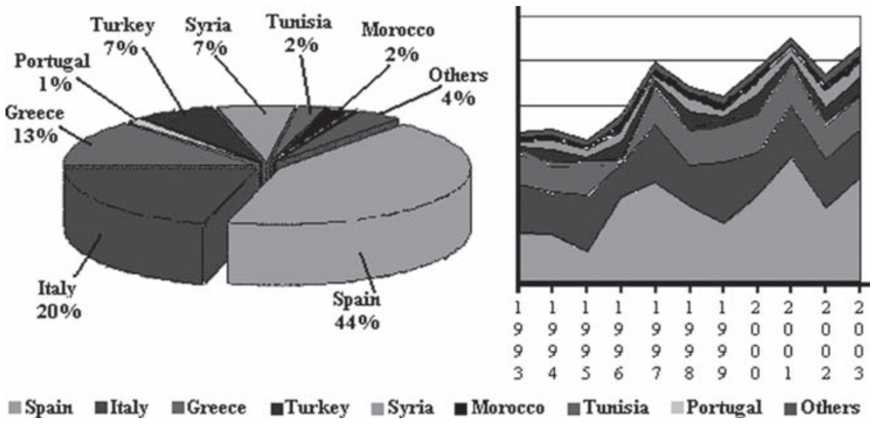

Figure 16.2 Main olive oil-producing countries in 2003 and the evolution of world production during the last 10 years (Food and Agriculture Organization Statistics, 2003)

Figure 16.3 Crude Zibar discharge into the Hasbani River by a local olive mill (Hasbaya, Zibar project, LAU)

Lebanon is one of the small olive oil producers. According to the FAOSTAT database, Lebanon produces around 6,000 tons per year of olive oil (Improlive project A2, 2000). The sector is mostly family owned and inherited from generation to generation as is the case with other Mediterranean countries; many Lebanese depend fully or partially on the income they generate from the olive oil industry; however, what distinguishes Lebanon from other countries is the absence of governmental supervision. Strict laws are not available to protect the environment from the OMW damages and if they exist are not implemented (Hashwa, 2003). It is estimated that around 1.6 million L of Zibar (BOD about 50,000 mg/L) is produced in Lebanon per year and disposed directly in the areas surrounding the olive mills in digs, wells, rivers, lakes and valleys (Al-Khudary et al., 2004). However, all alternatives used will lead to the seepage of OMW into the underground water table or other water sources (Hashwa, 2003).

Despite the existing environmental laws and regulations, disposal of Zibar into the nearest lands, streams and rivers by many olive oil mills, is still being practiced to avoid bearing additional economic costs of olive oil production in Lebanon (Figure 16.3).

16.3.2. Components of OMW

OMW constituents are similar in all olive oil mill production, but they differ in concentrations. Approximated constituents of the OMW and their relative concentrations are given in Table 16.1.

Each constituent of the OMW alone presents a threat to the environment and when combined together the threat is amplified. Tannins for example, that come from the olive skin, are not harmful for humans, animals or plants; however, they add a dark black–brown coloration to the water affecting light penetration. As a consequence, photosynthesis of many aquatic plants is impaired, respiration by aquatic organisms is diminished and the vision of many aquatic animals is weakened, leading to inability to locate their food sources. In the soil, OMW, containing acids, minerals and organics, could destroy the cationic exchange capacity of the soil. This in turn may inhibit growth of microorganisms, the soil–air and the air–water balance and consequently reduces the soil fertility (Improlive project A2, 2000).

16.4. Methodologies, Applications and Results

Several trials of Zibar biodegradation were performed at different scales. We report here on the results obtained with the pilot- and industrial-scale trials while the Lab and intermediate scale trials were reported elsewhere (Mhanna, 2006).

BOD, COD and pH values were measured and monitored through all trials to evaluate OMW properties changes before, during and after the treatment processes. Temperature and dissolved oxygen (DO) were additionally monitored to ensure optimal growth conditions for the microbial cultures.

16.4.1. Microbial Cultures

In most trial treatments, mixed bacterial and yeast cultures were used as inocula for starting and expediting the biodegradation process. The individual bacterial and yeast species were isolated in our laboratory from fresh crude Zibar.

The following microorganisms constituted the aerobic microbial culture according to the results obtained using the Biolog Identification System (Biolog Inc., USA): 10 bacterial strains: *Aquaspirillum dispar, Bacillus cereus / thuringiensis, Brevibacterium otitidis, Klebsiella pneumoniae, Proteus penneri/vulgaris, Pseudomonas fluorescence* biotype F, *Pseudomonas marginalis, Pseudomonas mendonica, Pseudomonas sp*. and *Pseudomonas viridilivida* and five yeast strains: *Candida boidini, Candida mogii, Candida memodendra, Pichia haplophia* and *Saccharomyces ludwigii*.

16.4.2. Pilot-Scale Aerobic Treatment

The pilot plant was a 5,000-L tank, with a working volume of around 4,000 L. This type of tank supplied and manufactured locally by Nassar Technology Group (NTG), Lebanon is used normally for domestic package wastewater treatment and was adapted for Zibar biotreatment (Figure 16.4).

Figure 16.4 Aerobic OMW pilot (5,000 L) treatment plant (LAU, Byblos Campus, Lebanon)

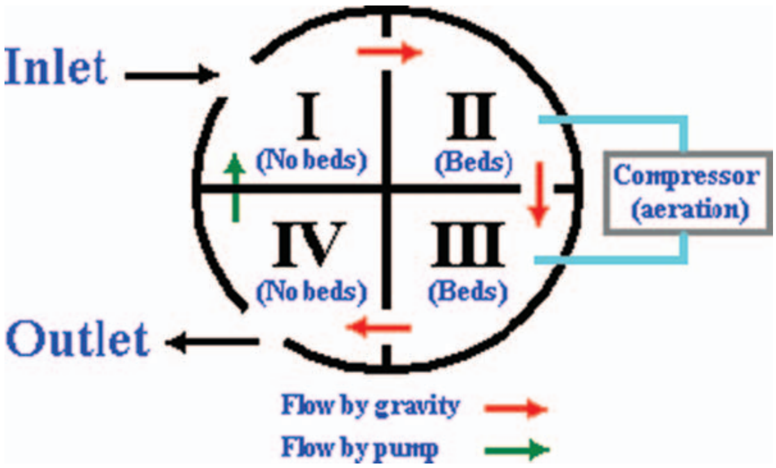

Figure 16.5 OMW cycle inside the pilot treatment plant at LAU, Byblos campus

The pilot tank (Figure 16.4), located at the LAU campus, comprised four interconnected compartments that allow continuous flow, aeration and/or sedimentation of circulating fluid (Figure 16.5).

Compartment II and III contained bed columns and were aerated with an air compressor (Rietschele, Germany; 1 bar, 43.5 m^3/h) housed on the top of the tank.

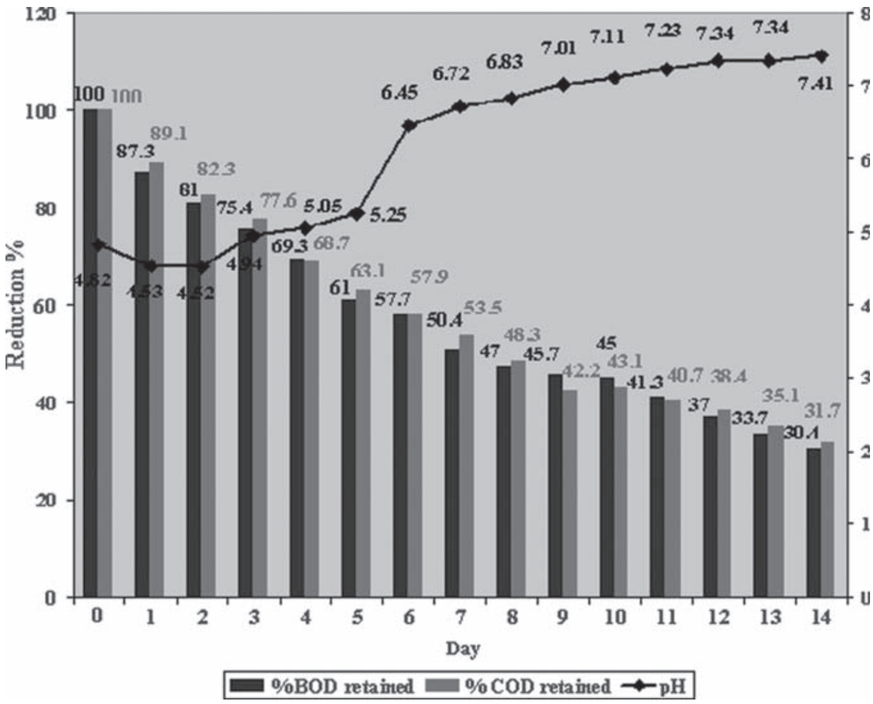

Figure 16.6 BOD, COD and pH changes in the pilot plant scale (5,0001) aerobic treatment of crude Zibar

Compartment IV contained a submersible pump, while compartment I was simply a Zibar holding container. Zibar was filled from the inlet of compartment I, overflowed from compartment I to II, III and then to IV. The cycling was sustained by a submersible pump located in compartment IV, a conical shaped chamber designed for biomass sedimentation (Figure 16.5).

In this series of trials a maximum of 69.6% reduction in BOD and 68.3% in COD values were achieved after 14 days of continuous treatment while an increase of 2.59 in the pH was detected (Figure 16.6).

16.4.3. Industrial-Scale Aerobic Treatment

The industrial-scale Zibar treatment assembly, located at Hasbaya, southern Lebanon, that was installed at the Ziad Abou Ghyda's olive oil mill comprised five large connected tanks (5,000 L each) filled with approximately 25,000 L of freshly pressed crude Zibar (Figure 16.7).

Zibar was pumped from the nearby olive oil mill by a polyethylene pipe (P1) to an intermediate holding tank (T0). Zibar then flowed by gravity from T0 to the

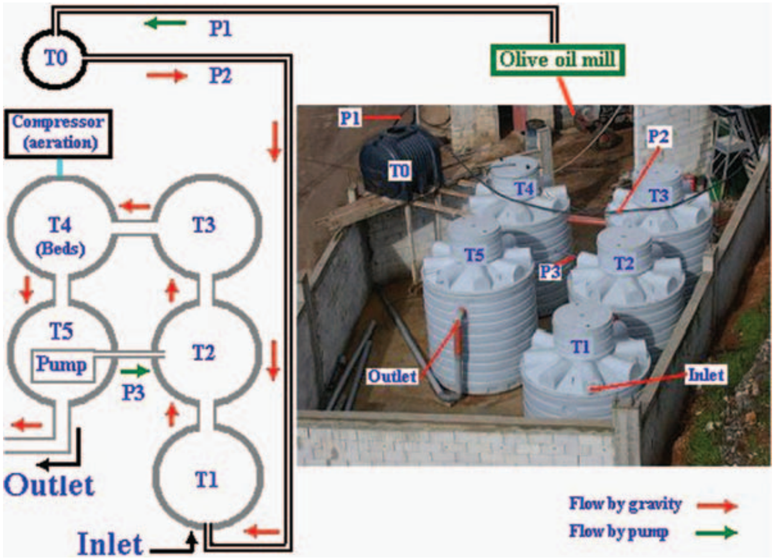

Figure 16.7 Industrial-scale (25,000 L) treatment plant at Hasbaya (P1–3, pipes; T0 holding tank; T1–5 treatment tanks connected in series as described in sketch)

first tank T1 through P2 (Figure 16.6). T1 was used for storage and settlement. Tanks 2, 3, 4 and 5 formed a closed treatment assembly (Figure 16.6). The biological reactor (T4) with compressor, plastic beds and diffusers was inoculated with bacterial and yeast mixed cultures and was sampled at fixed time intervals. The Zibar flowed from T4 into T5, an after-clarification step tank; where a submersible pump removed treated Zibar intermittently back to T2. This closed system was kept running for about four weeks. The treated Zibar was discharged out of T5 subsurface outlet into the neighboring Hasbani River (Figure 16.3). Zibar sludge sedimentation occurred in Tanks T2 and T3 served as sedimentation tanks allowing the partially clarified Zibar to reach the aerated Bioreactor tank (T4). T4 was inoculated with a freshly prepared mixed yeast and bacterial culture transported upon need from the 80 km distant LAU campus. The starter culture (inoculum) reached about 5% (v/v) of the total treated volume. Zibar samples were taken from T4 and occasionally from all other tanks on a weekly basis between October to March and were analyzed at LAU labs for Zibar biodegradation as indicated by changes in BOD_5, COD, as well as other parameters such as DO, pH and temperature.

Unlike the pilot plant scale, trials on the industrial scale stretched over a prolonged period of time reaching 31 days, where the freshly pressed crude Zibar was used directly from the olive mill. The microbial cultures were supplemented on a weekly basis. After 31 days of fermentation, a maximum decrease of 71%

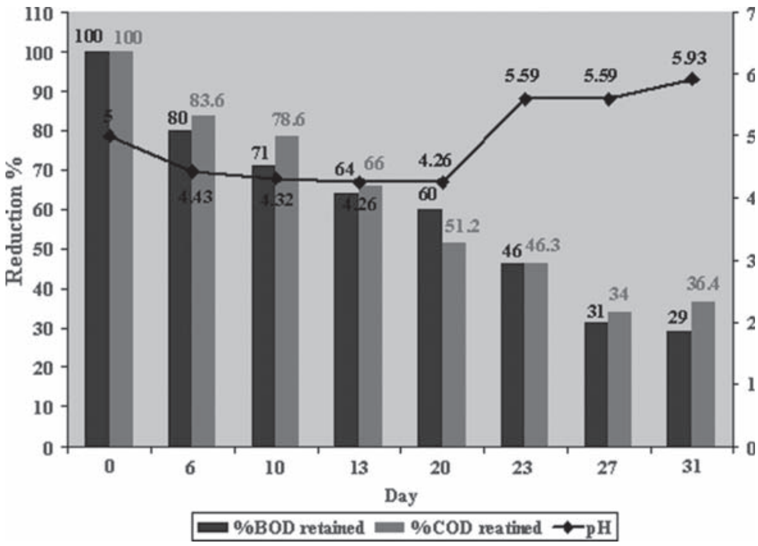

Figure 16.8 Biochemical oxygen demand, chemical oxygen demand and pH changes during industrial-scale (25,000 L) aerobic treatment of olive oil mill wastewater

in the BOD and 63.6% in the COD values and 0.93 increases in the pH were detected as illustrated in Figure 16.8.

16.4.4. Laboratory-Scale Anaerobic OMW Treatment

Erlenmeyer anaerobic flasks (Figure 16.9) were filled with 200 mL of a mixture of basal medium (Demirer and Speece, 1998), Zibar and omasomal juice (cows' third stomach). A control solution was also included to monitor the BOD base value and to determine the possible omasomal juice activity on the total BOD of the mixture.

The flasks were incubated in a water bath at 37 °C and vented to collect the biogas in inverted graduated burettes as shown in Figure 16.9. Changes in BOD, COD and pH values as well as biogas production were measured on a weekly basis for a period of six weeks.

The biogas (mostly methane) production under anaerobic conditions was followed over a period of six weeks under constant culture and incubation conditions. The formed gas mixture (CO, H_2, N_2, CO2, CH_4 and H_2S), which was bubbled first through a 2-L Erlenmeyer flask containing a solution of 20 g/L of KOH to remove CO_2 and other trace gases, was then released inside the inverted burettes (Figure 16.9). This

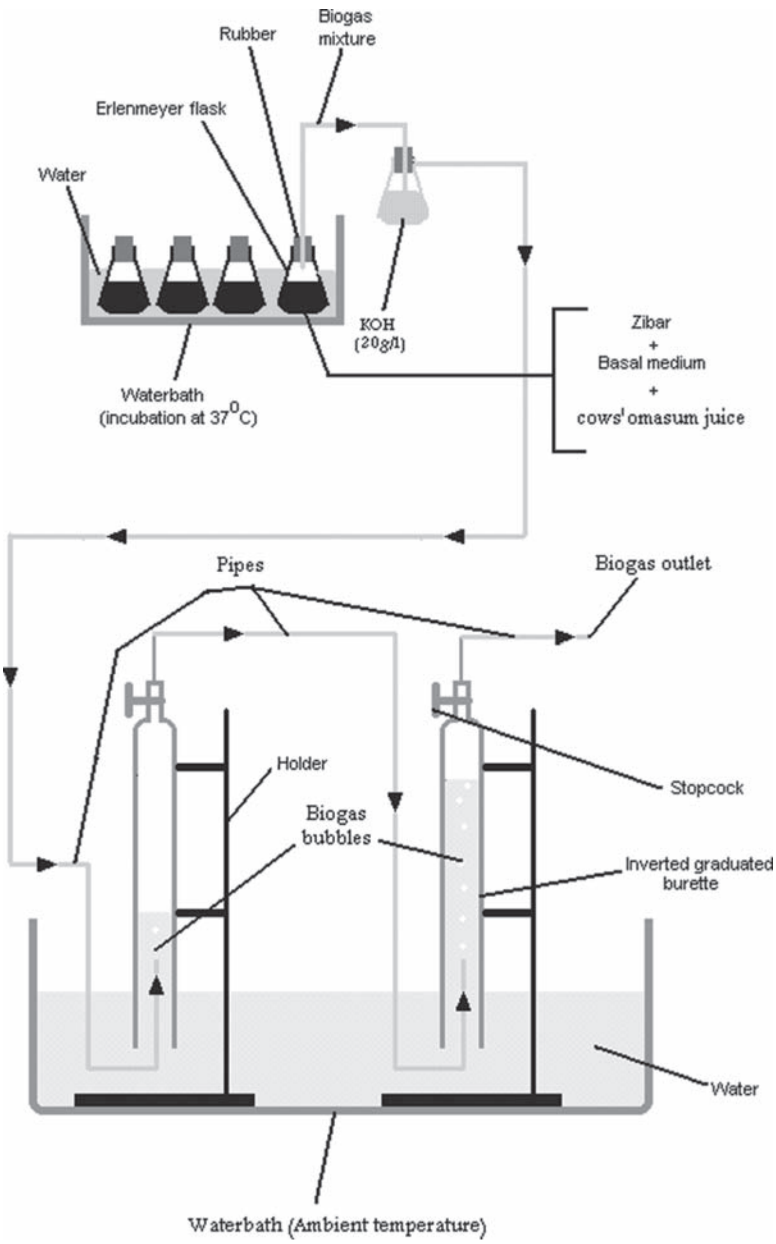

Figure 16.9 Sketch of the anaerobic system used for biogas collection

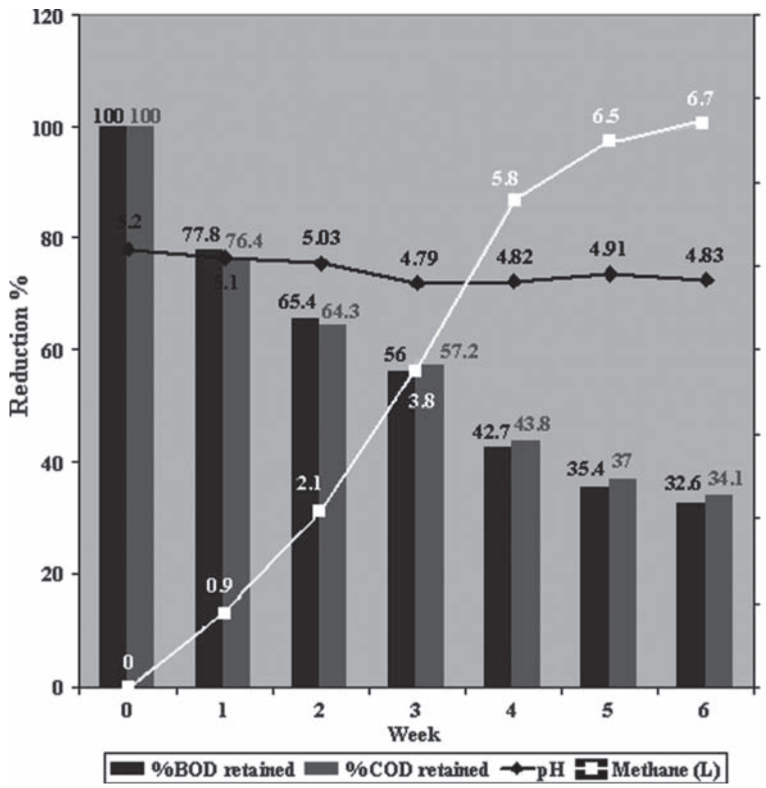

Figure 16.10 Anaerobic Zibar treatment of a crude Zibar using omasomal juice as inoculum showing changes in biochemical oxygen demand, chemical oxygen demand, pH values and Biogas production

released gas mixture was not qualitatively determined and was assumed to be 99% methane according to Demirer and Speece (1998) and Ergüder et al. (1999).

A maximum reduction of 67.4% in BOD and 65.9% in COD values and a decrease of 0.37 in the pH were established after six weeks of incubation of crude Zibar with anaerobic bacterial cultures (omasomal juice). During this period 37.2 L of biogas were produced per liter of Zibar as shown in Figure 16.10.

16.5. Discussion

The uncontrolled disposals of OMW in the environment represent a serious environmental problem. The antimicrobial activity (Capasso et al., 1995; Paixao et al., 1999), the inhibition of seed germination (Bonari et al., 1993; Perez et al., 1986), the phytotoxicity to herbage crops (Capasso et al., 1992; Tomati and Galli, 1992) and the production of stale odor by OMW have been demonstrated. OMW pollution is not limited to lands, but also could reach aquatic environments, surface and underground waters (Moreno et al., 1990; Mendia et al., 1986). Accordingly, high OMWs organic concentration and content of antimicrobial compounds, such as phenols, should be subjected to pretreatment before being discharged in the environment (Ehaliotis et al., 1999). OMW treatment can be achieved satisfactorily by various techniques. As these techniques have a high fixed and operational cost, the majority of olive mills cannot afford to adopt them leading to the spread of the pollution problems. The technical–economic problems such as the requirement of a large installation spaces, professional manpower and high economic costs of the proposed treatment techniques are all limiting factors in the OMW treatment (Ettayebi et al., 2003).

The main objective behind this research study was to develop a low capital cost operational OMW treatment techniques that can reduce the high organic load of OMW, minimize its toxic impacts on the environment and are suitable to be adopted by the Lebanese olive mills in order to minimize the OMW negative environmental impacts. The biological treatments or biotreatments of OMW are seen as treatments of choice, since they are low cost and are capable of converting toxic compounds to useful commercially valuable such as single-cell proteins, agricultural fertilizers, phenolic compounds and biogas (Ettayebi et al., 2003; MINOS Project 2004). This research study was conducted on two treatment strategies to look into effective means for the alleviation of the negative environmental impact associated with the discharge of OMW. These included: an aerobic biotreatment and an anaerobic biotreatment. The aerobic technique was carried at two different fermentation levels the pilot (5,000 L) scaled up to the industrial scale of 25,000 L. These aerobic treatments require relatively simple installation and maintenance; with both scales being low cost and affordable by small- to medium-sized Lebanese olive mills. In contrast the anaerobic biotreatment process, being more complex, was only experimented at the small laboratory scale. In the proposed aerobic treatment, the OMW was assessed in terms of the biodegradation parameters, BOD and COD reduction, while the released biogas under anaerobic conditions was collected and assessed to evaluate the efficiency of the small scale anaerobic fermentation.

To perform the different aerobic fermentations, the following microorganisms (bacterial and yeast cultures were isolated and purified from OMW enrichment cultures. The microorganisms included the following bacterial strains: *Aquaspirillum dispar, Bacillus cereus / thuringiensis, Brevibacterium otitidis, Klebsiella pneumoniae, Proteus penneri/vulgaris, Pseudomonas fluorescence* biotype F, *Pseudomonas marginalis, Pseudomonas mendonica, Pseudomonas sp*. and *Pseudomonas viridilivida* and yeast strains: *Candida boidini, Candida mogii, Candida memodendra, Pichia haplophia* and *Saccharomyces ludwigii*. A survey of the literature revealed very little information about OMW microbial aerobic fermentations and the specific species involved. Trials of aerobic fermentation for pilot and industrial scales, to the best of our knowledge, are not reported in the literature and such large-scale fermentation has been done only with anaerobic biotreatment. In the pilot plant

(5000 L) fermentation trials, a 69% reduction in BOD and a 68.0% reduction in COD values were recorded. These values were relatively similar to the ones achieved by Ettayebi et al. (2003; 69.7% COD reduction) while they were better than Mendoça et al. (2004) with 50% COD reduction. But this process required an incubation period of 14 days for the biodegradation crude Zibar inoculated with aerobic microbial mixed cultures.

The NTG pilot reactor in our study was originally designed for aerobic biodegradation of domestic wastewater. The domestic sewage has a pH ranging between 7.5 and 8 and a BOD around 0.35 g/L, while OMW pH varied between 4.5 and 6.0 and a BOD of 50 g/L (Improlive project A1, 2000; Lopez, 1992; Qureshi et al., 2004; Skerratt and Ammar, 1999). During summer time the temperature inside the pilot plant exceeded the optimal temperature of the aerobic microbial culture mixture to reach about 55 °C. Therefore the aeration time was reduced in order to decrease the temperature to cope with optimal needed range. The melting point of fats and long chain fatty acids is often well above ambient temperatures. At such temperatures, these substances become liquid and are more accessible to microorganisms and their lipolytic enzymes. Diffusion coefficients and the solubility of fatty acids in aqueous media increase significantly with rising temperatures allowing for a better mass transfer (Becker et al., 1999). The industrial-scale fermentation runs showed a maximal decrease of 71% of the BOD and 63.6% in the COD values after 31 days of incubation. These are relatively acceptable values. The two levels of aerobic biotreatment achieved acceptable results in BOD and COD reduction rates. The pilot plant and industrial-scale fermentations could be recommended in Lebanon and other Middle East countries as the OMW treatment of choice since they are low cost and can be locally manufactured and maintained.

The anaerobic biotreatment was only conducted on the small laboratory scale. The objective of using the anaerobic technology, inaddition to reducing the organic load, was to produce a valuable energy source, the biogas. OMW anaerobic treatment, inoculated with omasomal juice as the microbial starter culture, resulted in a reduction of 67.4% BOD and 65.9% COD with the concomitant production of 6.7 L of biogas during the six-week fermentation process. To the best of our knowledge the application of bovine omasomal juice in OMW treatment is reported here for the first time in the literature. According to Kamra (2005) the optimal growth for the rumen microorganisms is 39 °C. The experimental incubation temperature was fixed at 37 °C as set by Ergüder et al. (1999) for anaerobic biotreatment using old OMW sludge as inoculum. This could explain the higher values scored by Ergüder et al. (1999), where COD removal efficiencies ranged between 85.4% and 93.4 compared to 65.9 to 67.3% in our hands, and a 11.42 L of biogas during 44 days of incubation compared to 6.7 L during six weeks in our case. Additionally, good results were achieved by Marques (2001) with 70 to 80% reduction in the COD values inside an upflow anaerobic reactor during six to seven days while a removal efficiency of up to 85% in COD values were reported by Sabbah et al. (2003).

16.6. Conclusions

- This study reports that the aerobic microbial culture that was used in OMW biotreatment was constituted of the following microorganisms: ten bacterial strains: *Aquaspirillum dispar, Bacillus cereus / thuringiensis, Brevibacterium otitidis, Klebsiella pneumoniae, Proteus penneri/vulgaris, Pseudomonas fluorescence* biotype F, *Pseudomonas marginalis, Pseudomonas mendonica, Pseudomonas sp*. and *Pseudomonas viridilivida*; and five yeast strains: *Candida boidini, Candida mogii, Candida memodendra, Pichia haplophia* and *Saccharomyces ludwigii*.
- After 14 days of pilot-scale biotreatment, a 69.1% reduction was recorded in BOD values and a 68.2% reduction in COD values.
- A 71.0% BOD and a 63.9% COD reduction was observed after 31 days of industrial-scale biotreatment.
- Anaerobic OMW experimental laboratory-scale biotreatment using omasomal juice as inoculum culture achieved a reduction of 67.8% BOD and 66.6% COD with 6.7 L of biogas production after six weeks of incubation.

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References

Al-Khudary R, Hashwa F, Mroueh M. (2004). A novel olive oil degrading thermoactinomyces species with a high extremely thermostable activity. Engineer Life Sci 1:78–82.

Baccari M, Bonemazzi E, Majone M, Riccardi M. (1996). Interactions between acidogenesis and methanogenesis in the anaerobic treatment of olive mill effluents. Water Res 30:183–189.

Becker P, Koster D, Popov M, Markosian S, Antranikian G, Markl H. (1999). The biodegradation of olive oil and the treatment of lipid-rich wool scouring wastewater under aerobic thermophilic conditions. Food Sci Technol 33:653–660.

Bonari E, Marcchia M, Ceccarini L. (1993). The wastewaters from olive oil extraction: their influence on the germinative characteristics of some cultivated and weed species. Agric Med 123:273–280.

Capasso R, Cristinzio G, Evidente A, Scognamiglio F. (1992). Isolation, spectroscopy and selective phytotoxic effects of polyphenols from vegetable wastewaters. Phytochemistry 31:4125–4128.

Capasso R, Evidente A, Schivo L, Orru G, Marcialis M, Cristinzio G. (1995). Antibacterial polyphenols from oilve mill wastewaters. J Appl Bacteriol 79:393–398.

Demirer GN and Speece RE. (1998). Anaerobic biotransformation of four 3-carbon compounds (acrolein, acrylic acid, allyl alcohol and n-propanol) in UASB reactors. Water Res 32:747–759.

Ehaliotis C, Papadopoulou K, Kotsou M, Mari I, Balis C. (1999). Adaptation and population dynamics of *Azotobacter vinelandii* during aerobic biological treatment of olive-mill wastewater. Microbiol Ecol 30:301–311.

Ergüder TH, Güven E, Demirer GN. (1999). Anaerobic treatment of olive mill wastes in batch reactors. Report, ATD Environmental Technologies Ltd. and the State Planning Organization of the Republic of Turkey, Ankara, Turkey.

Ettayebi K, Errachidi F, Jamai L, Tahri-Jouti M, Khalid Sendide K, Ettayebi M. (2003). Biodegradation of polyphenols with immobilized *Candida tropicalis* under metabolic induction. Microbiol Lett 223:215–219.

Feria AL. (2000). The generated situation by the OMW in Andalusia. Workshop Improlive Anexo A1/ Annex A1, Spain.

Food and Agriculture Organization Statistics database. (2003). Available at: http://apps.fao.org/default.htm.

Hamdi M, Khadir A, Garcia JL. (1991). The use of aspergillus niger for the bioconversion of olive mill wastewaters. Appl Environ Microbiol 34:828–831.

Hashwa F. (2003). Biotreatment of olive mill liquid waste (Zibar) in Hasbaya. Report, Mercy Corps, Beirut, Lebanon.

Hoyos S, Neito L, Rubio F, Cormenzana A. (2002). Kinetics of aerobic treatment of olive oil-mill wastewater (OMW) with Aspergillus terrus. Process Biochem 37:1169–1176.

Improlive project A1. (2000). European Commission, Universidad Complutense De Madrid. Workshop Improlive AnexoA1/ Annex A1 2000. Final Report Annex A1, Spain.

Improlive project A2. (2000). Improvements of treatment and validation of the liquid solid waste from the two phase oil extraction. EC report (Fair CT96- 1420) Final Report- Annex A2, Spain.

Kamra DN. (2005). Rumen microbial ecosystem. Curr Sci 89:1–10.

Lopez R. (1992). Land treatment of liquid wastes from the olive oil industry (Alpechin). Fresenius Environ Bull 1:129–134.

Marques IP. (2001). Anaerobic digestion treatment of olive mill wastewater for effluent re-use in irrigation. Desalination 117:233–239.

Mendia L, Carbone P, Antonio G, Mendia M. (1986). Treatment of olive oil wastewaters. Water Sci Technol 18:125–136.

Mendoça E, Martins A, Anselmo AM. (2004). Biodegradation of natural phenolic compounds as single and mixed substrates by *Fusarium flocciferum*. Elect J Biotechnol 7:1.

MINOS Project. (2004). Process development for an integrated olive oil mill waste management recovering natural antioxidants and producing organic fertilizer. TERRA NOVA Ltd., Athens.

Mhanna YE. (2006). Aerobic and anaerobic biotreatment of olive oil mill wastewater. MSc thesis, Lebanese American University, Byblos, Lebanon.

Moreno E, Quevedo-Sarmiento J, Ramos-Cormenzana A. (1990). Antibacterial activity of wastewaters from olive oil mills: hazardous waste containment and treatment. Gulf Publications, Houston, TX, USA.

Papadimitriou EK, Alexiou IE, Chew TY, Balis C Stentiford EI. (1997). Guidelines for biological treatment of vegetable oil extraction wastewaters. Sixth IAWQ Asian WaterQual Conference, Seoul, Korea.

Paredes MJ, Monteoliva-Sanchez M, Moreno E, Perez J, Ramos-Cormenzanz A, Martinez J. (1986). Effect of wastewaters from olive oil extraction plants on the bacterial population of soil. Chemosphere 15:659–664.

Paixao SM, Mendoça E, Picado A, Anselmo AM. (1999). Acute toxicity evaluation of olive mill wastewaters: A comparative study of three aquatic organisms. Environ Toxicol 14:263–269.

Perez JD, Esteban E, Gomez M, Gallardolara F. (1986). Effects of wastewater from olive processing on seed germination and early plant growth of different vegetable species. J Environ Sci Health 21:349–357.

Pérez J, De La Rubia T, Ben Hamman O, Martínez J. (1998). *Phanerochaete flavido-alba* laccase induction and modification of manganese peroxidase isoenzyme pattern in decolorized olive oil mill wastewaters. Appl Environ Microbiol 64:2726–2729.

Pérez J, De La Rubia T, Moreno J, Martínez J. (1992). Phenolic content and antibacterial activity of olive oil wastewaters. Environ Toxicol Chem 11:489–495.

Qureshi S, Richards BK, Steenhuis TS, McBride MB, Baveye P, Dousset S. (2004). Microbial acidification and pH effects on trace element release from sewage sludge. Environ Pollut 132:61–71.

Ragazzi E and Veronese G. (1989). The effect of oxidative coloration on the methanogenic toxicity and anaerobic biodegradability of phenols. Bio Water 32:210–225.

Sabbah I, Marsook T, Basheer S. (2003). The effect of pretreatment on anaerobic activity of olive mill wastewater using batch and continuous systems. Process Biochem 39:1947–1951.

Saez L, Perez J, Martinez J. (1992). Low molecular weight phenolic attenuation during simulated treatment of wastewaters from olive oil mills in evaporation ponds. Water Res 26:1261–1266.

Skerratt G and Ammar E. (1999). The application of reedbed treatment technology to the treatment of effluents from olive oil mills. Final report, ENIS/LARSEN, Staffordshire University, UK.

Tomati U and Galli E. (1992). The fertilizing value of wastewaters from the olive processing industry: Humus, its structure and role in agriculture and environment. Elsevier Science, Amsterdam.

Tuncel G and Nergiz C. (1993). Antimicrobial effect of some olive phenols in a laboratory medium. Lett Appl Microbiol 17:300–302.

Ubay G and Ozturk L. (1997). Anaerobic treatment of olive mill effluents. Water Sci Technol 35:287–294.

Chapter 17
Cost-Efficiency in Water Management Through Demand Side Management and Integrated Planning

Dr. Ing. Ralf Otterpohl

Abstract In the context of regional planning for efficient management of water and wastewater, it is crucial to assess the specific local situation. This consists of measures for water demand side management, comparison of different scenarios of water and wastewater systems based on dynamic cost–benefit analysis for the decision of central, communal or decentral structures and the decision on technological approaches. There are extremely resource efficient solutions available, however, they are not very well known yet. In regions with a smaller population density, well designed onsite systems for wastewater management in combination with safe local reuse for irrigation can be very competitive. It has to be stated that decentral solutions require professional operation and management, as experience around the world shows that they fail otherwise.

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17.1. Demand Side Management: Less Water Consumption – Less Wastewater – Less Vulnerability

17.1.1. Introduction

Efficient management of water resources in urban as well as in rural areas is very much dependent on demand side management. This means that through public awareness campaigns, capacity building and activities of companies with good products water consumption is brought to a lower and stable level. Naturally, this has to be coordinated with gravity transport sewerage systems to assure sufficient flow for solids and deposits. Onsite systems have an advantage as they can be brought down to very low water consumption in times of scarcity without failures. Larger mixed gravity sewers will fail below a critical threshold of flow. This chapter describes are a number of measures that can lead to massive water savings often with little investment.

Of course, the water distribution and sewerage system quality is crucial for water efficiency and road safety. The high costs of rehabilitation of supply networks do often lead to unacceptable losses with the consequence of interrupted supply. This can lead to infiltration of wastewater into the water pipes and is to be considered as an emergency situation with potentially severe health risks. The maintenance costs have to be calculated in any case and the work has to be done regularly. In many countries with limited water resources around the world, there are high losses of water due to neglecting the simplest measures like adjusting the water pressure of taps to reasonable amounts or to have toilet flush equipment repaired when it is leaking constantly.

17.1.2. Efficient Showerheads

Showers belong usually to the main water consumers. Besides water, a lot of energy is often needed to heat the shower water. It is amazing to see that even in regions with limited water resources there are usually showerheads with high flows in use. In addition even in regions with very high solar radiation and colder winters there is often little solar water heating in place. Proper demand side management both for water and energy conservation can improve efficiency often even at lower costs. True political leadership is needed on all levels to keep out the environmental risks low.

One example for an efficient showerhead is calculated below. The comfort of the air-injection technology is the same as with wasteful high-flow equipment and water consumption can be reduced by one- to two-thirds easily. The calculation example shows the amazing potential (see Figure 17.1):

Consumption per day conventional showerhead: 18 L per minute * 5 minutes *18 = 90 L total per shower

State-of-the-art shower heads (e.g., Bubble Rain with air injection): 6 L * 5 minutes = 30 L

Figure 17.1 Water and energy savings through highly efficient shower heads (at the same time less wastewater generated; e.g., Bubble Rain)

$$\text{Per year } 360 \times 90\,\text{L} = 32{,}400\,\text{L}$$
$$\text{compared to } 360 \times 30\,\text{L} = 10{,}800\,\text{L}$$
$$\text{yearly water savings per person} = 21{,}600\,\text{L}$$

This makes around 20 m^3 per capita and day savings in freshwater consumption, at the same time 20 m^3 less wastewater to be treated and often additionally energy savings. It can usually not be expected that people discover this by themselves on a large scale. Consequently political leadership is needed and expressed in public awareness campaigns. In regions with the risk of severe droughts acceptance of high water consumption means putting the population to risks. Wasting water and depleting reservoirs means that consequences of droughts hit faster.

17.1.3. Water-Efficient Toilets and Urine Diversion

Another major water consumer besides showers is the flush toilet. Unfortunately, this type of sanitation is not well suited for regions with limited water availability and risk of drought. The minimum flush of the best designs that are still working well is 4.5 L per flush, hover most toilets work with 7 or even 9 L. Dual flush is helpful to reduce the freshwater consumption but even the small volume is 4 L. If dual flush is implemented it is advisable to have two different buttons, a small and a big one, otherwise many people will not operate them as intended. Water consumption for anal cleansing can be optimized by well-designed systems with appropriate flows.

High-quality waterless urinals are available and work very well when cleaning is done on a regular bases; the technical breakthrough was a membrane-smell trap that is made of gum and like a pipe on one side and flat on the other. The flat part is to the downside and opens when a liquid is flowing through while closing after the flow stops. All other systems are technically outdated. Such waterless urinals can save high amounts of water in public toilets and in addition avoid pipe clogging. It seems that the mixture of water and urine tends to produce a lot more scaling, especially when the water has a high hardness. Urine pipes should have a sufficient diameter and the flow should be without strong turbulences.

There are increasing cases of failing sewers and even toilets due to severe water shortages. Water shortages are mostly a consequence of mismanagement and installation of flush toilets in dry regions can be part of mismanagement. Unfortunately the development of modern dry toilets with urine diversion is relatively new and still lacks really good and cheap components. However, there are excellent examples for well-made projects around the world. The number of installations grows very fast and in an exponential way. There is more information in other chapters of this book, here it is only highlighted from the perspective a possibility for very high water efficiency and lessening the hygienic risks associated to failing or never finished flush systems. At the same time the rest of the wastewater is much less in volume and far easier transported, treated and reused.

Consumption per person per day of flush toilet above 30 L
Dry toilet: very little demand
Per year $360 \times 30l = 10{,}800$ L
yearly water savings per person = 10,800 L

There are also very water-efficient flush toilets, but their usage must be appropriate for the wastewater system. Sewerage systems require minimum flows in order to avoid clogging; on site can work well with low flush if installations are adequate. The flush volume should be adapted to the specific location. If modern dry toilets with urine diversion are used there is little concern about clogging sewers anymore. In this case the most difficult fraction of faecal matter will not enter the sewers at all. In this case, it can be possible to construct a cost efficient solids free sewerage system or to combine the grey water treatment with irrigation of the gardens. Direct irrigation with used water with little pollution is an option, too. The consequences of the usage of toxic household chemicals on the wastewater should be considered, this is especially true for disinfection and small business on backyard scale.

17.1.4. Rainwater Harvesting

Rainwater management is an important part of wastewater management. For rural and the less densely populated peri-urban areas it does obviously make a lot of sense to implement infiltration units instead of expensive rainwater sewers (also called "storm" sewers). This can be combined with rainwater usage. Internationally there are many activities that can be found under the keyword of rainwater harvesting. Directing rainwater into the aquifers while it rains had a major impact on the economic development in many rural areas with seasonal rains. Due to climate change seasonal rains may even fail for several years, what usually leads to loss of sometimes more than 90% of the population. The people leaving for the cities increase problems there and often do not return after the rains finally come. Rural development is hardly possible in such circumstances. Those villages that have installed the whole range of rainwater harvesting often driven by one single enthusiastic person for example in Gujarat, India, could still grow vegetables in the third year of draught. People could stay and found themselves more prosperous over the years even with less rainfall (CSE). Aquifer recharge through small check dams and converting wells to aquifer recharging units in the rainy season is an excellent way of making rural areas more stable. If this is combined with proper irrigation practices, proper choice of water-efficient plants and water-efficient household installations combined with wastewater reuse there are many regions with limited water resources that can dramatically improve their situations. As so often, the rare capacity of real leadership combined with good knowledge is required.

From experience in many countries, it must be stated that it makes a lot of sense to use the vast existing knowledge. Cisterns seem simple but there are some simple but crucial points for the design. The dirty first flush should better not be collected; screens and filters keep coarse pollution out. The tank outlet should not be at the very bottom to avoid the intake of sediments. If the underground conditions and legal situation allow it the infiltration into the aquifers can be an excellent way of a combination of flood prevention and storage also for reuse purposes. In Germany there is an increasing number regional water legislation that requires infiltration of rainwater runoff unless it is geographically impossible.

17.2. Cost-Efficiency Through Regional Planning: Central, Communal, Decentral?

17.2.1. How to Determine the Costs of Central or Decentral Solutions

The evaluation of options for the water and wastewater management options in rural and peri-urban areas is a complex task (see Figure 17.2). It does require good knowledge on the options and in addition there is a lot of work to make an economic

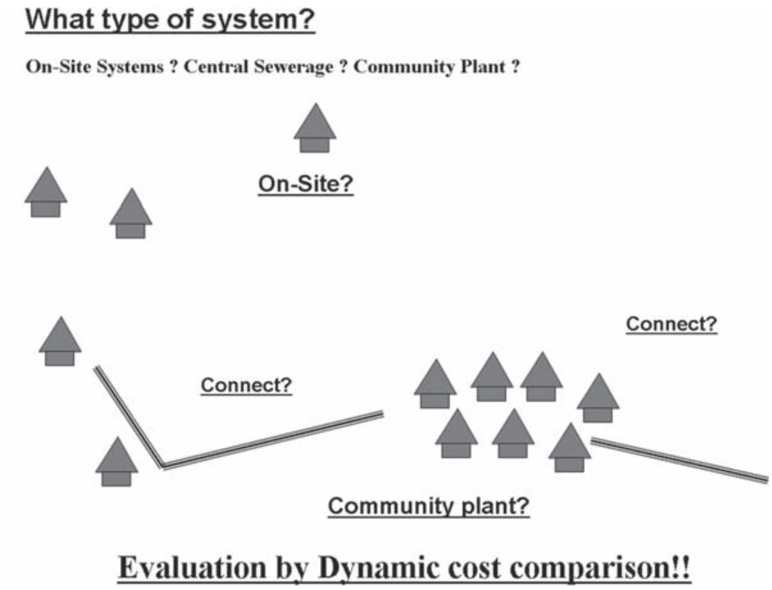

Figure 17.2 Central and decentral solutions in regional planning

comparison. Unfortunately, this does not work with simply adding up the investment costs of the different options. There are usually big differences in the trade-off between investment and operation costs. The option with the lowest investment costs can be very expensive in the long run because of excessive operation and maintenance costs.

There are straightforward methods, where the economic consequences of different options can be demonstrated. The choice of parameters must be done according to the real situation and shown openly in the report, numbers must be reproducible. Naturally, with big and also smaller investment there are people who have their favorite solutions. This may be for reasons of personal preference and assumed professional knowledge or for the simple and understandable reason that the uncle owns a company that will make huge profits from the most expensive solution. This does happen very often around the world unfortunately only with few exceptions like the Scandinavian counties. In the case of wastewater systems taking bribes is like stealing money from the whole community. If a project of wastewater management becomes very costly it is usually resulting in high wastewater prices for all people served by the system. The person accepting bribes is like a burglar who steals money from the pockets of all the people paying their bills, every day, for many years. It is unnecessary to say that this is criminal practice and this includes cases like helping an uncle or other relative to get a contract for a solution that is far too expensive.

The way to go for a cost efficient solution is through dynamic cost benefit analysis. For all solutions that could be reasonable the investment and operation costs are evaluated according to commonly agreed on criteria (interest rates, life spans of units and machinery, wages, etc.). Then with this data the net present value of each solution is calculated and the economic comparison can be done. Of course, there are often still differences in the ecological performance. In one case with the same net present value more money can go into local employment, helping the community to generate more income. In another one there can be mostly expensive import equipment where the money is spent to far away companies. These are additional decisions that are part of the overall decision making. Engineers should bring up a choice of feasible solutions and give the evaluation criteria. With this decision makers can choose in an open and transparent way. This type of decision making was developed for example in Germany after very many solutions where built, where the costs where much higher that necessary. Now there is an official procedure how to calculate and this is very helpful to get to cost efficient solutions where the fees for the citizens are in a reasonable range.

In combination with the question central/communal/decentral treatment the choice of appropriate sewer types is crucial in the process of evaluation of scenarios (see Figure 17.3). The initial material costs and life spans vary widely and different situations require different solutions. Besides gravity sewers there are the options of pressure or vacuum systems. These can be far cheaper in investment for longer transport distances, however, operation costs and life spans of pumps and equipment

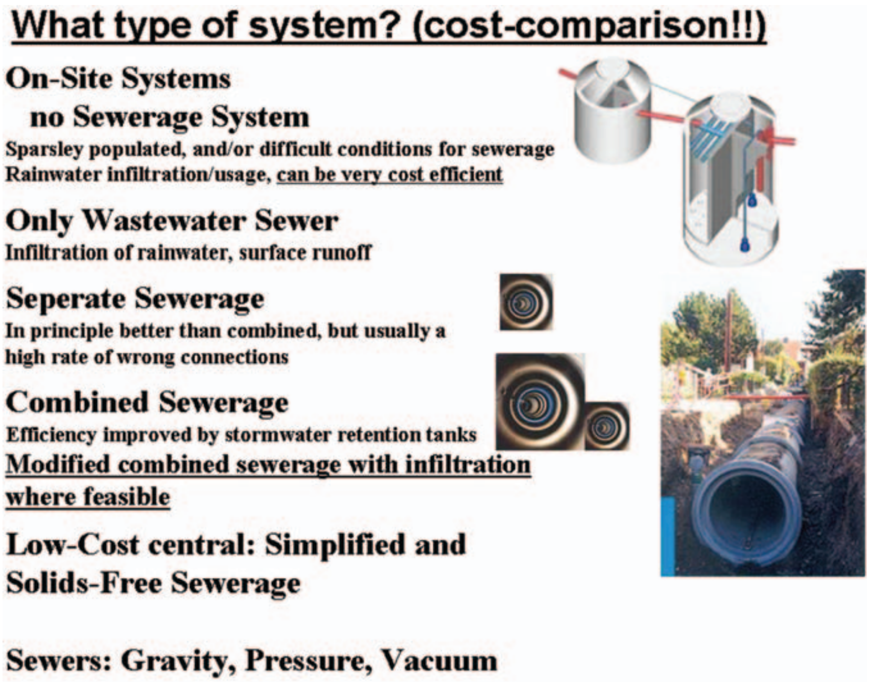

Figure 17.3 Options for transport of wastewater for different situations

Table 17.1 Regional planning for efficient wastewater management

1. Reduction of wastewater flows through demand side management, efficient usage of water (public awareness for water-efficient house installations, water saving toilets or for Ecosan units).
2. Keep rainwater runoff out of wastewater unless it is very polluted (first flush, roads with heavy traffic, industrial sites), infiltration through top soil with aquifer recharge and reuse where possible (rainwater harvesting).
3. Consider separate treatment of industrial effluents where appropriate.
4. Comparing central, communal or decentral solutions with dynamic cost comparison: - types of sewers: gravity, simplified, solids free, pressure, vacuum - pretreatment options: sedimentation, Imhoff tank, UASB (if temperatures above 20 °C all year), precomposting tanks - aerobic treatment options: activated sludge, trickling filters, rotating disc, membrane bioreactor (good for wastewater reuse), constructed wetlands, ponds (mosquito control) - for better effluent standards: post treatment with biofilters, membrane-filtration, constructed wetlands
5. Check local or regional reuse options: agricultural reuse (adjust for nutrients), industry, aquifer recharge where appropriate. Local reuse is simpler when Ecosan is installed and there is only grey water.
6. Manage construction and operation, capacity building, verify quality at any step.
7. Assure finance trough water fees and reinvestment according to life spans and actual state, verify (include this into the running management tools).

must be considered. This can be done with the dynamic cost-comparison method in the comparison of the scenarios.

For rural and many peri-urban areas it is mostly not necessary to transport rainwater runoff in sewers. Infiltration is often feasible and can help recharging local sewers. Please see Section 17.1.4 for more information. Table 17.1 shows the key steps toward efficient wastewater management in a regional context.

17.2.2. Considerations for Decentral Wastewater Management

Sewerage systems require a minimum flow; they fail with severe water shortages. Even though economic considerations may show an advantage towards central sewerage it can be prohibitive when severe water shortages can be expected for the future. In addition people will probably start to reuse their wastewater to irrigate their gardens. It may be wise to consider such situations especially now where the dramatic effects of climate change through global warming is reality. However, for all decentral wastewater systems long-term maintenance has to be assured on all levels. Technical options are discussed elsewhere in this book, and these should be combined in a proper way. The downside of decentral systems can be that very vulnerable water resources of small creeks are receiving wastewater that can be a heavy burden even with some treatment.

Around the world it can be observed that conventional designs are applied almost without looking for alternatives. Especially in areas with scarce resources and limited availability of income it can be very beneficial to look at water and waste systems from the side of income generation in the local context. It is possible to supply water for reuse, nutrients with the water or separate, biogas can be produced in larger units and aquifers can be recharged. The local operation of the systems can create jobs where the money that is paid for the service will be mostly spent in the local context. Grey water can be treated with productive gardens, irrigating (e.g., citrus trees or banana plants) while treating the wastewater. Regional planning should explore the opportunities and it will be beneficial to make use of the vast number of great examples from around the world.

17.2.3. One step Further: Resource-Oriented Sanitation/Ecosan

The International Water Association (IWA) has a specialist group that is called Resources-Oriented Sanitation or Ecosan (IWA). These types of systems can have a very high efficiency at reasonable costs. There is a wide range of technical solutions form high-tech to very simple low-tech. Examples are given in other chapters in this book. There is a lot of information on IWA, GTZ and Ecosanres, especially on the low-tech side. For medium- and high-tech systems there is some information on OW. Many people are afraid of implementing such systems, because they do not know them. This is a very normal reaction and it is good to have a critical look at new things. On the other hand it is good to get some experience because there are many situations where Ecosan-systems have strong advantages. Cultural aspects have to be taken into consideration. It will be good to have a demonstration unit in the region in order to get practical experience and to be able to create good local solutions. Long term maintenance and operation must always be assured, experience shows that professional management can lead to excellent performance. Information of potential users and demonstration of the choices are crucial before Ecosan projects can be started. The number of installations of Ecosan systems grows very strongly in many parts of the world—a lot of their regions are similar to those in the Middle East. It has been discovered as an interesting business area by several local and regional entrepreneurs. In the context of regional planning it has to be stated that the installation of Ecosan systems should be coordinated by the local authorities.

References

CSE Web site. Available at: www.rainwaterharvesting.org
International Water Association Web site. Available at: www.ecosan.org
OW Web site. Available at: www.otterwasser.de
GTZ Web site. Available at: www.gtz.de/ecosan
Ecosanres Web site. Available at: www.ecosanres.org

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Chapter 18
The LooLoop-Process: The First "Waterless" Flushing Toilet

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Abstract The environmentally open disposal of fecal contaminated wastewater flows from flushing toilets has caused and is causing a broad variety of qualitative threats and problems. Additionally, the extraction of freshwater for domestic uses from long-term renewable water resources, like ground water, and the following disposal into short-term renewable water resources, like rivers, which flow finally into the oceans within days up to a few months, is intensifying the decrease of continental fresh water resources. With this de-central concept innovation, most of this threats and problems can be solved, and the water demand for flushing of toilets can be reduced down to zero at the same time. Thereby, a closer look at the different types of buildings is essential to developing specific technologies according to the characteristics and amounts of their partial-stream separated wastewater flows. A characterization of different domestic buildings types is presented in this paper and a holistic approach is introduced, of how to reuse fecal contaminated flows, as well as how to reclaim valuables, like biogas, mineral fertilizer and soil-conditioner by simultaneously eliminating all pathogens and hazardous substances, like pharmaceuticals, hormones and multi-resistance plasmids.

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18.1. Objectives

The common state-of-the-art technology can be characterized by the term “centralized flow-through concept” comprising the following positions: *freshwater withdrawal → drinking water processing → usage → wastewater → sewerage (→ treatment plant) → disposal (most common in surface waters) → sea water*, whereby only a small fraction of the waste water is treated worldwide in sewage plants. Still most common is the discharge of raw sewage into the water bodies.

This disposal of fecal polluted waste water (black water) into our aquatic environments is still an unsolved ***quality problem*** causing Millions of deaths and infections per year. Whether treated in sewage plants or not, pathogens, hormones, resistance plasmids and pharmaceuticals are thereby released into our water resources, causing serious threats to our environments and public health. Thus, epidemics like the dreadful cholera epidemic of Hamburg, Germany, in 1892 (Evans, 1991) can be the consequences. In many countries and regions of the world (where the drinking water is extracted from surface water bodies), this shortcut between toilets and drinking water faucets still exists.

Where groundwater is the source of drinking water extraction, the centralized flow-through concept causes also a ***quantity problem***. Here, long-term renewable water resources (renewal rates of up to several Million years) are discharged via short-term renewable water resources (surface waters, e.g., rivers: renewal rates of days only, up to several months) into the seas. This shortcut is additionally intensifying the worldwide water crisis.

Additional treatment steps for end-of-pipe-systems, like ultra-filtration and/or ozone treatment, are applied to avoid the disadvantages. Increasingly common in the world’s arid and wealthy regions (e.g., Gulf Region) is the reuse of the treated and ultra-filtrated waste water for irrigation in agriculture, or for groundwater recharge. The problem arises here, that hormones, antibiotics-resistance plasmids and pharmaceuticals can pass the ultra-filtration membranes, arrive in the food chain and finally into humans. To degrade these contaminants, the entire ultra-filtrated sewage stream needs to be additionally treated with ozone, which would be very costly considering the quantity of these flows.

Another major drawback of the centralized flow-through concept is the high investment into sewerage systems, which are typically between 70 and 90% of the overall investment into wastewater systems. Moreover, the nutrients incorporated by humans and excreted with urine and feces are highly diluted and can hardly be recovered subsequent.

Within the last years, several concept innovations have been introduced. Within the Gesellschaft für technische Zusammenarbeit (GTZ), a specialized sector project called Ecological Sanitation (EcoSan), for this flow-oriented concept was established (GTZ, 2002). Many proposals and common technologies for a source control wastewater management including different flow treatment already exist (Otterpohl et al., 1999; Winblad, 1998). Some of these concepts are more suitable for rural areas; whereas some are more applicable for cities. In Otterpohl et al. (1999), 10 basic scenarios are classifying the variety of combinations of modules in dependence of the different geographic and socio-economic conditions around the world. Paris and Wilderer (2001) elaborated an extensive overview of realized concepts based on source control. Some of these concepts are based on urine separation (Braun, 2001), while this concept is based on normal flushing toilets, which don’t consume any water anymore (Braun, 1998). Well known is the so called vacuum biogas concept, which has been realized in Lübeck, Germany (Otterpohl et al., 1999; OtterWasser, 2002). In Freiburg-Vauban, Germany, a similar project has been realized with 40 inhabitants (Lange, Otterpohl, 2000). In Berlin, Germany, a maintenance building has been equipped with a sewage plant using urine separating toilets (BWB, 2002). In Norway, black water is treated anaerobically under thermophilic conditions (Skjelhaugen, 1998).

18.2. Building Types and Clusters

The term “building” comprises here all means, where humans can stay, including mobile means like trains, ships or airplanes. There is only little information available about the wastewater characteristics of different building types. Known is the highly concentrated composition and low pH of urine being excreted in the morning (“morning urine”). Known from interviews is also, that almost all women, but also many men are refusing to sit down on the lid of public toilets for esthetical reasons. The consequence is that most people are aspiring to defecate at home, where these esthetical obstacles don’t exist. Also obvious is that in several building types no washing machines, baths, etc. exist, which influences the amount and composition of grey water considerably. With logical implications an estimation regarding compositions and amounts of wastewater partial streams in different building types is possible. Table 18.1 gives an overview over different buildings and their assumed characteristics. It is obvious, that in the first eight types a public toilet is used, whereas private toilet usage is dominant in the last seven. These two groups also differ with regards to grey water. The first group is characterized by only little up to nearly no production of grey water, whereby the second group can be described with a high production rate of grey water. Interestingly, the second group are that kind of buildings, where people stay overnight, and in the first the kind of stay is restricted to the daytime only.

18.3. Materials and Methods

New and more efficient EcoSan technologies compared with the centralized concepts are based on source separation of toilet wastewater. In a close cooperation with the Hamburg University of Technology (TUHH), Intaqua AG has investigated the process for the reuse of urine as toilet flushing water—the loop processing of toilet waste water generating concentrated and thoroughly treated liquid

Table 18.1 Building types and characteristics

	Black water				Grey water				
	Toilet type		Urine type						
Building type	Public	Private	Morning	Daily	Bath	Kitchen	Laundry	Pool	Irrigation demand
Office	+++	—	–	++	–	+	–	—	–
Restaurant	+++	—	—	+++	–	+++	–	—	–
Roadhouse	+++	—	–	++	+	++	−	–	+
Shopping mall	+++	—	—	+++	–	+	–	—	−
Sport arenas	+++	—	—	+++	++	+	–	—	++
Ferry	++	–	−	+	+	+	–	–	—
Train	+++	—	–	++	–	−	—	—	—
Air plane	+++	—	+/−	+	–	–	—	—	—
Hotel	–	++	++	++	+++	++	+/−	++	+++
Hospital	−	+	++	++	+++	++	+/−	–	++
City center	−	+	+/−	+++	+	+	+	–	−
Urban settlement	–	++	++	++	+++	+++	+++	–	+/−
Rural settlement	–	++	++	++	+++	+++	+++	−	+++
Village	–	++	++	+++	++	++	++	−	++
Cruiser	−	+	++	–	++	++	++	++	–

Legend: +++: always; ++: almost always; +: mostly; +/−: half/half; −: rarely; –: almost never; —: never.

and solid fertilizers plus soil conditioners. This process has been researched at the TUHH, where a semi-technical (20 population equivalent [PE]) plant has been designed, built and operated for almost 2.5 years. The treatment steps of the process comprise of:

1. Separation of solids, which are further worm-composted (vermicomposting) in smaller units or digested (larger units).
2. Ureolysis/equalization tank.
3. Biological oxidation comprising nitrification of the liquids.
4. Ultra-filtration.
5. UVC radiation, respective ozone treatment.

The following compounds have been used:

1. Liquid/solids separation with a riddle-screen, SWECO Separators, model LS18S33, SWECO Inc., USA.
2. Ureolysis in a not aerated tank.
3. VA-steel-reactor with aeration membranes (in-house construction).
4. VUM (Vacuum Upstream Membrane) Ultra-filtration membrane of Hans Huber AG, Germany.
5. UVC radiation: sterilAqua UVC Radiator, model AQD2136-3 by sterilAir AG, Switzerland.
6. Ozone treatment: Sander Labor-Ozonisator, 20/80 g/m3, Erwin Sander Elektroapparatebau GmbH, Germany.
7. Sensor measurement: WTW IQ Sensor Net, System 2020.

Analytics have been proceeded with:

1. COD: Lange Küvettentest LCK314 TC, TOC, TIC autoanalyzer.
2. BOD: Respirometer Selutec BSBdigi.
3. $NH_4/NO_2/NO_3$: Dr. Lange Küvettentests, RQflex. MERCK.
4. Temperature, pH, O_2, conductivity, redox, TS: WTW IQ Sensor Net.

First, the process of ureolysis has been investigated. Three reactors and one reference reactor à 7.7 L have been constructed and operated:

a. reference reactor
b. aerated pumped fixed bed reactor
c. pumped fixed bed reactor
d. anaerobic stirred reactor

Besides the reference reactor, all reactors have been inoculated with sewage sludge. All reactors have been fed then with an increasing amount of artificial black water, composed of 12 g feces diluted in 1 L of urine. During increase, the activity of ureolysis was measured. Furthermore experiments regarding biological oxidation of urine, storage of oxidized urine and decolorization of urine have been carried out. Figure 18.1 illustrates the three research reactors:

After researching of this and some further topics the technical plant was installed. A normal flush toilet and a waterless urinal have been installed on the upper level

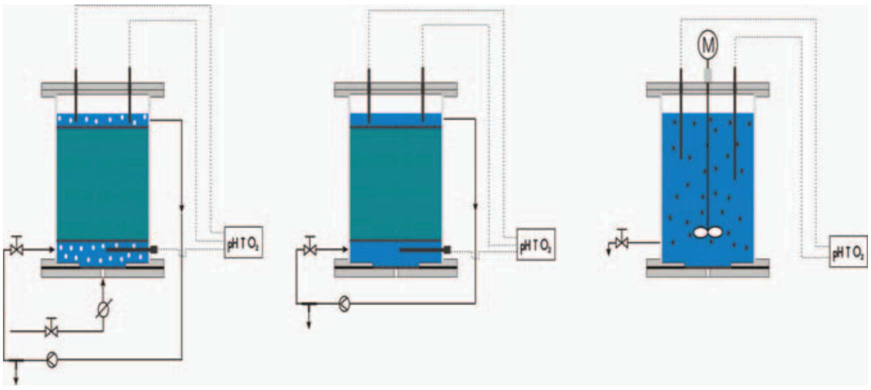

Figure 18.1 Ureolysis reactors: reactor b, aerated fixed-bed reactor with pump; reactor c, fixed-bed reactor with pump; reactor d, anaerobic stirred suspended growth reactor

Figure 18.2 Toilet area

of the TUHH test hall. Figure 18.2 shows the toilet area in the upper level; figure 18.3 shows the technical plant beneath.

The plant was designed for approximately 20 PE. The plant has been started-up with normal, untreated tap water. Students and scientific researchers of the institute have used the toilet facility and reported the kind of usage (urinating only, defecating only, or both). The plant was operated for almost 2.5 years.

18.4. Results and Discussion

18.4.1. Ureolysis

It could be shown, that the not aerated fixed bed reactor had the most effective results. During the experiments a close correlation between conductivity and NH_4 concentrations could be proven. In the aerated reactor, NO_2 could be measured, but

Figure 18.3 Technical test plant

disappeared from a retention time less than 60 minutes. Variations of recirculation had no influence on ureolysis efficiency. Simulating the entrance of nitrate into the reactors by addition of $Ca(NO_3)_2$, no nitrate losses could be detected in the aerated reactor, but a dissimilatory nitrate reduction ($NO_3 \rightarrow NO_2$) could be measured in not aerated reactors.

18.4.2. Riddle Screen

The riddle screen protected the membrane very well. The mesh size could be varied to investigate the effect on the dry matter content of the feces.

18.4.3. Biological Oxidation

Problems occurred during the start-up operation due to the low growth rates of the nitrifying bacteria. An accumulation of NO_2 could rarely be observed and NO_2 accumulation disappeared without external regulations. It was necessary to buffer the pH of the reactor. The reactor did not emit any smells—regarding this aspect it also could be operated without a cover. The reactor was not fed with substrate for several weeks, and then started again under full load within one day. The biocenoesis immediately started again without relevant efficiency losses. The oxidized black water in the reactor was of a brown color. During operation, the nutrients were increasingly concentrating in their oxidized forms (NO_3, SO_4, PO_4). The theoretical final molar nutrient concentrations (approximately equalling urine) could not be achieved so far.

18.4.4. Ultra-filtration

The membrane was under-dimensioned, because no smaller membrane unit was available. Within the 2.5 years of operation, no chemical cleansing of the membrane was necessary. By membrane filtration, the brown color changed into yellow.

18.4.5. UVC Radiation Respective Ozone Treatment

It could be shown, that after this step a clear, color and odorless flushing water can be produced, which is sensually indistinguishable from normal tap water. Drawbacks are the high energy consumptions. It could be shown, that the energy consumption of ozone treatment is significantly lower compared to decolorizing by UVC radiation.

During the period of operation, no major process disturbances (membrane blockage, etc.) of the process could have been observed. It could also been shown, that a black, humus-like compost can be hygienically safely produced from fecal matter and toilet paper. The optimal conditions for the composting process have been shown.

18.5. Conclusions

An integrating process is being introduced maximally conserving our renewable water resources. Potential recyclables are almost fully recovered: by-products of the process are compost, a fertilizer raw material and optionally and biogas. In combination with the "groundwater-loop process" (loop processing of grey water via a groundwater passage), even safe water self-sufficient settling becomes possible if 10 to 25 L of water per person and day can be renewed from local resources (rain, groundwater, river, lake, sea). Domestic wastewater can be physically eliminated—and with that all the known and unknown potential threats of public health and the environment caused by fecal contamination.

The urine flushing toilet can be seen as a core technology for highly efficient water management technologies, whereas for treatment and reuse a broad variety of possibilities exist. Two basically different reuses of grey water can be discriminated: Figure 18.4 illustrates the LooLoop Process in combination with an *external* grey water reuse (irrigation, landscaping, industrial process water, etc.). Figure 18.5 illustrates the indirect *internal* (after a passage through an artificial or natural groundwater body) reuse of grey water to drinking water, leading nearly to a water autarky with a remaining supply demand of 10 to 20 L per person and day, which can be produced from rain water in most regions of the earth.

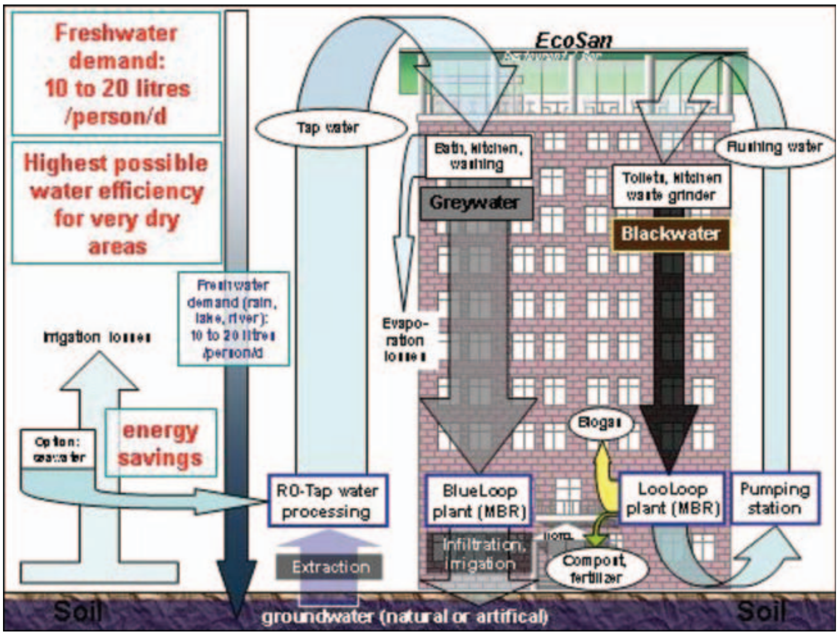

Figure 18.4 Black water loop and local water autarky

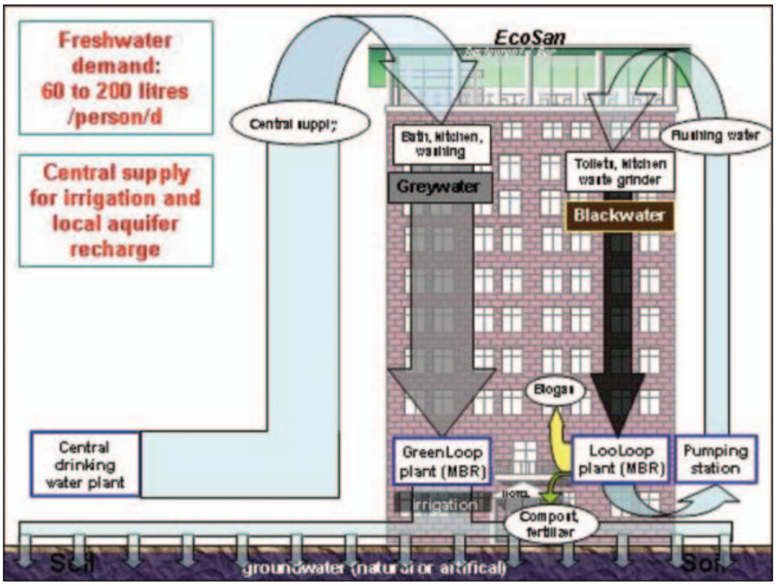

Figure 18.5 Black water loop and central supply with ground water recharge

18.6. Cost Comparison and Outlook

Environmentally open disposal of fecal contaminated waste water is causing several environmental impacts, which are difficult to express monetarily. For example, the monetary evaluation of indirect costs for public health, caused by flushing out of multiple resistances and pharmaceuticals from hospitals, hormones from private households, etc., is hardly possible. Also, the more efficient removal and recovery of nutrients (the LooLoop-Process is a zero-emission technology), and thus the lower nutrition of our surface and ground water bodies is difficult to evaluate monetary. In a serious cost comparison, these parameters should be considered.

In Germany, the subsidies for water and wastewater are widely abolished: the end user has to cover the actual costs, which are between 1 and 2 € per m^3 drinking water and 3 and 4 € per m^3 wastewater (4 to 6 €/m^3 in total). Due to the water frame work of the EU, water subsidies have to be abolished in all member states until 2010. This will cause a significant increase of waste- and drinking water fees in many countries.

As said earlier, main cost factor of the conventional centralized flow-through concept is the sewerage net causing 70 to 90% of the total investment in waste water infrastructure. Compared with centralized concepts, decentralized concepts can be more cost-efficient due to significant lower operating costs (Berliner Kompetenzzentrum Wasser, Tagung zum Abschlussbericht, Oldenburg, 2006).

First estimations are showing, that with the combination of the LooLoop-Process and a recycling of grey water a cost reduction of 30 to 50% is possible compared with 4 to 6 €/m^3 total costs. The bigger the connection size, the condensed the construction, the lower the total costs per m^3. Standardization of the plants for connection sizes (production-types) will further lower the costs of the LooLoop-Process. Market studies have shown, that there are several markets where these technologies can be financially very attractive:

1. Large functional buildings with high water consumption (hotels, hospitals, etc.).
2. New to construct settlements without sewer connection.
3. Existing settlements with sewer networks with high running costs (islands, mountainous regions, etc.).
4. All regions, where the water supply is based on sea water desalination.
5. Potentially all regions, where subsidies for water and waste water are cancelled.

The prices of nearly all raw materials and natural resources will increase in the future. This is especially valid for non-renewable resources like phosphate, which natural deposits will be totally depleted in 60 to 90 years. The German Bundestag stated in a response to a small inquiry (Bundestag, 2006), that an increased funding of efforts and technologies for recovery of Phosphate will be executed within the next years.

Due to decreasing availability of water (global warming), increasing water demand, increasing costs of water supply and disposal, increasing prices of raw materials, increasing funding of more efficient recovering technologies and decreasing plant costs, domestic waste water recycling technologies like the LooLoop-Process, have a very promising market potential in the future.

References

Berliner Wasser Betriebe. (2002). Available at: www.kompetenz-wasser.de/dt/projekte/proj-scst.htm.

Berliner Kompetenzzentrum Wasser, Tagung zum Abschlussbericht. (2006). Available at: http://www.kompetenz-wasser.de/fileadmin/user_upload/pdf/forschung/scst/SCST_Abschluss_Oldenburg.pdf.

Braun Ulrich. (1998). Verfahren und vorrichtung zum behandeln von abwässern (Method and device for sewage treatment), PCT number: PCT/EP98/03316.

Braun Ulrich. (2001). Verfahren und vorrichtung zum getrennten erfassen und ableiten von urin und fäkalien in urinseparationstoiletten (Method and device for separate collection and drainage of faeces and urine in urine separating toilets), PCT number: PCT/EP00/09700.

Bundestag. (2006). Phosphat-Versorgungssicherheit und Bodenschutz bei Phosphatdüngern. Drucksache, Bundesanzeiger Verlagsgesellschaft mbH.

Evans RJ. (1991). Death in Hamburg: Society and Politics in the Cholera Years. Penguin Books, New York, USA p. 21ff.

Gesellschaft für Technische Zusammenarbeit. (2002). Available at: www.ecosan.de.

Gesellschaft für technische Zusammenarbeit. Available at: www.gtz.de/ecosan/symposium.html

Lange J and Otterpohl R. (2000). Abwasser. Handbuch zu einer zukunftsfähigen Wasserwirtschaft. Pfohren, Mallbeton Verlag, stark erweiterte Auflage 2000.

Otterpohl R, Albold A, Oldenburg M. (1999). Source control in urban sanitation and waste management: 10 options with resource management for different social and geographical conditions. Water Sci Technol 39(3/4):153–160.

OtterWasser. (2002). Available at: www.otterwasser.de.

Paris S and Wilderer PA. (2002). Integrierte Ver- und Entsorgungskonzepte im internationalen Vergleich. GWA, Gewässerschutz, Wasser, Abwasser, **Band 188**, Aachen.

Skjelhaugen OJ. (1998). System for local reuse of blackwater and food waste, integrated with agriculture; Technik anaerober Prozesse, TUHH, Technische Universität Hamburg-Harburg, DECHEMA-Fachgespräch Umweltschutz.

Winblad U. (1998). (Hrsg.) Ecological Sanitation. SIDA, Stockholm.

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Chapter 19
Contribution of Sewage Sludge to the Fertility of the Soil and the Growth of Barley (Hordium Vulgare *L*) Variety Jaidor

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Abstract A greenhouse pot experiment was conducted at the Ben M'hidi University Centre during the year 2002/03 to study the effects of different levels of sewage sludge on soil properties and yield of barley (Hordium Vulgare *L*) Variety Jaidor. The treatments consisted of 20, 40, 60 t ha^{-1} of organic fertilizer (sewage sludge); 35 and 70 kg ha^{-1} of mineral fertilizer (urea) and a check (without fertilization).

The results showed that the response of the crop for most variables was very well expressed at the application rate of 40 t ha^{-1} of sewage sludge. A significant increase in the number of tillers, kernels per spike (KN/S) and spikes was found in the sewage sludge treated soils.

An improvement in soil physical and chemical properties was noticed with increasing addition of sewage sludge. The amendment effect was highly significant At 40 t ha^{-1}, the sewage sludge produced the best results in carbon content of the soil with 2.50%. The soil porosity and the hydraulic conductivity near saturation were significantly improved by the addition of sewage sludge.

19.1. Introduction

The use of sewage sludge (SS) in agriculture has become a common practice over the past years. Because it contains plant nutrients and organic matter, SS may be used to supplement or replace commercial fertilizers for crop production (Bozkurt, 2003).

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The beneficial effects of using SS in agriculture have been proven by numerous researchers. It has been shown that SS application improves the physical, chemical and biological properties of soil (Aggelides and Londra, 2000; Benitez et al., 2001; Selivanovskaya et al., 2001; White et al., 1997). Nutrients contained in SS increase plant biomass and yield (Brofas et al., 2000; Coogar et al., 2001; Snyman et al., 1998). According to the Food and Agriculture Organization of the United Nations (2000), SS can be used as a valuable source of plant nutrients to substitute the chemical fertilizer.

The SS application may lead to the accumulation of a number of potentially harmful components such as heavy metals in soil and crops. The presence of heavy metals in the applied sludge can result in phytotoxic effects, soil and water contamination and accumulation of heavy metal in food supplies (Bozkurt, 2003; Keller et al., 2002).

Proper management of SS application is essential for maximizing its beneficial effects and minimizing its adverse effects. Such management should consider several aspects as content of heavy metals and other contaminants, the crop type and its nutrient requirement, the amount and form of nutrients contained in the SS and soil chemical and physical properties (Mohamed and Athamneh, 2004).

The objectives of the experiment reported here were to study the effect of application of different ratios of SS on the fertility of a silty clay soil and the growth and the yield of barley (Hordium Vulgare *L*) variety Jaidor.

19.2. Materials and Methods

A greenhouse pot experiment was conducted on barley (Hordium Vulgare *L*) Variety Jaidor at the University of Oum El Bouaghi during the year 2002/2003. In a randomized block design with five replications, the following treatments were studied: control (C), SS at 20, 40 and 60 t ha^{-1}, mineral nitrogen (N) fertilizer at 35 and 70 kg ha^{-1}. The SS used in the experiment was obtained from the wastewater treatment plant of Ain Sfiha (Setif, Algeria). Dried in drying beds for more than six months, the SS was generated through an activated sludge process. The SS was air-dried and ground to about 5 mm granules and applied as such to the soil according to the treatments. The characteristics of this matrix are shown in Table 19.1. Before conducting the experiment a soil sample was sieved through 2 mm screen and was analyzed for general characteristics: Soil pH was measured on 1:2.5 soil–water suspension, soluble salts were determined by measuring the electrical conductivity of 1:5 soil–water suspension. The other characteristics were determined by standard methods and are given in Table 19.1.

Seven-liter pots were used as the experiment units. Each pot was filled with 7 kg air dry and sieved through 5 mm screen surface soil (top 20 cm). In each pot, the amount of the SS according to the treatments was mixed in the upper 5 cm of the soil. The pots were irrigated after application of SS with distilled water to

Table 19.1 Characteristics of the soil and the sewage sludge used in the experiment

Properties	Units	Soil	Sewage sludge
pH	—	7.82	7.6
CE	dS/m	0.28	5.8
Ca	(meq/L)	23.39	14.0
Mg	(meq/L)	6.81	18.2
K	(meq/L)	2.6	2.6
Na	(meq/L)	3.03	0.6
OM	%	1.73	2.82
C	%	1.01	16.4
N	%	0.24	1.7
C/N	—	4.12	9.64
CEC	meq/l	38.4	—
Total P	%	-	0.70
Texture	—	Silty clay	—

CE: electrical conductivity (dS/m); Mg: magnesium (meq/L); Ca: calcium (meq/L), K: potassium (meq/L); OM: organic matter; C: carbon (%); N: nitrogen (%).

reach the moisture content at saturation percentage and left for two weeks to elucidate the damage on seedlings and their roots resulted from the heat decomposition and to attain equilibrium. The inorganic fertilizer (urea) was applied at the tillering-jointing stage of the crop.

Barley (Hordium Vulgare *L*) variety Jaidor, was sown on March 2, 2003 at a 200 seeds m^2 rate. Germination was noted a week after and the crop was harvested on June 10, 2003. A presoaking irrigation was applied to each pot to the level of field capacity of the soil. The pots were watered with an interval of three days between irrigations when the soil reached around half of the field capacity. The urea was applied to the soil on April 10, 2003 at the tillering-jointing stage.

During the experiment, the electrical conductivity and the pH of the soil were monitored once in two weeks using a Consort C535 Multiparameter Analyzer.
The growth and yield parameters of the plant such as plant height, leaf area (LA), tillers number (TN), spike number (SN), kernels number (KN/S), thousand kernels weight (TKW), relative water content (RWC), kernels yield (KY), root volume (RV) and biomass at tillering (TB), jointing (JB), heading (HB) and maturity (MB) were recorded. Soil samples were taken after harvest to determine the fertility status: the bulk density (D_b), the porosity, the hydraulic conductivity (HC), carbon and N.

The collected data were subjected to an analysis of variance. Contrast was employed to test the significance of the following treatments effects: SS vs C, N vs C and SS vs N.

19.3. Results and Discussion

19.3.1. Effect of SS Application on Crop

The analysis of variance showed a significant treatment effect for the whole variables measured compared to the check and the mineral application except for the thousand kernel weight (TKW) (Table 19.2). The non-significant treatment effect for the TKW could be explained by the fact that this trait is formed when environmental growth conditions become less favourable (Tamrabet et al., 2006).

The comparison between the SS amendment and the control means indicated that the organic fertilization was beneficial to the expression of the measured variables of the crop except for the TKW and the RWC (Figure 19.1). The relative increase in the measured variables means ranged from 16% for the crop height to 528% for the RV. The amendment effect was negative for the TKW and the RWC, which are reduced by an average of 5% and 35% respectively, relatively to the mean expressed by the control treatment (Figure 19.1). The reduction in TKW could be attributed to the fact that the SS applied has a more pronounced effect on the accumulated above ground biomass than on grain yield (GY). During the experiment, it was noticed that check and mineral fertilized plants were more tardive than the rest of treatments. Consequently, younger plants kept more moisture in their leaves than the organic fertilized ones.

As evident, from Figure 19.1, a highly significant increase was recorded in LA (69 to 75%), RV (485 to 528%), TN (16 to 36%), TB (34 to 80%), biomass at JB (123 to 161%), HB (141 to 151%), MB (119 to 136%), number of spikes/pot

Table 19.2 Means squares of the analysis of variance of the measured variables

	B vs T	B vs N	N vs T	Treatment	Residuals
LA	156.43***	155.18***	6.28**	48.03***	0.90
RWC	2613.60***	4439.04***	2.13 ns	1162.05***	38.60
RV	273.38***	317.40***	5.33 ns	134.48***	1.28
Tallage TB	40.02***	67.20***	0.02 ns	23.69***	1.09
JB	7780.74***	7063.33***	421.13*	2759.60***	62.49
HB32634.01***	15720.45***	5908.03***	8120.66***	185.60	
MB	35420.66***	18578.20***	5752.74***	8820.54***	39.22
PHT	344.54**	0.20 ns	318.18***	109.89***	4.24
TN	7684.02***	9110.41***	132.30***	3247.57***	15.59
KY	0.77***	0.20***	0.24***	0.17***	0.01
SN/POT	821.40***	423036***	136.53***	239.17***	4.57
TKW	19.27 ns	36.51 ns	0.13 ns	13.57 ns	21.61
KS	451.00***	2.27 ns	357.14***	103.25***	8.96
dll	2	1	1	5	20

**: $p < 0.05$; ***: $p < 0.01$; ns: not significant; BM: biomass at maturity (g); HB: biomass at heading (g); JB: biomass at jointing (g); KS: number of kernels/spike; LA: leaf area (cm^2); N: nitrogen; PHT: plant height (cm); RWC: relative water content (%); RV: root volume (cm^3); SN/POT: spike number/pot; TKW: thousand kernels weight (g); TB: biomass at tillering (g); TN: tillers number/pot.

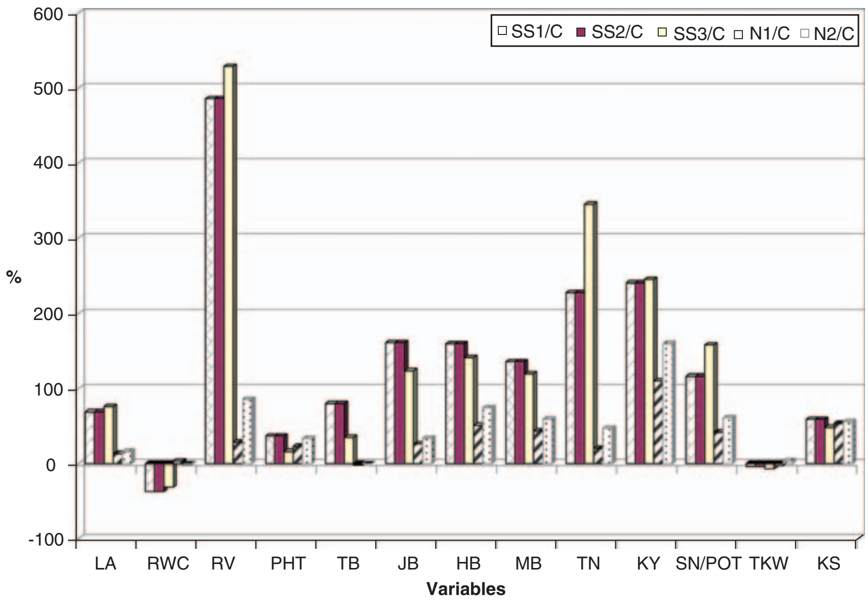

Figure 19.1 Contribution of the organic and mineral amendments to the increase in the mean values of the measured traits of the crop relatively to the mean values of the control

(SN/POT; 116 to 158%) and (240 to 245%) with incorporation of SS as compared to check treatments.

The improvement of the TN increases the number of spikes (Triboi and Gachon, 1980) and both the number of spike and kernels/spike determine the crop yield.

In the present study, SS3 treatment produced the best density of spikes/pot with 27.2 followed by mineral (18.5) and check (12.40) respectively.

The application of SS brought significant changes in the crop GY. The maximum GY was recorded in the treatment getting 60 t ha^{-1} of SS with 0.69 kg m^{-2} relatively to the rest of treatments and on average, the SS applications recorded the best yield by 0.64 kg m^{-2} followed by mineral and check treatments by 0.47 and 0.20 kg m^{-2} respectively.

Figure 19.2 and Table 19.3 showed that the maximum improvement in biomass was recorded in the treatment getting 40 t ha^{-1} SS (SS2), followed by the 60 t ha^{-1} application.

Vansholl (2002) stated that the SS stimulates the activity of the soil microorganisms that make the substances contained in the soil available for the crop. In addition to the N, the SS is a source of microelements that are essential for the growth of the crop. According to Jamil (2006), the increase in the above-ground biomass of the crop could be due to the abundant supply in organic matter and other nutrients by the SS. The positive effect of SS on the crop is due to its improvement of the physico-chemical characteristics of the soil (Halitim and Benbadji, 1978).

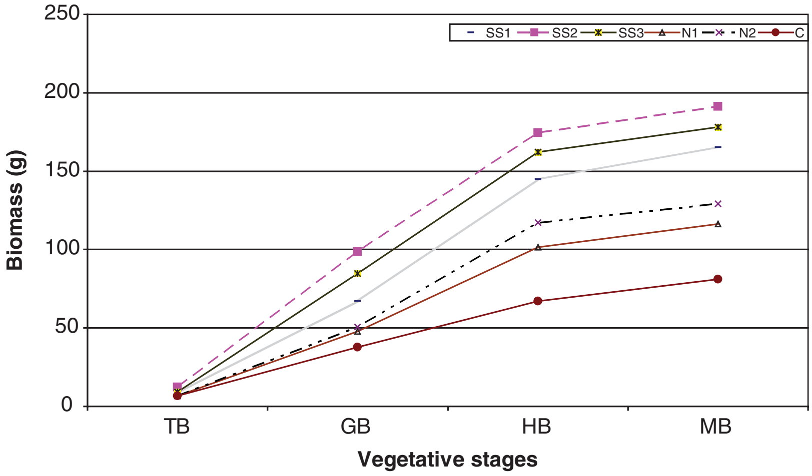

Figure 19.2 Contribution of the organic and mineral amendments to the increase in the mean values of biomass

Table 19.3 Mean values of the different treatments

	SS1	SS2	SS3	N1	N2	C
LA	15.20	15.68	16.33	10.51	10.79	9.28
RWC	64.80	53.40	58.20	88.00	84.00	85.20
RV	8.50	20.50	22	4.5	6.5	3.5
PHT	45.07	47.43	40.36	42.71	46.23	34.70
Tallage TB	8.80	12.24	9.16	6.68	6.76	6.80
JB	67.20	98.73	84.60	47.80	50.60	37.80
HB144.96	174.36	162.14	101.48	117.12	67.20	
MB	165.20	191.40	178.26	116.20	129.08	81.10
TN	47.00	61.60	83.60	22.60	27.60	18.80
KY	0.57	0.68	0.69	0.42	0.52	0.20
SN/POT	22.80	26.80	32.00	17.60	20.00	12.40
TKW	40.40	41.00	39.60	41.40	44.20	42.60
KS	31.59	29.97	27.89	28.97	29.42	18.85

BM: biomass at maturity (g); C: control; HB: biomass at heading (g); JB: biomass at jointing (g); KS: number of kernels/spike; LA: leaf area (cm^2); N: nitrogen; PHT: plant height (cm); RWC: relative water content (%); RV: root volume (cm^3); SN/POT: spike number/pot; TKW: thousand kernels weight (g); TB: biomass at tillering (g); TN: tillers number/pot.

19.3.2. Effect of SS on Soil Properties

The analysis of variance showed a significant treatment effect for the carbon content and the N of the soil (Table 19.4). The non-significant treatment effect for the electrical conductivity, pH and the D_b could be explained by the fact that these characteristics need longer time to be able to evaluate their impact on the soil.

Table 19.4 Means squares of the analysis of variance of the physical and chemical characteristics of the soil

	SS vs C	SS vs N	N vs C	Treatment	Residuals
Carbon	2.12***	2.69***	0.02 ns	1.38***	0.07
N	0.02***	0.02***	0.00 ns	0.01***	0.00
Porosity	225***	84.10 ns	53.39 ns	63.39 ns	41.62
K(6)	4607.90***	5088.89***	117.28 ns	2089.33 ns	481.69
K(30)	67.82 ns	43.54 ns	8.10 ns	42.25 ns	28.61
dll	2	1	1	5	20

K(6), K(30): Hydraulic conductivity at the pressions of 0.06 and 0.30 kpa, respectively; **: $p < 0.05$; ***: $p < 0.01$; ns: not significant; C: control; N: nitrogen; SS: sewage sludge.

Table 19.5 Mean values of the different treatment

	Units	SS1	SS2	SS3	N1	N2	C
Carbon	%	0.90	2.25	1.10	0.48	0.69	0.58
N	%	0.16	0.09	0.11	0.02	0.07	0.018
Porosity	%	47	52	51	43	47	40
K(6)	mm/h	80.92	81.89	129.20	48.63	53.94	41.91

K(6): Hydraulic conductivity at the pression of 0.06 kPa; C: control; N: nitrogen; SS: sewage sludge.

The SS2 application gave the highest value of organic carbon content (2.25%) over the rest of treatments (Table 19.5). The application of increasing doses of SS boosted the biological activity and consequently, increased the mineralization of organic matter within the SS.

The organic amendment showed that the SS treatments increased the carbon content of the soil by (100 to 400%), N by (400 to 788%), porosity (17 to 27.5%) and the HC by (93 to 208%) relatively to the mean values of the check treatment (Figure 19.3).

Increases in organic matter can be confirmed by the high content in organic carbon in the SS (Table 19.5) (Clapp et al., 1986; Moore et al., 1995). Krause (1988) stated that high rates of SS improved the aggregate size and stability with the increases in soil organic matter. Jamil et al. (2006) reported that SS application improves soil aggregation and increases soil aeration.

The analysis of variance showed that the effect of SS was significant for 0.06 kPa of applied pression. At this pression, the highest value of HC (129.20 mm h^{-1}) corresponds to the SS3 application, where it is only 41.91 mm h^{-1} for the check and 53.94 mm h^{-1} for the mineral application (Table 19.5). According to Al-Samarrai (2005), the HC increases with increasing organic amendment doses applied to clay loam and sandy clay loam soils.

The improvement in soil porosity is attributed to the high content of the SS in organic matter which has a powerful effect on aggregation process. The findings of Al-Sammarai (1999) and Mazurak (1977) are in good agreement with present results.

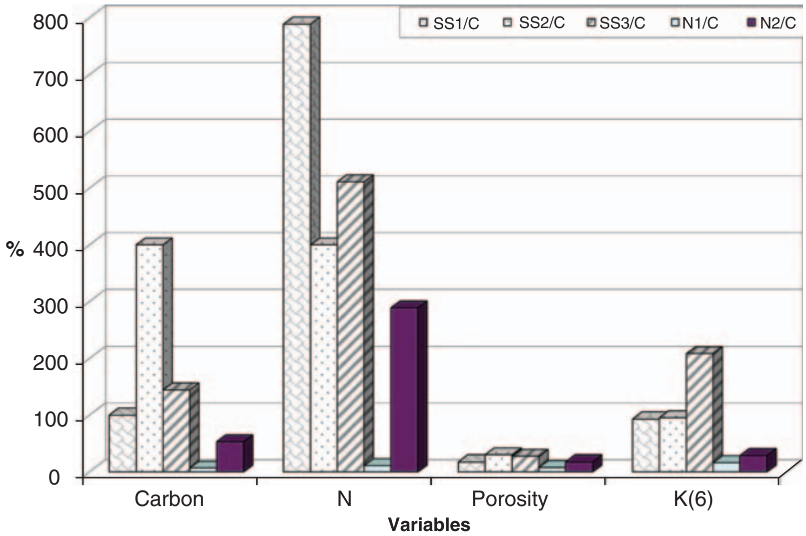

Figure 19.3 Contribution of the organic and mineral amendments to the increase in the mean values of the measured traits of the soil relatively to the mean values of the control

19.4. Conclusion

The application of the SS to the soil increases the plant production. The effect of the SS on the crop is much pronounced on the above ground biomass. As a result, it is suitable to use it for the production of forage crops.

The SS affects positively as well the soil physical, chemical and biological properties. It improves the structure and the water retaining capacity of the soil and provides macro- and microelements to the soil and consequently of many crops requirements.

References

Aggelides SM and Londra PA. (2000). Effect of compost produced from town wastes and sewage sludge on the physical properties of a loamy and clay soil. Biores Technol 71:253–259.

Al-Sammarai MA. (1999). Soil physical properties and sugar beet growth as affected by the application of soil amendments. Mesopotamia J Agric 31:25–35.

Al-Sammarai MA, Al-Lami MM, Abdul-Majeed H. (2005). Effect of soil amendments on hydraulic characteristics of two types of soil. Asian J Plant Sci 4:597–599.

Benitz EM, Gomez M, Gallardolaro F, Nogales N. (2001). Biosolid and biosolid ash as sources of heavy metals in plant-soil system. Water Air Soil Pollut 132:75–87.

Brofas G, Michopoulas P, Alifragis D. (2000). Sewage sludge as amendment for calcareous bauxite mine spoils reclamation. J Environ Qual 29:811–816.

Bouzerzour H, Tamrabet L, Kribaa M. (2002). Response of barley and oat to the wastewater irrigation and to the sludge amendment. In: Proceedings of the International Seminar: Biology and Environment. September 9–11, University Mentouri, Constantine, Algeria, pp. 71.

Bozkurt MA and Yarilgaç T. (2003). The effects of sewage sludge applications on the yield, growth, nutrition and heavy metal accumulation in apple trees growing in dry conditions. Turk J Agric Forest 27:285–292.

Food and Ariculture Organization of the United Nations. (2000). Users manual for irrigation with treated wastewater. Regional Office for the Near East, Cairo, Egypt, pp. 53–56.

Francois JM and Laflèche M. (1981). Utilisation agricole des résidus urbains dans les sols limoneux des plateaux de brie. Ministère de l'environnement et du cadre de vie, pp. 43.

Coogar CG, Bary AI, Fransen SC, Sullivan DM. (2001). Seven years of biosolids versus inorganic nitrogen applications to tal fescue. J Environ Qual 30:2188–2194.

Guivarch A. (1998). Valeur fertilisante a court terme du phosphore des boues de station d'épuration urbaines. Rapport intermediaire, ADEME.

Halitim A and Benbadji A. (1978). Etude expérimentale de l'influence du compost urbain sur les sols et la production de tomate en présence d'eau chargée en NaCL. Agro 8:58.

Jamil M, Qassim M, Umar M. (2006). Utilization of sewage sludge as organic fertilizer in sustainable agriculture. J Appl Sci 6:531–535.

Keller C, McGrath SP, Dunham SJ. (2002). Trace metal leaching through a soil grassland system after sewage sludge application. J Environ Qual 31:1550–1560.

Mazurak AP, Chenin L, Thijeel AA. (1977). Effect of beef cattle manure on water stability and soil aggregates. Soil Sc Soc Am J 41:613–615.

Mohammad MJ and Athamneh BM. (2004). Changes in soil fertility and plant uptake of nutrients and heavy metals in response to sewage sludge application to calcareous soils. J Agronomy 3:229–236.

Mollier A. (1999). Croissance racinaire du mais (Zea–Mais-L) sous déficience en phosphore: Etude expérimentale et modélisation. Université Paris Sud, unité d'agronomie INRA, Bordeaux.

Triboi G. (1981). Modèle d'élaboration du poids du grain chez le blé tendre. Agro 10:183–199.

White CS, Loftin SR, Aguilar R. (1997). Application of biosolid to degraded semiarid rangeland: Nine-year responses. J Environ Qual 26:1663–1671.

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Chapter 20
The Use of a Pilot-Scale Membrane Bioreactor in Treating Domestic Wastewater with Variable Characteristics for Potential Water Reuse on a University Campus

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Abstract The aerobic treatability of domestic wastewater from a university campus area with diurnally and seasonally variable characteristics was investigated using a pilot-scale submerged membrane bioreactor (MBR) operated for eight months. Operating conditions including sludge retention time (SRT), mixed liquor suspended solids (MLSS) concentration, permeate flux and hydraulic retention time (HRT) were varied during the operation to investigate their impacts on overall treatment performance, water production and membrane fouling. Although the influent characteristics were highly variable, including sudden sharp increases in total dissolved solids (TDS) and organic matter concentrations and wastewater temperatures were as low as 6 °C during winter months, the MBR system performed well throughout its operation. The treatment performance was not negatively influenced by variations in food to microorganism ratio, MLSS and dissolved oxygen concentrations, temperature, SRT, organic loading rate, specific substrate removal rate and permeate flux. Sustainable nitrification and organic carbon removals were achieved even during periods with extreme conditions. Irreversible fouling of membranes did not occur during the eight months of operation. No chemical cleaning was performed during the six months of operation at normal flux (23 to 25 L/m^2-h), except routine back-pulsing with permeate. For the high flux operation (36 to 39 L/m^2-h), chemical maintenance cleaning was employed two times over two months, which consisted of back-pulsing membranes with chlorine dosed

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(250 mg/L) permeate for five minutes without draining the MBR tank. No further intensive chemical cleaning was necessary even at high flux conditions. Overall, eight months of pilot-scale tests indicated the robustness of MBR process in terms of achieving very high quality of treated water without any operational limitations including fouling and permeability reduction problems. This study will guide the decision on a potential full-scale MBR application and reuse of the treated wastewater for irrigation in the campus area.

20.1. Introduction

Membrane bioreactor (MBR) technology is now more widely applied in wastewater treatment plants mainly due to decreasing costs, improvements in membrane performances (i.e., increased water productivity) and the surge of water reuse. It has been used for various specialty treatment applications for nearly 30 years (Daigger et al., 2005; Stephenson et al., 2000). MBR is a suspended growth-activated sludge treatment system that relies on membrane equipment for liquids/solids separation before discharge of the treated effluent, thereby replacing the solids separation function of the secondary clarifier (Gunder, 2001; van der Roest et al., 2002). Replacing external membranes with immersed ones reduced capital and operating costs and increased the range of applications for which MBRs can be cost-competitive (Adham et al., 2001). Furthermore, membrane costs have declined by an order of magnitude over the past decade, dramatically reducing MBR costs (Daigger et al., 2005). The MBR process was demonstrated to be cost-effective over conventional water reclamation systems for urban irrigation (Adham and Trussell, 2001).

There are a number of benefits associated with MBRs compared to conventional wastewater treatment processes. Biomass is completely retained in the bioreactor allowing the proliferation of slow-growing microorganisms with low yields. Relatively long solid retention time (SRTs) can be achieved thus reducing sludge production and improving nitrification. SRT can be reliably separated from hydraulic retention time (HRT) in a MBR system, allowing independent control of both. Because secondary clarifiers and/or conventional filters are eliminated from the process, plant footprint area can be reduced allowing for upgrades without area expansion. Unlike clarifiers, in the MBR the quality of solids separation is not dependent on the mixed liquor suspended solids (MLSS) concentration or characteristics such as settleability. Because elevated MLSS concentrations are possible in the MBR, the bioreactor volume can be reduced, higher organic loadings can be applied and the biomass can be more tolerable to shock toxic loadings. Excellent effluent quality can be obtained generally suitable for reuse as membranes provide high removals of pathogens including bacteria, protozoa and viruses resulting in excellent physical disinfection. Also, MBRs allow for exceptional versatility in the design of new plants or the retrofitting of existing wastewater treatment plants because membranes can be added in modules into existing aeration tanks (Cicek et al., 1998; Daigger et al., 2005; Fane and Chang, 2002; Gander et al., 2000; Kraume et al., 2005; Lesjean et al., 2004; Stephenson et al., 2000; Visvanathan et al., 2000; Yang et al., 2006). However, membrane fouling, which reduces productivity and increases maintenance and operating costs, is one of the major drawbacks of MBR processes (Chang et al., 2002; Judd, 2004).

The main objective of this study was to investigate the treatability of domestic wastewater from a university campus area with diurnally and seasonally variable characteristics using a pilot-scale submerged MBR unit. The unit was continuously operated aerobically for eight months to achieve nitrification and carbon removal. Operating conditions, including SRT, permeate flux and HRT, were varied during the operation period to investigate their impacts on overall treatment performance, water production and membrane fouling. A total of three major operational phases were tested during the eight-month period: no sludge wastage, flux: 23 to 25 L/m^2-h; SRT: 20 days, flux: 23 to 25 L/m^2-h; and SRT: 20 days, flux: 36 to 39 L/m^2-h (high flux conditions). The operation periods for phases 1, 2 and 3 were 125, 57 and 51 days, respectively.

20.2. Methodology

The pilot-scale operation was conducted at the campus of Suleyman Demirel University, Isparta, Turkey. A single-tank MBR (ZW® 10, Zenon Environmental Inc.) unit containing an immersed hollow fiber membrane module (0.9 m^2 active membrane area) was used in the work. The membrane module has a nominal pore size of 0.04 µm. The treatment capacity of the unit was 380 to 1,500 L/day. The bioreactor (high-density polyethylene (HDPE) tank with 230-L solution volume) was fed with raw domestic wastewater withdrawn from the sewage system of the university campus. The bioreactor was initially seeded with biomass obtained from the activated sludge process of a nearby municipal wastewater treatment plant. The MBR system consisted of a 1000-L polyethylene feed tank, 25-L HDPE back-pulse tank, 1000-L polyethylene permeate collection tank and the bioreactor, which was located in a closed building. The raw wastewater was delivered from the sewage system to the feed tank using a wet pump and 25-m tubing (1.27-mm ID), which was insulated to prevent freezing during winter months. The wastewater was then pumped from the feed tank to MBR using a peristaltic pump after prescreening (1-mm) to remove materials which may damage membranes.

Permeate was collected from the MBR through the membrane module using a bidirectional vacuum pump, which was also used for backpulsing the membranes with permeate to remove foulants deposited on the fibers. Permeate was pumped from the backpulse tank to the permeate collection tank using a peristaltic pump. An air supply blower (maximum flow of 119 L/minute) with a control valve and flow meter was used to provide air for bio-oxidation and membrane scour and to adjust aeration velocity. The system was partially automated with a central control panel. The bioreactor and associated equipment were mounted on an epoxy coated carbon steel frame. A routine back-pulse regime (15 seconds after each 10 minutes of permeate production) was employed by automatic reversal of the permeate pump. The flowrate of backpulse was twice that of normal permeation (0.75 and 1.10 L/minute in phases 1 and 2 and phase 3, respectively).

Samples were taken from raw wastewater (feed) and permeate and analyzed for chemical oxygen demand (COD), biochemical oxygen demand (BOD_5), total organic carbon (TOC), nitrate, nitrite, ammonia, total kjeldahl nitrogen (TKN), total nitrogen, total phosphorus, suspended solids, turbidity, total dissolved solids, conductivity, pH, fecal coliform, total coliform and total organisms. Sampling frequency was either daily or twice a week depending on the parameter measured. Feed and permeate flow rates, transmembrane pressure (TMP) and air flow rates were measured and/or controlled at least twice a day. Furthermore, mixed liquor dissolved oxygen (DO), temperature, pH, conductivity, mixed liquor suspended solids (MLSS) and mixed liquor volatile suspended solids (MLVSS) levels in the bioreactor were frequently measured. Based on these measured parameters, membrane permeability, net, instantaneous and temperature corrected flux, food to microorganism ratio (F/M), organic loading rate and specific substrate utilization rate were calculated.

In the first operation phase, sludge was not wasted to achieve MLSS concentrations as high as 10,000 mg/L and to investigate the impacts of infinite SRT conditions on biological treatment performance and membrane filtration. In other words, in this phase, extended aeration biological treatment systems with low organic loading and very low biomass growth rates were simulated. HRT and target permeate flux employed in this phase were 10 hours and 25 L/m^2-h, respectively, values typically employed in full-scale MBR plants. In the second operation phase, 20 days of SRT was employed without changing other variables. Thus, the aim was to investigate the impact of SRT alone on treatment performance and filtration. The target permeate flux was increased to 39 L/m^2-h in the third operation phase with a SRT of 20 days, in an effort to investigate the impact of very high flux and decreased HRT (seven hours) conditions.

20.3. Results and Discussion

The pilot-scale MBR unit was operated between late January and mid-August, 2005. While the operational parameters were measured from the immediate start, the water quality parameters were regularly measured starting from early March. During the entire operation period, the characteristics of raw wastewater exhibited both diurnal and seasonal variations mainly due to infiltration to the sewage system especially in wet weather and changes in student populations during weekends, holidays, etc (Table 20.1). Wastewater temperatures dropped to as low as 6 °C in winter months when the operation was started. Organic loading (as measured by BOD_5 and COD) to the unit exhibited a wide variation. Similar significant variations were also observed for TSS and turbidity. One major problem was the sudden increases in TDS (as measured by conductivity) concentrations, which occurred few times during the eight-month operation. Peaks in feed conductivity values as high as 8300 μs/cm (6600 μs/cm in the MBR) was noted, which was due to the

Table 20.1 Treatment performance of the MBR during eight months of operation

Parameter	Feed Average (Min. to Max.)	Permeate Average (Min. to Max.)
BOD_5 (mg/L)	205 (95–380)	1.5 (1–6)
COD (mg/L)	316 (105–695)	9.5 (5–60)
NH_3-N (mg/L)	24 (7–44)	1.0 (0.1–10.5)
NO_3-N (mg/L)	14.5 (5.5–37.5)	32.0 (16.0–45.5)
NO_2-N (mg/L)	0.7 (0.1–1.6)	0.5 (0.1–2.6)
TKN (mg/L)	34 (18–52)	1.7 (0.1–3.6)
TN (mg/L)[a]	49 (29–70)	34 (18–48)
TP (mg/L)[b]	4.8 (2.0–7.0)	3.4 (0.6–5.0)
TSS (mg/L)	109 (52–276)	0.6 (0.3–1.3)
Turbidity (NTU)	90 (40–213)	0.08 (0.02–0.41)
Conductivity (µS/cm)	1441 (642–8300)	1329 (556–4190)
pH	7.91 (6.95–8.32)	7.74 (7.36–8.26)
Temperature (°C)[c]	18.8 (6.0–25.5)	
Total organism (cfu/100 mL)	1.67×10^{10} ($8.0 \times 10^8 - 6.0 \times 10^{10}$)	2.5×10^2 ($6.0 \times 10^1 - 1.06 \times 10^3$)
Total coliform (cfu/100 mL)	5.4×10^9 ($6.0 \times 10^7 - 2.0 \times 10^{10}$)	1.6×10^2 ($4.0 \times 10^1 - 8.4 \times 10^2$)
Fecal coliform (cfu/100 mL)	5.8×10^8 ($1.0 \times 10^7 - 3.0 \times 10^9$)	1.9×10^1 ($2.0 \times 10^0 - 1.4 \times 10^2$)

[a] Anoxic zone was not employed.
[b] Anaerobic zone was not employed.
[c] Temperature values in the MBR tank.

regular cleaning events in buildings and dormitories. After each prescheduled cleaning, similar peaks in feed conductivity and surface-active agent concentrations were observed. Except during these peak loading events, the conductivity values in the MBR were generally 1,000 to 1,500 µS/cm. Since ultrafiltration is employed in the MBR dissolved salts are not removed. The pH values of raw wastewater exhibited less variation (7 to 8.3) owing to moderately high alkalinity concentrations (150 to 200 mg/L as $CaCO_3$) in the campus source water. Permeate pH values were generally less than (about 0.05 to 0.40 units) those of corresponding feed values which was due to the consumption of alkalinity during carbon oxidation and nitrification. The average pH in the MBR was 7.6 during all operation.

Although the influent characteristics were highly variable including sudden increases in TDS levels and the wastewater temperatures were as low as 6 °C during winter months, the MBR consistently performed well throughout its eight months of operation (Table 20.1). Sustainable organic carbon removal and nitrification were achieved even during these extreme conditions (Figures 20.1 and 20.2). This sustainable and successful performance of the biological activity demonstrates the presence of a robust biomass mixture (i.e., halophilic bacteria tolerable to high salt loadings) in MBRs that could respond to sudden variations. This can be attributed to the fact that all biomass is retained in MBRs by micro- or ultrafiltration, resulting in high MLSS concentrations and a community structure with a wide spectrum of degradation capability. In addition to the advantages of membrane filtration, this is another

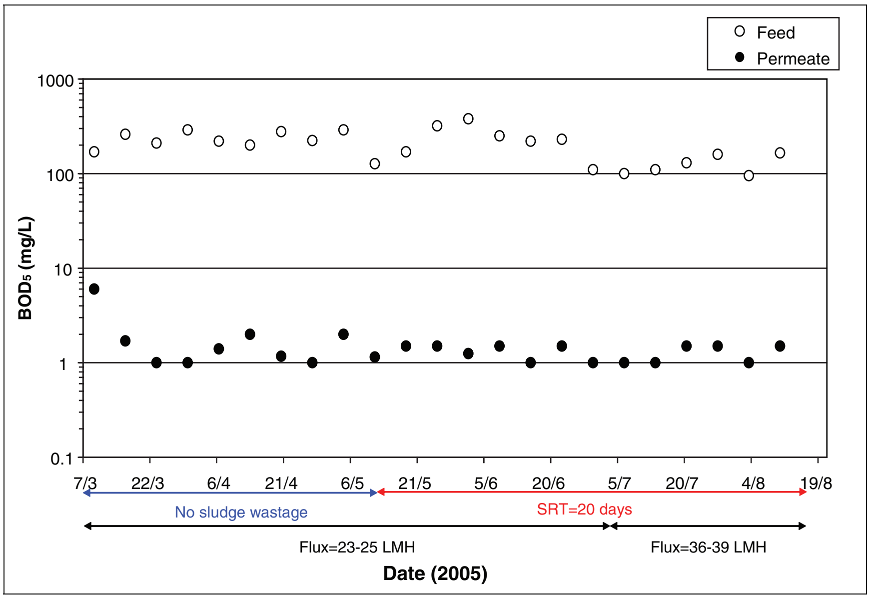

Figure 20.1 BOD_5 removal throughout membrane bioreactor operation

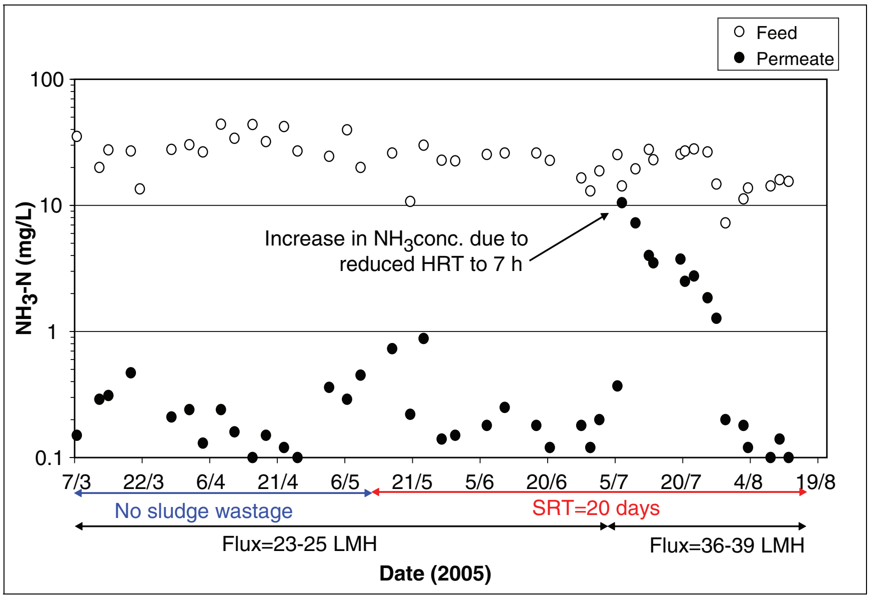

Figure 20.2 NH_3-N removal throughout membrane bioreactor operation

advantage of MBRs over conventional activated sludge processes from a biology perspective. The average BOD_5 and total COD (dissolved and particulate) concentrations in permeate were 1.5 and 9.5 mg/L, respectively. Permeate COD concentrations were generally less than 10 mg/L, except for the start-up phase (first three weeks) in which COD concentrations as high as 60 mg/L were measured. These COD concentrations, which were much lower than those obtained in conventional activated sludge processes, suggest that a significant portion of both dissolved and particulate organic matter is removed in the MBR through biological activity and membrane separation. Nitrification was negatively affected only in the initial part of the third phase in which HRT was suddenly reduced to seven hours due to increased target permeate flux. Apparently, nitrifiers with slower growth rates than those of heterotrophic bacteria could not immediately adjust to this change. However, nitrifiers recovered within about two weeks and permeate NH_3-N concentrations as low as 0.1 mg/L were achieved. Nitrite concentrations in the feed and permeate were close to each other (generally less than 1 mg/L), indicating that complete nitrification to nitrate was achieved, as supported by increased nitrate concentrations in permeate.

In addition to generally tolerating variations in raw wastewater characteristics, organic carbon removal (as measured by BOD_5 and COD) and nitrification were also not negatively influenced by variations in operational parameters including MLSS concentration, F/M ratio, organic loading rate, specific substrate removal rate, SRT and DO concentrations. It is known that conventional activated sludge systems, especially the ones employing nitrification, may experience instability and even failure when such operational parameters fluctuate. The range of MLSS concentrations in MBR was 3,000 to 7,100 mg/L and 2,600 to 5,200 mg/L at no sludge wastage conditions and SRT of 20 days, respectively. Although the MBR was operated for 125 days without wasting sludge, the maximum MLSS concentration achieved was only 7,100 mg/L, which may be caused by two reasons: the wastewater temperatures were generally less than 10 °C during this phase, which may have significantly decreased biomass growth rates and the SRT was infinite at no sludge wastage conditions essentially making the growth rates extremely low approaching biomass decay rates. The ranges of F/M ratio, organic loading rate and specific substrate utilization rate calculated for no sludge wastage conditions were 0.04 to 0.12 kg BOD_5/kg MLSS-day, 0.36 to 0.63 kg BOD_5/m^3-day and 0.07 to 0.12 mg BOD_5 utilized/mg MLSS-day, respectively. For the 20-day SRT condition, such ranges were 0.04 to 0.30 kg BOD_5/kg MLSS-day, 0.25 to 0.88 kg BOD_5/m^3-day and 0.04 to 0.30 mg BOD_5 utilized/mg MLSS-day, respectively. Since MLSS concentrations dropped after starting sludge wastage F/M values slightly increased in the second phase of operation. F/M values in conventional activated sludge processes typically range between 0.1 and 1 kg BOD_5/kg MLSS-day, higher than those in MBRs due to much lower MLSS levels. Both of the tested SRT conditions provided similar degree of high treatment performance (Figures 20.1 to 20.4). Similarly, increasing the permeate flux to 36 to 39 L/m^2-h (HRT: seven hours) did not deteriorate the permeate quality, except for NH_3-N removal during the initial period. Carbon removal and nitrification were found to be independent of MBR DO concentrations, which mostly ranged between 0.5 and 4 mg/L. During the start-up, when water temperatures were very low, resulting in higher DO saturation concentrations and slower bioactivity, DO concentrations reached a maximum of 12 mg/L. Air flow rate was reduced to minimum (about 14 L/minute) during this period, an amount sufficient for keeping the biomass suspended. However, after three weeks from the start-up, DO concentrations dropped to less than 5 mg/L level. DO

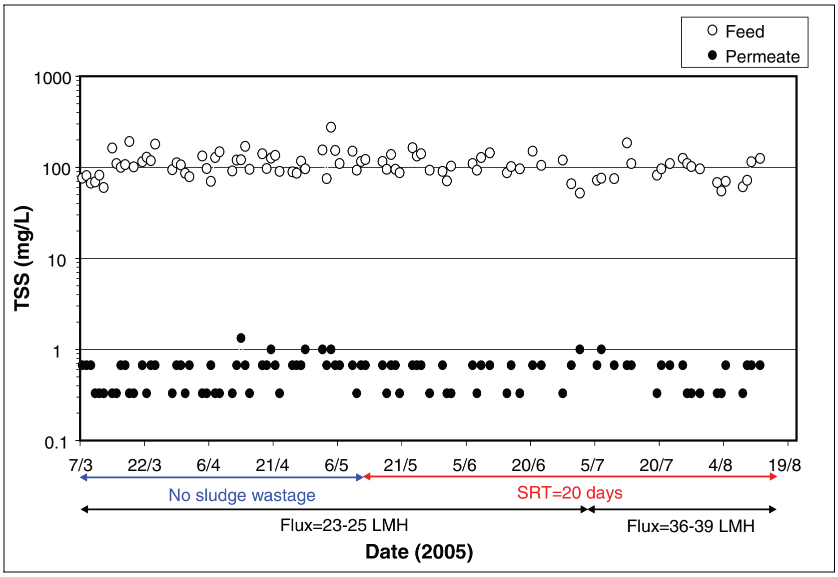

Figure 20.3 Total suspended solid removal throughout membrane bioreactor operation

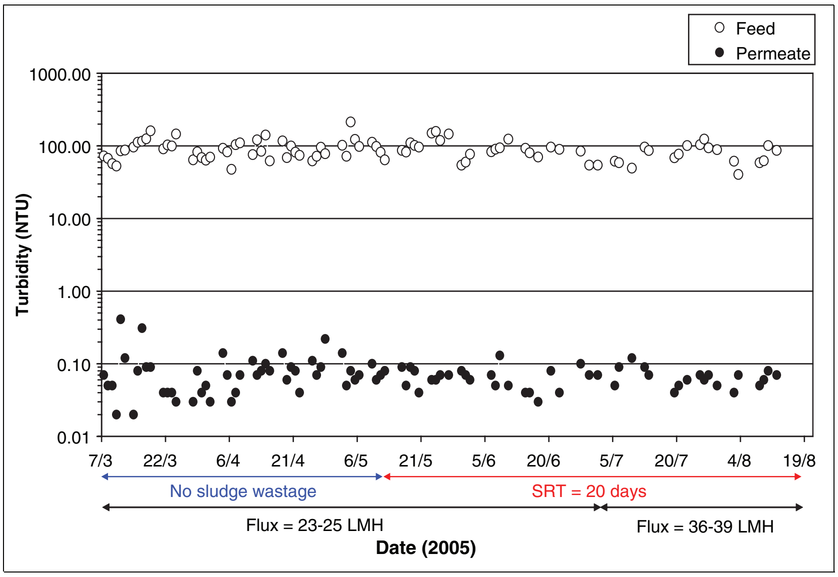

Figure 20.4 Turbidity removal throughout membrane bioreactor operation

concentrations were continuously monitored and air flow rate was adjusted based on a target DO range of 1 to 3 mg/L for bio-oxidation and air requirement for membrane scour and biomass suspension.

Figure 20.3 shows TSS removal during MBR operation. Permeate TSS concentrations less than 1 mg/L were consistently obtained independent of operational conditions including peak loadings, MLSS, SRT and operating flux. Similarly, permeate turbidity values were generally less than 0.1 NTU, with an average of 0.08 NTU (Figure 20.4). These results indicate that in addition to high performance of organic carbon removal and nitrification excellent removal of particulate matter was also achieved by the MBR independent of operational conditions. Furthermore, these low TSS and turbidity levels in permeate suggest the integrity of hollow fiber membranes during the eight-month operation. Figure 20.5 shows the fecal coliform removals. As expected from ultrafiltration membranes, more than 6-log removals were found for total coliform, fecal coliform and total organisms, independent of operational conditions, further proving the integrity of the used membranes. However, if the MBR effluent is to be reused for irrigation further disinfection such as by chlorine or UV is required to obtain non-detectable fecal coliform levels.

Figure 20.6 shows the instantaneous, net and temperature-corrected (based on 20 °C) instantaneous permeate flux values during all operation. Net flux values were calculated from measured instantaneous flux values considering the permeate production loss due to 15 seconds of backwashing at each 10 minutes. Temperature-corrected instantaneous flux values were calculated using the below empirical equation provided by the membrane manufacturer, Zenon Environmental Inc.

$$J_{tm}(20°C) = \frac{Q_p . e^{-0.0239(T-20)}}{S} \tag{20.1}$$

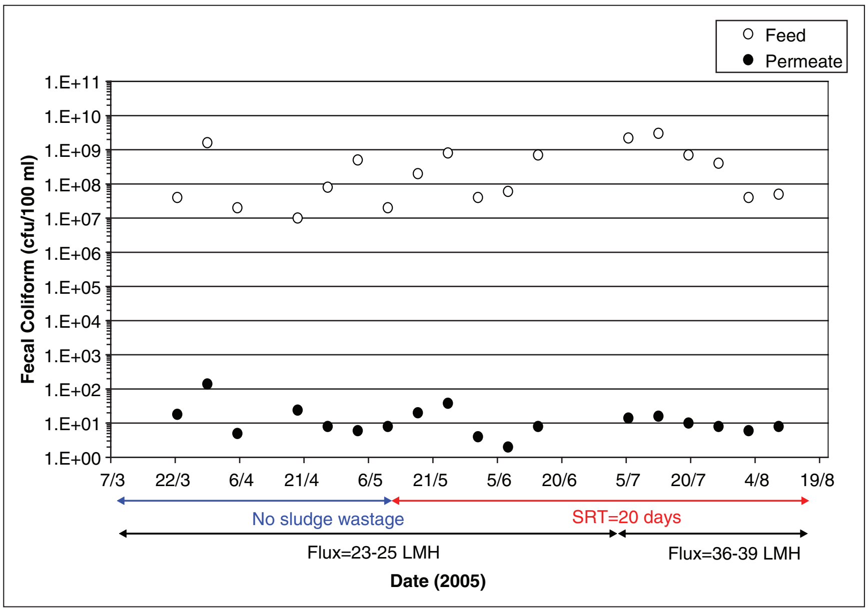

Figure 20.5 Fecal coliform removal throughout membrane bioreactor operation

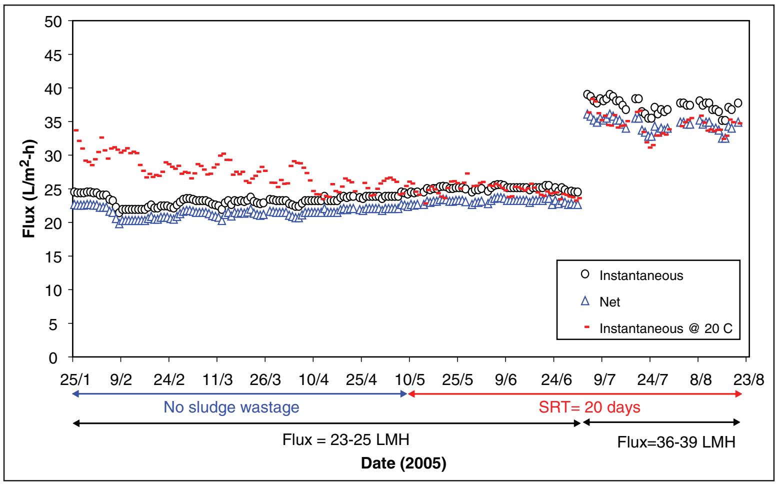

Figure 20.6 Permeate flux throughout membrane bioreactor operation

where,

J_{tm} = instantaneous specific flux at 20 °C (gallon/ft^2-d)
Q_p = measured instantaneous permeate flow rate (gallon/d)
T = temperature during instantaneous flux measurement (°C)
S = total active membrane surface area (ft^2).

Target instantaneous permeate flux values were 25 and 39 L/m^2-h for normal and high flux conditions, respectively. Permeate pump speed was set constant at each condition. The measured flux values were 23 to 25 and 36 to 39 L/m^2-h during the normal and high flux conditions, respectively, showing that target flux values were achieved with small variation. As expected, temperature-corrected flux values were higher than those of instantaneous flux values in winter months due to lower water temperatures and higher viscosity. TMP values measured at normal and high flux conditions were 90 to 172 millibar (1.3 to 2.5 psi) and 170 to 317 millibar (2.5 to 4.6 psi), respectively. All these results indicated that irreversible fouling of membranes did not occur during the eight months of operation even at high flux condition, which is unusual compared to full-scale plants. Furthermore, no chemical cleaning was performed during the six months of operation at normal flux. Only routine back-pulsing with permeate were employed. For the high flux operation continued for approximately two months, chemical maintenance cleaning was only employed two times when the measured TMP reached 4.5 psi. A maximum operation TMP of 8 to 9 psi is recommended by the manufacturer. Although these values were not experienced even at high flux conditions, chemical cleaning was performed to stop the increasing trend of TMP and thus prevent any potential damage to the membrane fibers. Chemical cleaning consisted of back-pulsing with chlorine dosed (250 mg/L) permeate for five minutes without draining the MBR tank. No further intensive chemical cleaning was necessary even at high flux conditions. However, the MBR was continuously aerated (14 to 56 L/minute) to scour membrane fibers, prevent extensive deposition of particles on the membrane fiber surface and provide enough DO (1 to 3 mg/L) for bio-oxidation.

20.4. Conclusions

The characteristics of raw wastewater exhibited both diurnal and seasonal variations during the MBR operation. Although the influent characteristics were highly variable including sudden increases in TDS (e.g., 8300 µs/cm) and the wastewater temperatures were as low as 6 °C during winter months, the MBR system performed well throughout its eight months of operation. The treatment performance was not negatively influenced by variations in operational parameters including F/M ratio, MLSS and DO concentrations, temperature, organic loading rate and specific substrate removal rate. Very high quality effluent obtained in all operational phases indicated that the performance was independent of tested SRT (20 days and no sludge wastage), HRT (7 to 11 hours), MLSS (2,600 to 7,100 mg/L) and permeate flux (25 to 39 L/m^2-h) values. Sustainable nitrification (except for the initial part of the third phase in which HRT was suddenly reduced to seven hours) and organic carbon removals were achieved even during periods with extreme conditions such as very low wastewater temperatures or peak loadings. This sustainable and successful performance of the biological activity demonstrated the presence of specialized biomass in the reactor that could respond to sudden variations. Biomass is retained in MBRs by micro- or ultrafiltration, resulting in high MLSS concentrations and a community structure with a wide spectrum of degradation capability. In addition to the advantages of membrane filtration, this is another advantage of MBRs over conventional activated sludge processes from a biology perspective. The average values of some parameters in permeate samples for all operation period were as following: COD, 9.5 mg/L; BOD_5, 1.5 mg/L; TSS, 0.6 mg/L; NH_3-N, 1.0 mg/L; turbidity, 0.08 NTU; fecal coliform, 19 cfu/100 mL; total coliform, 160 cfu/100 mL. Turbidity values less than 0.1 NTU and more than 6-log coliform removals consistently achieved in permeate indicated the integrity of membranes throughout the operation.

Irreversible fouling of membranes did not occur during the eight months of operation based on TMP, flux and permeability measurements. No chemical cleaning was performed during the six months of operation at normal flux, except routine back-pulsing with permeate. For the high flux operation, chemical maintenance cleaning was only employed two times, which consisted of back-pulsing membranes with chlorine dosed (250 mg/L) permeate for five minutes without draining the MBR tank. No further intensive chemical cleaning was necessary even at high flux conditions. Overall, eight months of pilot-scale tests indicated the robustness of MBR process in terms of achieving very high quality of treated water without any operational limitations including fouling and permeability reduction problems.

Such performance of the MBR pilot system treating a highly variable campus wastewater will guide the decision on a potential full-scale MBR application and reuse of the treated wastewater for irrigation in the campus area.

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References

Adham S and Trussell R. (2001). Membrane Bioreactors: Feasibility and Use in Water Reclamation. San Diego, CA, Final Report, Water Environment Research Foundation, Alexandria, VA.

Adham S, Gagliardo P, Boulos L, Oppenheimer J, Trussell R. (2001). Feasibility of the membrane bioreactor process for water reclamation. Water Sci Tech 43:203–209.

Chang IS, Le Clech P, Jefferson B, Judd SJ. (2002). Membrane fouling in membrane bioreactors for wastewater treatment. J. Environ. Eng-ASCE 128:1018–1029.

Cicek N, Franco JP, Suidan MT, Urbain V. (1998). Using a membrane bioreactor to reclaim wastewater. J Am Water Works Assoc 90:105–113.

Daigger GT, Rittmann BE, Adham S, Andreottola G. (2005). Are membrane bioreactors ready for widespread application? Environ Sci Technol 39:399A–406A.

Fane A and Chang S. (2002). Membrane bioreactors: design and operational options. Filtr Sep 39:26–29.

Gander M, Jefferson B, Judd S. (2000). Aerobic MBRs for domestic wastewater treatment: a review with cost considerations. Sep Purif Technol 18:119–130.

Gunder B. (2001). The Membrane-Coupled Activated Sludge Process in Municipal Wastewater Treatment. Technomic Publishing, Lancaster, PA.

Judd SJ. (2004). A review of fouling of membrane bioreactors in sewage treatment. Water Sci Tech 49:229–235.

Kraume M, Bracklow U, Vocks M, Drews A. (2005). Nutrients removal in MBRs for municipal wastewater treatment. Water Sci Tech 51:391–402.

Lesjean B, Rosenberger S, Schrotter JC, Recherche A. (2004). Membrane-aided biological wastewater treatment- an overview of applied systems. Membr Technol August:5–10.

Stephenson T, Judd SJ, Brindle K. (2000). Membrane Bioreactors for Wastewater Treatment. IWA Publishing, London.

Van der Roest HF, Lawrence DP, van Bentem AGN. (2002). Membrane Bioreactors for Municipal Wastewater Treatment. IWA Publishing, London.

Visvanathan C, Ben Aim R, Parameswaran K. (2000). Membrane separation bioreactors for wastewater treatment. Crit Rev Environ Sci Technol 30:1–48.

Yang W, Cicek N, Ilg J. (2006). State-of-the-art of membrane bioreactors: worldwide research and commercial applications in North America. J Membr Sci 270:201–211.

Chapter 21
Socioeconomic Aspects of Wastewater Treatment and Water Reuse

Bahman Sheikh

Abstract Wastewater treatment is an essential prerequisite for water reclamation and reuse. Proper treatment and disinfection of wastewater is also a public health necessity in human communities. Treatment and distribution of recycled water involves great expenditure of resources, which in many developing countries is either lacking or is devoted to more urgent national priorities. Also, an appropriate valuation of water and its benefits to society is often lacking due to a misperception of abundance and taking water for granted—a gift of nature, to be used at will. This attitude must be changed with proper educational tools if the relatively constant amounts of water now available are to be sufficient for increasing populations of the future.

Water reuse projects face additional impediments. One major impediment is that the agreement of two or more governmental entities is required before a project can be implemented. These institutional barriers are not insurmountable, but they involve lengthy negotiations and much give-and-take on the part of the involved entities. Ideally, a single entity would be managing all matters related to the entire water cycle, but this is rare. Pricing of recycled water is another issue complicating the ability of water managers to pay the costs of implementing water-recycling projects. Recycled water is often priced significantly below the price of potable water.

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Public attitude toward the reuse of reclaimed water for non-potable applications is generally positive. However, there have been several instances in California and Australia where resistance to indirect potable reuse has derailed a few otherwise excellent projects. Fortunately, public outreach and educational programs have been devised by professionals in the field for early public involvement and prevention of dissemination of misinformation by project opponents. The effects of global warming on future water supplies is not expected to be uniform everywhere, but it will be drastically limiting in certain parts of the arid and semi-arid regions of the world. Already, the continent of Australia is experiencing a 10-year drought, attributed to global warming. This increases the urgency for development of water use efficiency measures, such as water recycling in these regions.

21.1. Introduction

Protection of the public health is the prime reason for providing a supplemental source of water supply—recycled water. The entire process of treatment, distribution and final use of recycled water must, therefore, be also highly protective of the public health.

Until the latter part of the 20th century, decisions about public water supply resources were made strictly by engineers and other experts. There was virtually no consultation with the public regarding their preferences, the impacts of such decisions, or the relative merits of various resources considered for development. In recent years, the public has become involved increasingly with water supply decisions and planning for future water resources for the community. The greater involvement of the public in such decisions—at least in the industrialized nations of the world—has exerted an immense impacts on the nature of the decisions, length of time needed to implement a water project, and especially on the use of water reclaimed from wastewater for reuse.

The socioeconomic impacts of wastewater treatment and reuse have always been an integral part of wastewater management decisions. In the past, the impacts became felt long after it was too late to go back and make a better decision. Only in recent years have these impacts come to the forefront of decision making and selecting options for the future.

21.2. Recognizing the Full Value of Water

At the heart of the public involvement in water matters is an awakening awareness of the real value of water on the part of the general public. While this awareness is not widespread yet, it is spreading. The fact that middle- and lower-class individuals now routinely pay for bottled water more than 1,000 times more than the cost of tap water is an indication of their willingness to pay for the higher-quality drinking water. This is despite of the fact that municipal tap water is, in most cases, comparable to any bottled water on the market in microbial and chemical quality. To convince the same public to pay a fair price for municipally provided tap water requires a herculean effort involving service improvement, reliability of continuous water supply at the proper pressure, assurance of high quality and safety of the served water, and trust in the system providing public water service. Obviously, this cannot be accomplished overnight. But, without a conscious effort, it is hard to imagine that it would ever be accomplished on its own accord.

In many developing countries, there is a strong and widespread reluctance to pay for water service, even among those who can afford the relatively low costs involved. Collection of fees for wastewater treatment and proper disposal or reuse is even more difficult, because the service is often not perceived as necessary, beneficial, or resulting in immediate satisfaction of any desire on the part of the rate payers. This results in a vicious cycle of poor service, dissatisfaction with the poor service and greater lack of desire to pay for that poor service. This cycle of perpetuating degradation of service is graphically illustrated in Figure 21.1. It explains why so many communities lack a safe potable water service and a proper wastewater management system. The cycle underlies the difficulty of reclaiming the wastewater and treating it to a high enough level that would make it suitable for a variety of beneficial reuse.

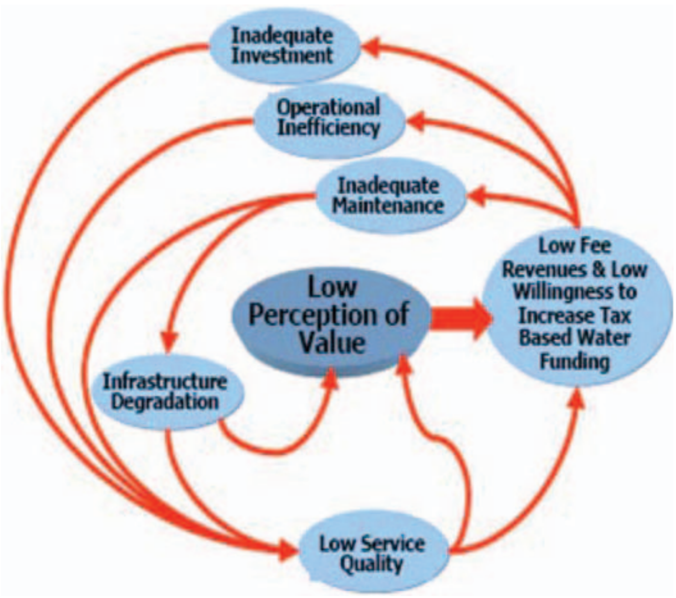

Figure 21.1 The vicious spiral of low funding. Source: Pacific Institute, Oakland, California, USA

21.3. Public Perceptions of Water Reuse

The public generally does not think much—if at all—about the origins of its water supply; however, there is an assumption of purity and virginity associated with potable (drinking) water that defies reality. The reality is that all water is recycled and there is virtually no "new" water created and no naturally occurring water is "pure." With human and animals interacting with water, with urbanization and industrialization, nearly all raw water supplies are contaminated biologically and chemically. This information, plus the capability of technology to take contaminants out of any source of water and return it to a safe and pure status must be communicated to a community on the verge of adopting expensive wastewater treatment and water reuse—as shown in Figure 21.2, adapted from Asano et al. (2006). Otherwise, the public's reluctance to pay, their aversion to recycling of water by technological means and the mistrust of utilities to use public money honestly and efficiently will prevent realization of the water cycle projects.

A graphic depiction of changes of water quality with treatment stages is presented in Figure 21.2. This depiction is primarily aimed at biological quality of the water, but it can also be used to convey changes in the microconstituents in water and wastewater as it undergoes use, reuse, treatment and further treatment.

Over the coming years, giant strides will be taken by WateReuse Association and its research arm, the WateReuse Foundation, in solving the difficult problem of overcoming public apprehensions about use of recycled water. Already, several

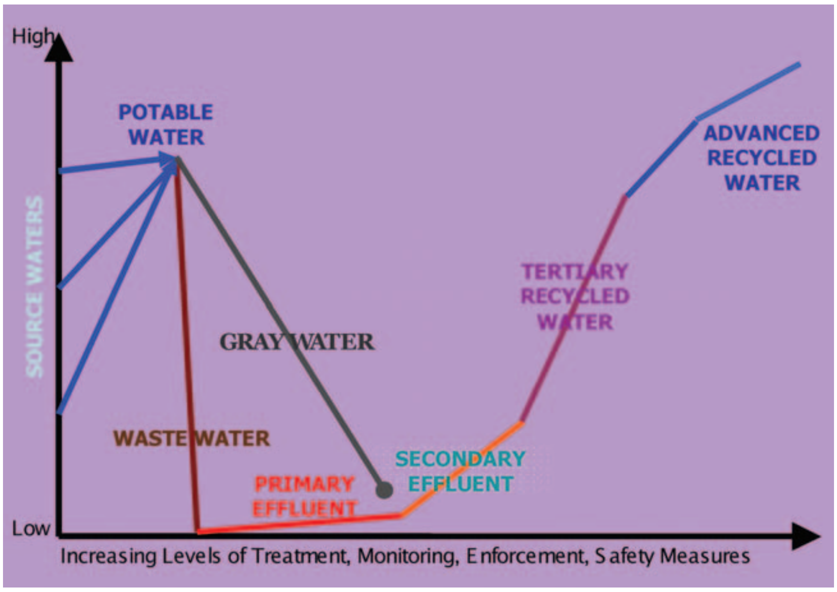

Figure 21.2 Change in water quality with use and treatment levels. Adapted from Asano et al., 2006

guidebooks and special projects have been completed (WEF/AWWA, 1998; Humphreys, 2006; Reutten, 2004). Research projects to further delineate public response to wastewater treatment and water reuse have been funded and are "in the pipeline." These projects will bear fruit over the coming decades and result in a much more systematic approach to public education and outreach during the planning process for water recycling projects.

21.4. Pricing Recycled Water

Recycled water is generally priced below potable water to provide a significant incentive for customers to switch from potable to recycled water. The extent of discount ranges from 15% to as high as 100%—giving the recycled water free of charge to the users, in cases where the agency desperately needs a reliable disposal venue. This requirement for discounting the price of reclaimed water places one more impediment on implementation of new water reuse projects: inability (or difficulty) to repay the costs of project implementation, operation and maintenance. Because of the significant benefits of water reuse to society and the environment, incentive subsidies are often provided by regional and central governments to local governments to overcome these fiscal imbalances. This is a means by which the larger society pays for the benefits derived from every water reuse project implemented by local governmental agencies.

The pricing schemes for potable water are generally below the actual cost of operation and maintenance—not to mention the capital costs of constructing the treatment and distribution systems—of the supply systems in many communities. Thus, the even lower price of reclaimed water cannot be expected to cover the normal costs of operating a water reuse system. Subsidies, incentive payments and transfers from other funds are commonly used to defray the gap between cost and revenue of water reuse systems.

21.5. Interagency Collaboration and Agreements

Several agencies with different authorities and responsibilities must come together and form an agreement before a water reuse project can be initiated in a community.

At the national level, the departments of agriculture, public health, finance, water resources and the environment are involved and each have a jurisdictional authority over the project. At the local level, the agreement must be reached among water wholesalers, retailers, wastewater managers and public works officials. These multi-agency agreements are difficult at best and impossible at worst. A strong motivation for the agencies to agree to cooperate must come from one of the following sources of pressure:

- water scarcity and a public health crisis situation arising from deficiency of water
- environmental impact of discharge of treated wastewater, loss of threatened and endangered species of plants and/or animals

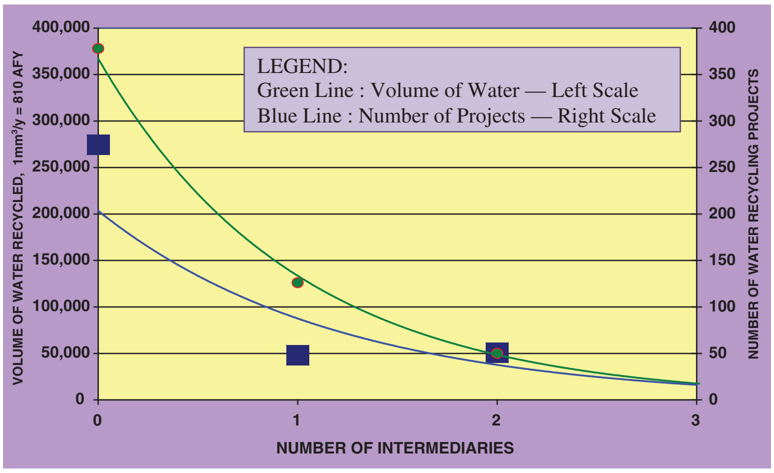

Figure 21.3 Relationship between number of intermediaries and number of successfully implemented projects and the volume of recycled water generated

- need for increasing the reliability of water supply and independence from imported sources of water

Each of these motivators or a combination of several of them can be strong enough to move the diverse parties to the negotiating table and eventually make it possible to start a water recycling program. Another possible motivation is the anticipated effects of global warming on water supplies—discussed further in Section 21.6, below.

The greater the number of agencies involved in decision-making regarding a new water reuse project, the longer it takes to accomplish the necessary tasks, reach an agreement and complete the project. In a recent study of more than 200 California water-recycling projects, an inverse relationship was discovered between the number of "intermediaries" (between the producer and the final customer of recycled water) and the volume of water reclaimed and reused (Sheikh et al., 2004). A similar relationship was also found between the number of intermediaries and the number of successful projects in operation. These relationships are graphically depicted in Figure 21.3.

21.6. Global Warming

Global warming[1] as affected by human activity since the onset of the industrial revolution is accepted as a phenomenon of grave concern by climate scientists. The phenomenon is questioned by some with a vested interest in the short-term economic

[1] Climate change is a euphemistic reference to the same phenomenon for those uncomfortable with the massive evidence for global warming or for the role of humans in the accelerated rate of warming of ocean waters, atmosphere, and the gradual melting and disappearance of continental glaciers.

and political consequences of decisions that may lead to a reversal of recent (last two centuries) trends in the concentration of carbon dioxide in the earth's atmosphere. However, decisions to reduce carbon emissions are necessary if planet earth is to remain hospitable to human existence for the foreseeable future.

One of the critical (and relatively immediate) effects of global warming is expected to be a major elevation of temperatures throughout the world, resulting in higher evapotranspiration rates, directly leading to increased demand for irrigation water. Irrigation of agricultural lands accounts for more than 75% of the demand for water in countries in the arid and semi-arid regions of the world. A global-warming induced increase in demand for water—not balanced by an equal increase in supply—would be terribly stressful on available sources of water supply and will likely increase the motivation for implementation of water conservation and recycling programs. After all, recycled water is a local source of water, most readily available to the urban sector, at least for non-potable applications. Potable applications are on the horizon; Namibia, Singapore and Australia are leading the way.

21.7. The Need for Storage of Recycled Water

Recycled water is generated at a relatively steady pace and does not require as much storage as precipitation occurring in only a few months of the year. If used for agricultural irrigation, recycled water receives less treatment than if used for indirect potable use. However, it will need large storage volumes to balance seasonal demand against year-round supply. Figure 21.4 shows a groundwater recharge basin and pump station for recycled water from Tel Aviv, Israel. The aquifer-stored water is pumped from wells and sent to Negev Desert (50 km away) for unrestricted irrigation of a variety of crops. Israel recycles more than 70% of its wastewater in planned and managed projects. Unplanned reuse of wastewater and wastewater effluent (by way of withdrawals from local streams dominated with upstream wastewater discharges) is common and sometimes approaches 100% of the discharge in some countries. However, this practice is not protective of the public health and is not considered a sustainable water reuse policy.

If highly treated recycled water is used for direct potable purposes—rather than for landscape irrigation—the demand will be matched by supply and there will be no need for seasonal storage. Indirect potable reuse, on the other hand, involves augmentation of surface reservoirs or groundwater aquifers. Thus, storage capacity is needed to provide the necessary mixing and detention time.

21.8. Sustainable Water Resources Planning

Most water resource planners now recognize the importance of integrated water resources planning, taking into account a variety of sources of water, rather than just one or two traditional sources. It is no longer unusual for an agency to have

Figure 21.4 Groundwater recharge of reclaimed water for treatment, storage and agricultural irrigation in Israel

access to a number of well-defined sources of water supplies from one or more of the following categories:

- imported water from far away
- imported water from a neighboring watershed, transfers and exchanges
- local surface runoff, stored in surface storage reservoirs
- local groundwater resources used within a safe-yield management plan
- local groundwater resources used to extinction—not a sustainable resource, but sometimes an inevitable short-term or interim solution
- water conservation through a variety of water use efficiency measures
- water reclamation, recycling and reuse
- desalination of brackish water or seawater

Each one of these water supply sources can play a vital role in a given community's integrated water resources plan, managed with full recognition of its limitations, water quality, environmental impact of its utilization and other services and benefits related to each resource. Bringing recycled water into the supply mix initially may represent a small fraction of the total water demand, or it may be a significant portion of the supply, depending on local circumstances. Either way, its presence in the overall mix will improve dependability of supply of water and will release much needed potable supply from the irrigation sector.

An important attribute of recycled water is that its availability and quantity increases with increasing population—quite the opposite of the trend for other

Example of landscape irrigated with recycled water in San Jose, California, USA

resource, which are either diminishing in quantity, or quality, or both, as a direct function of population increase. Therefore, provisions for inclusion of this sustainable source of water in the community's integrated water resources plan is a wise and visionary step.

References

Asano T, Burton F L, Leverenz H L, Tsuchihashi R., and Tchobanoglous G. (2006). "Water Issues: Current Status and the Role of Water Reclamation and Reuse". McGraw Hill, New York, p. 29.

Humphreys L. (2006). Marketing Nonpotable Recycled Water: A Guidebook for Successful Public Outreach and Customer Marketing. WateReuse Foundation, Alexandria, VA.

Reutten J. (2004). Best Practices for Developing Indirect Potable Reuse Projects: Phase 1 Report. WateReuse Foundation, Alexandria, VA.

Sheikh B, Rosenblum E, York D., and Hartling E. (2004). Institutional requirements in California and Florida for implementation of water recycling/reclamation projects. Presented at WateReuse Symposium XIX, Phoenix, AZ.

Water Environment Federation and American Water Works Association (WEF/AWWA). (1998). Public information outreach programs. In: Using Reclaimed Water to Augment Potable Water Resources, Alexandria, Virginia, Denver, Colorado. Water Environment Federation, Denver, CO, USA.

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Chapter 22
Cost–Benefit Analysis for Centralized and Decentralized Wastewater Treatment System (Case Study in Surabaya-Indonesia)

Maria Prihandrijanti(✉), Almy Malisie, and Ralf Otterpohl

Abstract Around 3 million inhabitants of Surabaya, the second biggest city in Indonesia, still dispose their wastewater to water bodies. This is mainly caused by lack of adequate wastewater treatment system. However, large-scale centralized wastewater treatment is not an economical option particularly for people living in low-income urban areas. Decentralized wastewater treatment systems that are more affordable are being developed.

To have a sustainable wastewater treatment system, an integrated assessment of each alternative based on its economical, environmental, social, health and institutional aspects is necessary. This study explores the economical aspects of three scenarios of wastewater treatment system, with Kalirungkut subdistrict, a densely populated urban area in Surabaya, as a case study area. The costs and benefits of alternative interventions are evaluated using the cost–benefit analysis (CBA) method to support the decision making process by bringing elements of transparency and objectives.

The results of CBA in this study showed that the decentralized system was more feasible economically for this case study, since the centralized wastewater treatment

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system had the highest net present value cost and the lowest cost–benefit ratio (C/B ratio). To support decision making regarding the sustainable wastewater treatment system for this area, further assessment on environmental, health, social and institutional aspects are recommended.

22.1. Introduction

Almost 3 million inhabitants of Surabaya, the second largest city in Indonesia, still dispose their wastewater to water bodies. This is mainly caused by the lack of adequate wastewater treatment system. To have a sustainable wastewater treatment system, an integrated assessment of each alternative based on its economical, environmental, social, health and institutional aspects is necessary. This study explores the economical aspects of three scenarios of wastewater treatment system, with Kalirungkut subdistrict, a densely populated urban area in Surabaya, as a case study area. The costs and benefits of alternative interventions are evaluated using the cost–benefit analysis (CBA) method in order to support the decision maker by bringing elements of transparency and objectivity.

As Surabaya does not have a functioning sewerage system and its treatment, the existing system is mainly pour-flush toilet with septic tank. This system requires that settled sludge in the septic tanks is periodically desludged with a septage hauler truck and then transported to the septage treatment plant. Septage is collected from septic tanks all around Surabaya by fleets of tankers operated predominantly by private companies. Unfortunately, not all tankers discharge at the sewage treatment works. There are also direct discharge points along the Surabaya River, which provide no treatment to the septage at all. The river water in the vicinity of the outfalls is predominated by the septage.

Like the general sanitation condition in Surabaya, there is currently insufficient sanitation infrastructure in the case study area (Kalirungkut subdistrict) and no sewerage networks or sewage treatment for domestic wastewater, except the onsite systems. The range of options available for improving access to sanitation is wide, especially in low-income settings where large portions of the population have access to only the most basic facilities. For developing countries, World Health Organization (WHO) favors intervention options that are low cost, feasible and do not require heavy maintenance.

22.2. Centralized Wastewater Treatment System

Surabaya Sewerage and Sanitation Development Programme (SSDP) has done a study on the sewage treatment works in Surabaya (PT Indulexco Consulting Group, Mott MacDonald Ltd., PT Dacrea, 1997). Surabaya has one sewage treatment work, which is located in Keputih (Sukolilo district, east Surabaya). It is a public work operated by Dinas Kebersihan (municipal cleaning department), treating septage from household septic tanks from all areas in Surabaya. The overall appearance of the work was poor due to badly maintained inlet works and the casual use of any spare land for sludge drying. The mechanical and electrical plant was working and the oxidation ditches were apparently operating, although failures and faults exist. The works were probably suffering from overloading and so could not be expected to meet the design quality standards.

In 1996, a master plan for sewerage and sanitation development was written for the city of Surabaya. In this master plan for the year of 2020, the wastewater would be treated in off-site modules by using shallow sewerage as conveyance system and Imhoff Tanks as the treatment technology. This master plan has not been in implementation yet due to some constraints and lack of funds. In addition, large-scale centralized wastewater treatment is not an economical option particularly for people living in low-income urban areas. Decentralized wastewater treatment systems (DEWATS) that are more affordable are being developed. To solve water pollution from domestic wastewater, decentralized wastewater systems have been constructed in some areas.

22.3. DEWATS

The DEWATS-Indonesia project is publicly funded private cooperation between the German and Indonesian nonprofit organizations: Bremen Overseas Research and Development Association (BORDA), Institute for Rural Technology Development (*Lembaga Pengembangan Teknologi Pedesaan*) and Bina Ekonomi Sosial Terpadu. DEWATS is based on the principle of low-maintenance since most important parts of the system work without technical energy inputs and cannot be switched off intentionally. DEWATS applications provide state-of-the-art technology at affordable prices because all of the materials used for construction are locally available. DEWATS applications are based on basic technical treatment modules which consist of baffled upstream anaerobic reactors for grey water treatment and anaerobic digester (AD) for blackwater treatment (Figure 22.1).

Another decentralized wastewater treatment alternative that this study considers is Ecological Sanitation (EcoSan) that follows similar ideas as DEWATS. In addition, EcoSan aims to full reuse of nutrients. Another EcoSan principle is to prevent mixing of pathogenic bacteria from human waste with the wastewater that are going to be returned to the environment. Furthermore, EcoSan is able to recover valuable nutrients from domestic wastewater, particularly in human urine and fecal matter. These nutrients would not be recoverable if they are diluted with large amounts of domestic wastewater in the conventional sewerage systems.

The EcoSan pilot plant in the case study area separates wastewater into blackwater (human fecal plus flushing water), yellow water (human urine plus flushing water) and grey water. Blackwater is contained and treated separately into compost using worms (vermicomposting), while yellow water is stored separately for six months to have a hygienization process. Grey water, which originates from bathing

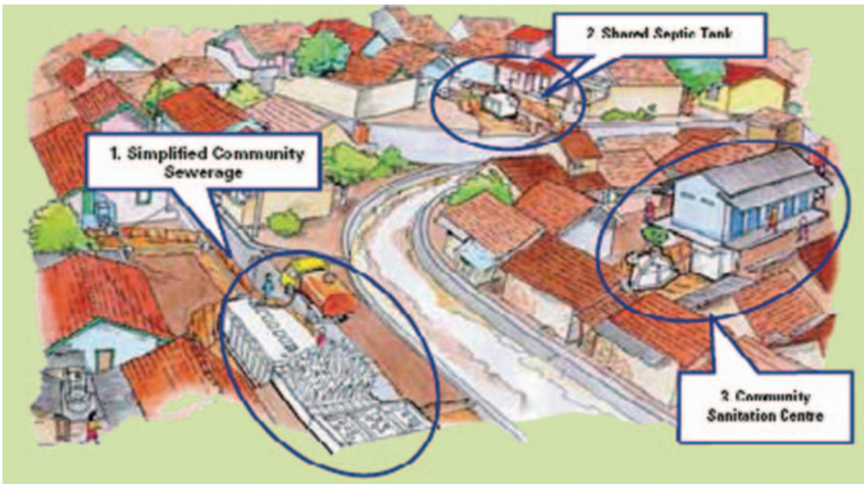

Figure 22.1 Examples of decentralized wastewater treatment system

and washing activities, contains less organic matter and is treated with horizontal flow subsurface constructed wetland.

22.4. Cost–Benefit Analysis (CBA)

According to the 2001 Census, kecamatan (district) Rungkut has a population of 111,286 people, who live in six kelurahans. It has an area of 21.02 km^2, thus the population density of this district is 5,294 people/km^2. The Surabaya Industrial Estate Rungkut (SIER) is located in this district. This industrial estate houses 371 companies and has an area of 332 hectare. Kelurahan (subdistrict) Kalirungkut, with a population of 35,090 people and an area of 2.58 km^2, has the highest population, because a big university and several industries are located in this subdistrict. Therefore, in this subdistrict there are a high percentage of non-permanent residents, who came here from other towns or villages to work, study or live there.

Considering the location profile (e.g., population, density, etc.), it would be better and easier to divide the wastewater management scheme into smaller modules to make it easier to manage and minimize the need for expert operation and maintenance (O&M) (Figure 22.2). With smaller modules, the O&M of the facilities would not be so complex as bigger plants and the community or someone from the community who acts or paid as a caretaker could be trained to manage the facilities. As the smallest neighborhood coordinating organization, Rukun Tetangga (RT) would be very suitable to facilitate community-based sanitation management, because it takes care of the social and administrative matters of a neighborhood. The base scenario is based on

Figure 22.2 Community toilets (existing) in case-study area

the empirical study, which has been done in Kelurahan (subdistrict) Kalirungkut, Kecamatan (district) Rungkut in Surabaya-Indonesia.

The case study area, Kalirungkut subdistrict, has 35,090 population (in 2001), which is divided into 83 RTs. With an assumed average growth rate of 1.57%/year, it is projected that the population would be 50,000 people/RT in 2020. This study is based on implementation of two alternative sanitation systems in comparison to conventional wastewater treatment (household connections to the sewerage system and partly-centralized wastewater treatment of domestic wastewater). Thus, there were three different alternatives for this study, which are outline in Sections 22.4.1. to 22.4.3.

22.4.1. Scenario 1

Ecological Sanitation (EcoSan) system alternatives integrated with DEWATS technologies to render domestic wastewater safe and use the safe products of sanitized human excreta for agricultural purposes, as follows:

a) Communal toilet for a part of the RT's population who does not have their own/private WC (Water Closet) with source separation (yellow, brown and grey water) followed by urine (yellow water) storage in urine tank, treatment of fecal material (brown water) with pre- and vermicomposting in Rottebehaelter and treatment of grey water with horizontal subsurface flow constructed wetland (HSFCW) or baffled septic tank (BST) (Figure 22.4).
b) Decentralized domestic wastewater treatment for the rest of the RT's population who have their own WC with BST for grey water treatment. The brown water and urine are treated/stored in every household. Thus, only the grey water from every household are sent to the decentralized treatment unit. For the decentralized alternative, BST is preferred instead of HSFCW because HSFCW would require very large area for the whole population served.

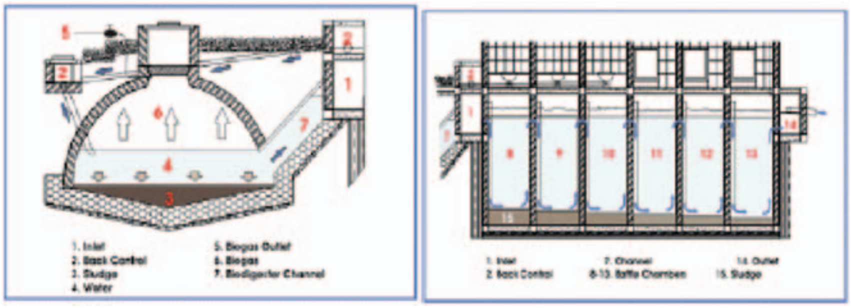

Figure 22.3 (A) Anaerobic digester. (B) Baffled septic tank

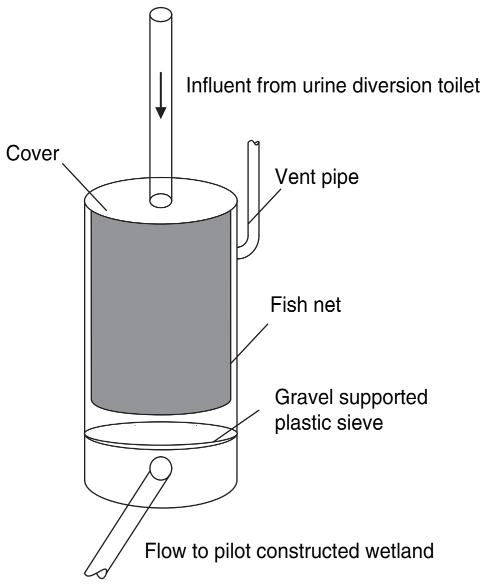

Figure 22.4 Scheme of Rottebehaelter

22.4.2. Scenario 2

DEWATS technologies to render domestic wastewater safe and use of the biogas product for cooking purposes, as follows:

a) Communal toilet for a part of the RT's population who does not have own/private WC with partly separated treatment of grey- and blackwater (wastewater from WC). The grey water is treated with BST and the blackwater is treated with AD onsite.
b) Decentralized wastewater treatment for the rest of the RT's populations who have their own WC with BST for grey water treatment and AD for blackwater treatment (Figure 22.3).

22.4.3. Scenario 3

Household connections to the sewerage system and off-site wastewater treatment of domestic wastewater from the community according to the Surabaya SSDP (Sewerage and Sanitation Development Programme) Master Plan 2020. Based on this Master Plan, the wastewater in this scenario would be treated in off-site module by using shallow sewerage as conveyance system and Imhoff Tank as the treatment technology.

Shallow sewer was chosen because it can be laid to flatter gradients and lower depths than conventional sewers and placed in non-trafficked areas. This rendered it suitable for small community systems in low-income Kampung-type areas. Shallow sewer could be utilized in conjunction with a conventional system. Small low-income areas could be connected to a conventional sewer in a nearby street by a shallow sewer network. Imhoff-tank works similar to communal septic tank. Imhoff tank provides minimum wastewater treatment facilities for household influents. Due to the underground construction, land use is very limited. An Imhoff tank can be constructed under roads or other public areas. Construction costs are higher than septic tank. Manual or vacuum desludging is required more often than septic tank. Reduction of BOD is about 30 to 40%; very moderate reduction of infectious organisms and quite effective sedimentation of coarse particles.

Few studies measured the costs and benefits of alternative interventions to provide policy makers with the information to choose the most efficient intervention from the viewpoint of society or the health sector. Generally, it would seem that there has been inadequate attention to economic issues in water and sanitation interventions (WHO, 2001), but especially in the field of sanitation system alternatives (other than sewerage system).

The purpose of this study was to explore the costs and benefits of the sanitation system alternatives/scenarios, with Kalirungkut-Surabaya as case-study area. This work was not meant to state or declare that an alternative sanitation system is the best or the only option for every low-income urban area in general, because the choice of a best sanitation system for a certain area/community depends on various aspects; so that it might differ very much from one to the other.

The viewpoints of this analysis were:

- economical (building, O&M costs)
- society (direct expenditures avoided due to less illness from diarrheal disease)

This financial CBA will include available information related to:

- an analysis of the costs of constructing each sanitation system alternative for case study area in monetary terms
- an analysis of the benefits obtained from implementing the sanitation system alternatives expressed in monetary terms

However, as in other CBA in water and sanitation field, there are many intangible costs as well as benefits, which are not constituted or represented by a physical object and very difficult to measure, such as the impact (in monetary terms) of different effluent qualities from several wastewater treatment systems, benefits from the reduction of soil infertility through usage of human excreta as fertilizer/soil conditioner, etc. Costs for each scenario have been compiled and presented in Table 22.1.

Due to problems in measurement and quantification/valuation, and also because of substantial variability between settings and absence of adequate data, only the averted health expenditure from the private point of view and revenue from user fee will be valued as benefits in this study. Further, as in the case also in other studies on C/B ratio of sanitation interventions, this analysis also makes use of some general assumptions for all scenarios. The averted health expenditure incorporates:

- treatment and transport costs
- income gained due to days lost from work avoided
- opportunity cost of days of school absenteeism avoided
- productive parents' days lost avoided due to less child illness

The benefits have a general characteristic, which could result from all sanitation interventions. Therefore, they are assumed the same for each scenario. Total annual benefit for each scenario was 658,125€.

The results of CBA were presented in the following tables. For this analysis, the present value of costs and benefits are combined in Table 22.2 to calculate the net present value (NPV) and C/B ratio. The flow of costs and benefits were discounted at 10% discount rate. It can be seen from the table that scenario 1 has a positive NPV and a C/B ratio greater than one, whereas the others have not. Nonetheless, this does not mean that the others are not practicable, because the results above are only based on calculations on a given aspects of costs and benefits. Scenario 2 could also have a positive net benefits if, one of the parameters changes. Scenario 3 is obviously the most expensive option in comparison to EcoSan and DEWATS, especially if the costs of clean water wasted through its utilization as a means of wastewater conveyance are also calculated. Still it is apparent that the EcoSan scenario has more benefits than the other systems, such as the added benefit from urine and feces as fertilizer or soil conditioner, its

Table 22.1 Total investment and operation and maintenance costs of each scenario

System	Construction (€)	Land (€)	First cost (€)	Operation and management (€)
Scenario 1	4.203.528	562.500	4.766.028	68.750
Scenario 2	5.037.007	800.500	5.837.507	68.750
Scenario 3	6.173.838	Included in construction	6.173.838	308.692

Table 22.2 Net present value of the three scenarios.

Scenario	PV Cost (€)	PV Benefits (€)	NPV (€)	B/C ratio
1	4,896.635	5,397.481	500,846	1.10
2	5,870.717	5,397.481	−473,236	0.92
3	8,144.312	5,397.481	−2,746.831	0.66

Table 22.3 Net present values from sensitivity tests result.

Scenario	Base case	r=8%	r=12%	Construction +30%	O&M +30%	User fee +30%	Benefit start delayed 1 yr
1	500.845	1.110.639	17.147	−134.214	331.693	746.884	−97.456
2	−473.236	118.557	−939.577	−1.119.245	−642.388	−227.197	−1.071.538
3	−2.746.831	−2.441.540	−2.979.370	−5.190.120	−3.506.329	−2.500.792	−3.345.133

Table 22.4 Cost–benefit ratio from sensitivity tests result

Scenario	Base case	r=8%	r=12%	Construction +30%	O&M + 30%	User fee +30%	Benefit start delayed 1 yr
1	1.10	1.22	1.00	0.98	1.07	1.15	0.98
2	0.92	1.02	0.84	0.83	0.89	0.96	0.82
3	0.66	0.72	0.62	0.51	0.61	0.69	0.59

closing-the-loop concept and simple technology. Nevertheless, not all benefits could be valued in this study because a wide range of further studies are still needed, such as how to optimize the fertilizer value of human excreta as well as its marketing strategy.

A number of sensitivity tests were carried out to show how changes in certain parameters would affect indicator values. The results for the NPV values and B/C ratio are in Tables 22.3 and 22.4, respectively. The parameter changed are: discount rate 8% and 12%, construction cost 30% higher, O&M cost 30% higher, user fee 30% higher and benefit start delayed by one year.

The results showed that if the discount rate was changed to 8%, C/B ratio of scenario 1 and 2 will be higher, but scenario 3 still has C/B ratio less than scenario 1. If the discount rate was changed to 12%, then only scenario 1, which still has a positive NPV, even though it is very small (C/B ratio = 1.00). If the O&M costs were increased by 30% or the user fee was increased by 30%, only scenario 1, which still has positive NPV. All scenarios will have negative NPV or C/B ratio less than one if the benefit start was delayed for one year and if the construction cost was increased by 30%. Sensitivity tests show that these results are very sensitive especially to changes in construction cost and benefit start.

22.5. Conclusions and Recommendations

The results of CBA in this study showed that the decentralized system was more feasible economically for this case study, since the centralized wastewater treatment system had the highest NPV cost and the lowest C/B ratio. To support decision making regarding the sustainable wastewater treatment system for this area, further assessment on environmental, health, social and institutional aspects are recommended.

References

Hutton G and Haller L. (2004). Evaluation of the Costs and Benefits of Water and Sanitation Improvements at the Global Level. World Health Organization, Geneva, Switzerland.

Indonesian Water Supply and Sanitation Policy Formulation and Action Planning Project. (2002). Sanimas: Sanitasi oleh Masyarakat. Water and Sanitation Program, Jakarta, Indonesia.

PT Indulexco Consulting Group, Mott MacDonald Ltd., PT Dacrea. (1997). Final Report Master Plan Surabaya Sewerage and Sanitation Development Programme 2020. Surabaya, Indonesia.

Sasse L. (1998). DEWATS: Decentralised Wastewater Treatment in Developing Countries. BORDA, Bremen, Germany.

Snell M. (1997). Cost–Benefit Analysis for Engineers and Planners. Thomas Telford Services Ltd., London, England.

Sugden R and Williams A. (1978). The Principles of Practical Cost–Benefit Analysis. Oxford University Press, Oxford, England.

Sukarma R and Pollard R. (2001). Indonesia: Overview of Sanitation and Sewerage Experience and Policy Options. Water and Sanitation Program for East Asia and the Pacific, Jakarta, Indonesia.

World Health Organization (WHO) and United Nations Children's Fund (UNICEF). (2001). Access to Improved Sanitation: Indonesia. WHO/UNICEF Joint Monitoring Programme for Water Supply and Sanitation Coverage Estimates 1980–2000.

World Health Organization (WHO) and United Nations Children's Fund (UNICEF). (2000). Global Water Supply and Sanitation Assessment 2000 Report.

Chapter 23
Sustainable Community Water Project Implementation in Jordan

Samira Smirat, Stan Benjamin, and Noel Keough(✉)

Abstract From 1998 to 2004 the authors participated in a water management project in the central Jordan Valley focused on community participation in the design and installation of an integrated zero effluent septage treatment facility and farming operation. This chapter begins with a summary of the water management context in Jordan and internationally. The participatory methodology employed in the project, including

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the sustainable community model guiding the project, is outlined and the major activities associated with the project and the community outcomes achieved, are discussed. Finally, the lessons learned, including participation in project design, management and facility ownership and operation; water privatization, pricing and valuation; women and water, cultural versus technical approaches to water management and the role of the professional in community water management are discussed. Our experience suggests the need to adopt a new paradigm in water management whereby all actors—engineers and technicians, community process facilitators, local authorities, other levels of government and the community—work in partnership. To date, decentralization has focused on the technical requirements and economic advantages of decentralized technologies. This project highlights the equally important cultural, social and political dimensions of successful water management decentralization.

23.1. Introduction

From 1998 to 2004 PLAN:NET Ltd., a Canadian development consulting company with Middle East regional offices in Amman, Jordan, was involved in The *Jordan Valley Integrated Waste Management Project (JVIWMP)*, a project jointly funded by the Government of Jordan and The Canadian International Development Agency. PLAN:NET's work focused on community participation in the design and installation of an integrated zero effluent septage treatment facility and farming operation in the central Jordan Valley.

This chapter begins with a summary of the water management context in Jordan and internationally, followed by the outline of the participatory methodology employed in the project, including the sustainable community model developed for the project. A discussion of the project's evolution including a description of the water management technology employed, the major activities associated with the project and the community outcomes achieved is presented. Finally, lessons learned, including participation in project design, management and facility ownership and operation; water privatization, pricing and valuation; women and water, cultural versus technical approaches to water management and the role of the professional in community water management, are discussed.

23.2. Water Management in Jordan in International Context

Peter Beaumont (2002) identifies Jordan as being in the worst water deficit and supply situation of any Middle East country, where already aquifers are being mined to meet supply, that merely diverting water from irrigation to residential and industrial uses will not meet demand to 2025 and that alternative supply either through desalination or diversion from Lebanon via Israel will be a costly endeavor, and politically unrealistic, respectively. In fact, Rafid Alkhaddar et al. (2005) report that by 2010 Jordan faces the prospect of 7 million inhabitants consuming a mere 85 m^3/capita/year. Exacerbating the meagre water supply, the Ministry of the Environment (2006) reports that, in 2002, 7 percent of Jordanians lived on less than $2 USD/day and 11.7 percent of Jordanians lived below the national absolute poverty line.

Another important dimension of the water management landscape in Jordan is the recent push toward privatization and a pricing regime for the cost recovery of water management infrastructure (Ibrahim and Rabadi, 2003). In addition, there is a growing critique of centralized sewerage systems as a silver bullet solution to wastewater treatment and recovery—particularly in relation to the cost of centralized systems, which becomes even more prohibitive in smaller cities and low density urban and peri-urban areas (Mara, 1998).

Finally, Asit Biswas (2005) observes that water management policy is still too uni-sectoral, too engineering oriented and too hierarchical and top-down. To highlight one dimension of the issue raised by Biswas, it is universally recognized that women play a large role in water management at the household level. There is, however, considerable literature demonstrating that the fact of women's dominant role does not guarantee the improvement of women's lives if they are not included in project planning processes (Mosse, 1994; Van de Molen, 2001) and if their unique needs (e.g., their multiple roles in the household [Uphahyay, 2001] or need for micro-credit [Uphadyay et al., 2005]) are not factored into water management programs.

23.3. A Methodology in Support of Community Participation and Sustainability

Brazilian adult educator Paulo Feire (1968) argued that development has to be about the empowerment of individuals to understand their world and to act to change it for the better. Robert Chambers (1999), whose work was inspired by Freire, is perhaps the most well-known practitioner of participatory rural or rapid appraisal (PRA), a family of approaches and methods to enable rural people to share, enhance and analyze their knowledge of life and conditions, to plan and to act. Chambers proposes PRA as a means to ensure that change will be sustainable. He outlines three foundations of PRA: the change of behavior and attitudes of outsiders toward facilitation, not domination; the shift of methods from closed to open, from individual to group, from verbal to visual and from measuring to comparing; and partnership and sharing of information, experience, food and training between insiders and outsiders and between organizations.

The following principles, elaborated by Keough (1998), and inspired by Freire and Chambers, approximate PLAN:NET's approach to the JVIWMP: put people first, take a holistic approach, approach each situation with humility and respect, understand the potential of local knowledge, acknowledge diverse ways of knowing, recognize the relativity of time and efficiency, put reality before theory, embrace uncertainty, adhere to democratic practice and maintain a sustainability vision. Our sustainable community model (Figure 23.1) integrates four themes:

Figure 23.1 Sustainable community model

waste management, public health, sustainable local economic development and water conservation.

23.4. The Jordan Valley Integrated Waste Management Project

23.4.1. The Technology

The purpose of the JVIWMP was to "significantly and sustainably improve the environmental and public health conditions in the Jordan Valley arising from current practices of septage waste management practice, and to share the experience directly with government officials of Jordan and other countries of the Middle East." (Komex, 2005) The JVIWMP was designed to test the application of a proprietary technology of the project's Canadian executing Agency, Komex International. The technology uses a combination of mechanical, chemical and biological treatment employing trickling filters, ultraviolet treatment and engineered wetlands. The recovered water and nutrients (compost) were to be used to rehabilitate a tract of dry land into a profitable farming operation. The farm was to eventually be sold to the private sector (Komex, 2000). The project aimed to demonstrate the cost savings associated with medium-scale sewage treatment systems as compared to the projected per household cost of US$ 2200 (four times the average of all urban wastewater projects constructed in Jordan between 1996 and 1997) for a proposed Jordan Valley rural sanitation project using conventional centralized technology (Loredo and Thompson, 1998).

23.4.2. Community Engagement and Participatory Research

The project area (Dier Alla district, less than two hours drive west of Amman) encompasses communities with a total population of about 60,000 people. Though the Jordan Valley is the centre of the agricultural industry in Jordan, most agriculture is large scale and controlled by a small number of individuals and private entities. The existing modus operandi vis-à-vis septage treatment in the Valley consisted of tanker trucks periodically collecting septage household by household, for a fee, and then dumping the septage either in an unsecured open pit or illegally into wadis and streambeds.

The first visit of PLAN:NET Ltd. personnel to Jordan occurred in October 1999. At that time a local project coordinator and Near East Foundation (NEF), a local NGO, were contracted, and the design of the community participation process begun. Three project objectives were identified for the community component of the JVIWMP's design phase: inform the community about the project, communicate community concerns and opinions to the project team and engage the community in the design process.

To fulfill these objectives, a community participation process was designed and carried out. Meetings with local leaders in November 1999 resulted in an endorsement of the community participation process. Through a series of community meetings preliminary concerns were raised. A local stakeholder committee (LSC) was formed. The LSC included representation from the Jordan Valley Authority (JVA) and the Municipal government of Dier Alla, a project team representative and 14 community representatives selected by the community.

A PRA research team composed of members of the LSC, the NEF and the Jordanian project coordinator was formed to document the concerns and opinions of the community. A seven-step research process was implemented in January 2000. The process began with PRA training for the LSC, including completion of the research design. Three thematic areas were identified for the research: issues related to the current wastewater dumpsite, community awareness of plans to construct a new wastewater treatment plant and local community perspectives on the new plant. A research protocol was prepared for each theme including main issues and PRA research tools to be used. The research team was divided into three research groups, with each group conducting household interviews. Households were chosen with attention to income levels, family size, nuclear and extended families, local leaders, government specialists and young educated community members. In addition to the household visits, focus groups were conducted with local mayors, youth, the Bedouin community and both genders. Data analysis was conducted with the entire research team and recommendations from the research arrived at by consensus. The final PRA report contained three distinct sets of recommendations from the community, the LSC and the project team. These recommendations are presented in Table 23.1.

23.4.3. The Project Design Workshop: A Pivotal Event

A pivotal event in the process was the project design workshop held in February 2000 in Amman, bringing together all the main components of the project team.

Table 23.1 Recommendations from the participatory rural appraisal exercise

Source of recommendation	Recommendation
Community recommendations	Build at the location of the current dumpsite
	Construct a new paved road to the plant
	Transfer plant ownership to Twal Al-Janoubi Municipality
	Charge extra fees for the wastewater tankers coming from other areas
	Give employment priority to Twal Al-Janoubi
	Make sure the poorer communities have access to the service
	Ensure public safety and environmental standards for the Project
Local stakeholder committee recommendations	Provide training for local people to work in the facility
	Assist with building local socioeconomic programs beyond the facility
	Provide a role for the LSC in operation and monitoring of the plant
	Project should provide tanker trucks for municipality
Project team recommendations	In future projects involve the community earlier in the process including needs assessment and site selection
	Design a wastewater management awareness program to facilitate participation in plant design and construction
	Provide a stakeholder committee orientation in the project process

The workshop was originally conceived of as the venue for Jordanian and Canadian project specialists to finalize the details of the project design. Through the PRA, the LSC expressed a desire to be part of the workshop. The project team responded by extending an invitation to the LSC to join the two-day workshop. The majority of the LSC took up the offer. For some, it was their first time in Amman.

As a result of the presence of the LSC members, the workshop agenda was altered significantly. The workshop began with a sight visit by project team members from both Canada and Jordan and a presentation of the PRA research findings by the LSC. In Amman, the workshop included project component presentations including engineering design, environmental assessment, agricultural design and community participation. The presentations were each followed by lively debate and discussion with the participation of the Canadian project team specialists, Jordanian government officials, mayors from local communities in the Jordan Valley and the LSC.

One highlight of the workshop was the negotiation of the PRA research report recommendations. Agreement was reached on those recommendations that had already been implemented, those that were accepted by the project team, those to be taken under consideration and those that were deemed to be outside the scope of the project. The results of that negotiation are shown in Table 23.2.

The final element of the workshop was a series of thematic sessions on each project component (e.g., agriculture, construction) with representation from the project team specialists, Jordanian government officials and LSC members in each thematic session. Detailed designs were reviewed and revised and selections made among various design options. Community concerns and suggestions were incorporated into the final project component designs. The precedent and positive outcome

Table 23.2 Recommendation response matrix

Recommendation	Imple-mented	Recommen-dation accepted	To be taken under consideration	Not within project scope
1. New road	X	X		
2. Transfer ownership to municipality			X	
3. Extra fees for municipalities outside of Twal Al-JAnoubi			X	
4. Priority employment	X	X		
5. Height of stack	X			
6. Sewage system for TAl Al-Manteh and Um Hammd				X
7. Ensure standards will be honored	X			
8. Training for community members	X			
9. Continuing socioeconomic programs				X
10. Community role in plant operation and monitoring	X			
11. Donation of two tanker trucks				X
12. Subsidized agricultural products			X	

of the community participation in the design workshop eventually paved the way for the inclusion of LSC members on a hiring committee, a tipping fees committee and an environmental monitoring committee.

The participatory model continued to guide the construction phase of the project. PRA methods were used to identify and rank community issues and identify solutions to priority issues. A series of workshops delivered with the LSC addressed issues of community project management, bookkeeping, fundraising, solid waste management and wastewater management including recycling and composting, public health, financial management, revolving credit management, business planning, report writing, and presentation skills. Two successful community proposals secured outside funding for a household composting demonstration project and a revolving loan fund. Study tours were used as an education method with community members visiting protected areas, including the Dana and Al-Asraq Ecological Reserves, to learn about water conservation and organic farming. Another study tour included visits to existing wastewater treatment plants to learn the basics of wastewater treatment.

23.4.4. Status of the Technology Transfer

At the time of writing of this paper the waste treatment facility was fully operational. Effluent from the facility is within the standards set by the government of Jordan. There appear to be no problems associated with odor or pests. As a result of the project approximately 125,000 m^3 of sewage effluent is being treated annually that otherwise would not have been (Komex, 2005). The originally barren land upon which the facility is located has been significantly greened with date palm trees and other drought resistant species and wildlife is beginning to inhabit the area. The farm is not yet operational but work continues on rehabilitation of the land and planning for agricultural production. It is unclear whether the JVA will continue to operate the farm or contract the operation to a private entity.

23.5. Community Outcomes

Through the hiring committee dozens of local people worked as project laborers and community people were eventually hired as plant operators. Tipping fees were agreed upon by the JVA, the community, the truck drivers and the facility managers. The open unsecured dumping pit has been closed and illegal dumping of waste water has significantly decreased.

Through end-of-project interviews with LSC members it was clear that members of the LSC had been empowered by their experience and that individuals and the community had benefited. Perhaps the most significant unanticipated outcome of the community engagement component of the JVIWMP is the increased capacity and profile of the That Al-Netakien Women's Society.

Before the project, the Society was on the verge of closing. It was conducting few activities and did not have enough revenue to cover its expenses. The rental of its facility for JVIWMP workshops and meetings provided the Society with some income with which to pay their overhead expenses. In 2003, the Society modified its by-laws to include the implementation of socioeconomic and environmental projects as an objective. This gave the Society the ability to apply for, and receive funds, for such projects.

At the beginning of the project, men in the community were resistant to the idea of women joining the LSC. The Coordinator's strategy was to work slowly, build trust and gradually encourage women's participation and involvement. Remarkable progress was made. At the beginning, men and women sat separately at LSC meetings and workshops, and women spoke rarely. By the end of the project, the women were actively involved and taking on leadership roles within the LSC.

In post-project interviews (Smirat, 2006) female LSC members said that because of this project, they had gained information, skills and confidence. This has carried over into their home lives, making them more confident and able to participate more fully and equally in family decision making. One woman described the benefits as follows:

"*I never dreamed of the benefits I would gain from my involvement in the project as an administrative member in That Al-Netakien Women's Society and a member of the LSC. I gained a lot of skills and knowledge from my participation in all the workshops that were conducted over the course of the project. I was able and confident to represent our NGO in the regional activities. I have participated in regional activities in a workshop in Syria and am going to participate in the Urban*

Development workshop, which will take place in Egypt in November 2004. This was my first travel outside Amman. Even at home my personality has been changed, I get involved more in our family issues and I participate in making family decisions, which was not the case before. I never dreamed this could happen to me."

Another illiterate female member of the LSC from a poor socioeconomic background described her experience in the following way:

"*I am an illiterate woman, I work in the farms. It never crossed my mind that I could sit around the same table with highly educated people and decision makers. The project offered me this opportunity and I received the same training as the other educated LSC members. This added a lot to my knowledge and to my personality. I became more confident when the community asks me questions about the project and I reply to them and gave the same answers the mayor or the headmistress does. In addition, I gained economic benefit from the Revolving Credit Fund project. I received a loan of 700 JD, I bought a cow and I sell yoghurt, milk and labneh. I make 35 JD per week and am now contributing to covering my family's expenses, paying school fees, giving my kids pocket money in addition to the food I can provide for my family. The most important thing is that I can decide myself on purchasing some things without getting my husband's approval as I did in the past, because I needed him to give me the money*."

That Al-Netakien is now a thriving NGO in the Jordan Valley, its advice is sought and contracted by government, a successful revolving credit scheme is ongoing and young women have taken on positions of leadership. The project itself has become a focus of study tours organized by other community projects including a Habitat for Humanity project located in the north Jordan Valley and the Karak Greywater Treatment Project. In March 2005, the Society was able to secure funds through the Ministry of Planning for an agriculture project. On the strength of its own efforts and with assistance from Jordan's Industrial Development Bank, the Society has recently secured sufficient funds to construct its own building.

23.6. Lessons Learned

23.6.1. Problem Definition Should Precede Prescription of a Solution

There is little doubt that water management is a pressing issue in Jordan. However, the community was resistant to the imposition of a solution to a problem that had been defined independent of community consultation. Two assumptions were made prior to any engagement with the community: water management is the primary issue in the community and a particular technology is the most appropriate response to the water management issue. Water management interventions are vital but singular interventions in communities that generally struggle with many issues including employment, local economic development, housing, health care, education, energy supply and gender roles and equity. The lesson from this experience is that water management interventions should be planned and implemented within an integrated approach to community sustainability.

23.6.2. Project Management Protocols Should be Agreed Upon by All Parties

In the conventional approach the professionals manage, and moreover, the technical professionals play a dominant role. In participatory approaches there is a much greater demand for co-management or community-led management with all professionals playing a supportive, facilitative or consultative role. In this project the participatory process kindled community aspirations to be masters of their own house. A struggle ensued between project team members and the community as to how the project should be managed. This points to the need for all partners to establish clear management protocols and a mutually respectful working relationship at the very outset of a project.

23.6.3. Project Ownership has to be Explicitly and Transparently Resolved

Early on in the project the municipality of Twal Al-Janoubi expressed the desire to investigate the potential for the municipality to be granted ownership and management of the waste treatment plant. LSC members felt that perhaps with local ownership there would be more local economic development opportunities, a greater likelihood that the plant would be operated and maintained in an environmentally safe manner and that local people would be able to obtain employment at the facility. How feasible are these local aspirations and how valid are the assumptions underlying them? These are questions that should be addressed in a participatory fashion during project planning.

23.6.4. Privatization and Universal Access to Basic Services

This project grappled with the issue of access to water management services in a climate of privatization. The financial viability of the JVIWMP was based upon the assumption that tipping fees would be charged for truckers to dump septage at the facility, that households would therefore pay a fee to have their septage removed, and that the tipping fee and returns from the farm operation would make the whole operation an attractive investment for the private sector. Truck drivers and owners were concerned about the tipping fee they would be charged to use the facility. The community voiced its concern that the poorest members of the community not be excluded from access to services. It remains to be seen if low-income households will be able to afford the septage pumping costs. Within a national policy environment favoring privatization, what policy and legal frameworks are required to ensure that universal access to water management services is maintained?

23.6.5. Sustainable Water Resource Pricing and Valuation

There has also been a significant international move toward privatization of water provision and waste treatment. One issue that arises in such a discussion is the resource valuation of the water whether in a usable form or in a form that requires an extraction process (i.e., septage treatment). In the JVIWMP the executing agency created a business plan based on market-oriented economics. The raw septage was a necessary component of that business plan, and was assumed to be a free good. In such a scenario what is the value of the raw resource possessed by the household (i.e., water contained within the sewage). Like raw ore to a mining operation the water in the sewage presumably has some value greater than zero.

If establishing a value for sewage would result in a higher percentage of cesspools being lined and pumped, what social and environmental benefits might accrue to individuals and to society? Would the benefits be great enough that a household might actually be paid for its raw material (septage) rather than paying to have it removed? Accepting that water is a vital national asset, and a human right, more innovative economic models are needed to equitably and effectively apportion the costs and benefits associated with sustainable water management including the economic valuation assigned to water, so that the most effective model for water protection and conservation is employed.

23.6.6. Women and Water Management: A Clear Case of Mutual Benefit

The project demonstrated not only that women can be included as full participants in projects but that the inclusion of women can provide a powerful "value added" to the project and the community. Women initially played a more peripheral role, but with a clear mandate to enhance equal gender participation in the project, the project team was able to facilitate the emergence of women into significant roles. Women emerged as key actors in the Local Stakeholder Committee and have leveraged that experience to go on to play a key role in the ongoing development of their communities. Women gained some monetary independence, assumed leadership roles in the community and reported increased self-esteem; in short, they experienced empowerment.

23.6.7. The Technical Versus Social Definition of the Problem

Is water management a technical or a sociocultural issue? Well, it is both obviously. However, in the JVIWMP the PLAN:NET team initially found itself in a contractual relationship with an technical executing agency that appeared to make the assumption that the role of the community participation partner was public relations, or gaining community acceptance for the planned water technology transfer. The question this raises is as follows. Is water management a matter of social engineering to ensure the smooth operation of a given technological solution, or is the search for an appropriate technological response to a problem embedded in a larger social–cultural process? In other words does the technology or the community drive the project? In our view it has to be the community. And this has profound implications for the planning and management of water management projects. Community engagement has to come much earlier, be much more profound and be afforded significantly more resources for water management interventions to be effective and sustainable.

23.6.8. The Evolving Role of the Professional in Water Management

This project highlights the need to consider a new role for the professional or expert in a more decentralized and integrated water management paradigm. The vision we would propose for effective local water management is much more profoundly integrated and collaborative than the conventional model. Partnership is key. Roles are less rigid and more flexible as all partners define their contribution and working relationships. It is no less than a cultural evolution where many of the norms and conventions are questioned and examined and revised. In the new paradigm engineers and technicians get a better result by using the power of the community organizers to get their water systems up and running and maintained. Community organizers get a better result using the technologies as an entry point to empowering the community and its marginalized citizens. The community experiences all of the benefits associated with an effective water management technology by learning how to use its newly gained confidence and skills for social and economic benefits. And to the extent that the model works and is replicable, the nation benefits through the stability that effective water management and ultimately, increased wellbeing, provides for its citizens.

23.7. Conclusion

The road to decentralization of water management is a potentially fruitful but complex one. To date the debate has focused on the technical and economic advantages of decentralized technologies. Examination of this case study highlights the equally important cultural, social and political dimensions of the debate.

References

Ibrahim A and Rabadi A. (2003). Commercialization and public–private partnership in jordan. Water Res Dev 19:159–172.

Alkhadder Rafid M, Sheehy William JS, Nadhir Al-Ansari. (2005). Jordan's water resources: supply and future demand. Water Int 30:294–303.

Beaumont P. (2002). Water policies for the Middle East in the 21st century: the new economic realities. Water Res Dev 18:315–334.

Biswas Asit K. (2005). An assessment of future global water issues. Water Res Dev 21:229–237.

Chambers R. (1999). Whose Reality Counts? Putting the First Last. Intermediate Technology Publications, London.

Freire P. (1968). Pedagogy of the Oppressed. Seabury Press, New York.

Keough N. (1998). Participatory development principles and practice: Reflections of a western development worker. Com Dev J 33:187–196.

Komex International Ltd. (2005). Jordan Valley Integrated Waste Management Project—Final Report, Transfer Phase. Komex International Ltd., Calgary, Canada.

Komex International Ltd. (2000). Jordan Valley Integrated Waste Management Project—Project Implementation Plan. Komex International Ltd., Calgary, Canada.

Loredo D and Thompson R. (1998). Assessment of Jordan Valley Rural Sanitation Feasibility Study. Environmental Health Project (EHP), Activity report no. 44. USAID, Washington, DC.

Mara DD. (1998). Low-cost sewerage. In: Simpson-Hébert M, Wood S, eds. Sanitation Promotion. Report no. WHO/EOS/98.5. World Health Organization/Water Supply and Sanitation Collaborative Council, Geneva, Switzerland, pp. 249–262.

Ministry of the Environment. (2006). Environmental Profile of Jordan. Ministry of Environment, Amman, Jordan.

Mosse D. (1994). Authority, gender and knowledge: theoretical reflections on the practice of participatory rural appraisal. Dev Change 25:

Author: Please provide page range for Mosse 1994.

Smirat S. (2006). Role of Women in Water Management and Conservation in Jordan. PLAN:NET Ltd., Amman, Jordan.

Upadhyay B. (2004). Gender Roles and Multiple Uses of Water in North Gujarat—Working Paper 70. International Water Management Institute, Columbo, Sri Lanka.

Upadhyay B, Samad M, Giordano M. (2005). Livelihoods and Gender Roles in Drip-Irrigation Technology: A Case of Nepal—Working Paper 87. International Water Management Institute, Columbo, Sri Lanka.

Van der Molen Irna. (2001). An Assessment of Female Participation in Minor Irrigation—Working Paper 8. International Water Management Institute, Columbo, Sri Lanka.

Chapter 24
Waste Water Reuse for Agriculture Pilot Project at the Jordan University of Science and Technology

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Abstract Because of water scarcity in Jordan, marginal water (treated wastewater in particular) use in agriculture is highly required. However, this needs to be done with precautions to avoid harming the valuable agricultural soils and to prevent any consumer health risk.

The Jordan University of Science and Technology (JUST) has a large campus ($11\,km^2$) and has reused water from the university treatment plant for almost 20 years. The campus plant has a design capacity of $2{,}500\,m^3/d$ but is currently operating at about $600\,m^3/d$. The other source of effluent water is

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located off campus at Wadi Hassan area about 4 Km south of the university campus. The design capacity of this plant is 2,200 m^3/d and it has been in operation since September 2001. There are two storage lakes on campus: a 132,000-m^3 lined pond and a 110,000-m^3 capacity reservoir. These sources of effluent water and the existing infrastructure have encouraged the University to irrigate additional portions of the campus and to support the production of cash crops, field crops and forest trees by reclaimed wastewater. There is also a desire to have local community involvement and to train local farmers in the management and use of reclaimed water.

JUST pilot has been involving researchers and students in the water reuse activities. Local farming communities and other stakeholders have also been exposed through visits and field days to the reuse activities at JUST so that more positive attitudes can be created about reuse. The U.S. Agency for International Development (USAID)-funded water reuse activities at JUST are of great value also for JUST as a university and for the country due to the research and demonstration value of this activity. JUST water reuse pilot continues its activities to demonstrate and document safe reuse of reclaimed water and reaching at a wider social acceptance of this valuable resource for Jordan.

The goal of the pilot project is to evaluate the efficacy and economics of growing new types of crops in the northern area of Jordan utilizing the flow from the existing JUST wastewater treatment plant (WWTP) as well as the Wadi Hassan WWTP. The crops for the pilot study are selected based on their applicability to the climate and soils of JUST, as well as their marketed value.

24.1. Background

In recognizing the imperative to identify renewable water resources for Jordan, the Ministry of Water and Irrigation (MWI) and the USAID have collaborated in implementation of a major project to demonstrate that recycled water can be used for various requirements efficiently, cost effectively and in a safe, environmentally friendly manner. As found in most regions of the world, and the Middle East in particular, water resources are becoming increasingly limited, and innovative solutions are sought to address critical shortfalls. In Jordan, over-exploitation of aquifers has resulted in ever-decreasing levels of ground water. At the same time, the population is rapidly increasing and the concomitant demand for supplies of fresh water. Against this backdrop, the government of Jordan through the MWI has made the decision to institutionalize water reuse, to monitor it and control it in a manner that does not threaten the environment. It is recognized that reclaimed water can be applied within a limited and controlled context, and under appropriate conditions will be a critical factor in securing adequate water supplies for the future.

The overall goal of the Jordan program is to implement direct reuse of treated wastewater in Jordan that is reliable, environmentally sound and cost-effective.

Institutional capacities to develop water reuse and implement relevant supporting actions have been significantly enhanced through a number of significant initiatives. The Government requires the capacities to plan, manage and monitor practical and sustainable direct water reuse projects associated with other medium and small-scale WWTP in the Kingdom. Demonstration of the practical application of water reuse principles has been achieved through the implementation of pilot demonstration farms in three areas of the country; these are Wadi Musa, Aqaba and the JUST campus.

24.2. Objectives

The specific objectives of the water reuse pilot project at JUST are:

- Expand irrigated areas to maximize the utilization of reclaimed water generated by JUST's WWTP and allocation from Wadi Hassan WWTP.
- Develop the procedures of operation and maintenance for the pilot that are accepted by the system operators and users.
- Documenting the safe practice of reuse and impacts on crop quality, health and the environment.
- Develop a cost-effective on-farm management of agricultural inputs to achieve the sustainability for the pilot.
- Provide a baseline for the policy makers in the reuse sector. Baseline values concern the reuse operation and maintenance, product quality, health and environmental impact in agriculture.
- Educational outreach program for students and local community.
- Develop a business plan for the pilot's crop varieties.

24.3. Site Description

The JUST Pilot Demonstration Project is being managed through a contractual agreement entered into with USAID in March 2003. The project is located on the campus of the university and includes five sites where treated wastewater is applied to a variety of agricultural crops utilizing several different irrigation methodologies. The five sites encompass a total of 720 dunums (72 hectares) of land that was not previously cultivated. The university provides technical support, including a variety of agricultural and engineering specialists.

Starting in January 2003, water reuse implementation activities have been operating at four sites on the JUST campus, in addition to one site at the nearby wastewater treatment plant under the management of the Water Authority of Jordan (Wadi Hassan WWTP). Addressing human and environmental health concerns usually associated with reuse of treated wastewater is at the forefront of the study objectives.

The project design includes installation of both trickle and sprinkler irrigation in a controlled pilot are systems in all sites, planting of cash crop trees (such as almonds, pistachio, pine) as well as field crops (such as barley, vetch, alfalfa).

Harvest of cultivated crops is sold and income is donated to support the needy students through the "needy student" fund in the university. Figure 24.1 shows the reuse sites of the projects that are located within the JUST campus. Following are the areas of these sites: site 1, 100 donums; site 2, 400 donums; site 3, 35 donums; site 4, 66 donums; and site 5, 120 donums in Wadi Hassan (off campus).

24.4. Irrigation Systems

Drip irrigation is the main irrigation system installed in the five project sites. Almost all types of drippers were used. These include inline emitters (GR), online pressure compensating emitters and beside new adjustable emitters (click emitters). Moreover, about 12 dunums grown with fodder crops/alfalfa in site 1 is served by a new innovation of sprinklers called floppy sprinklers.

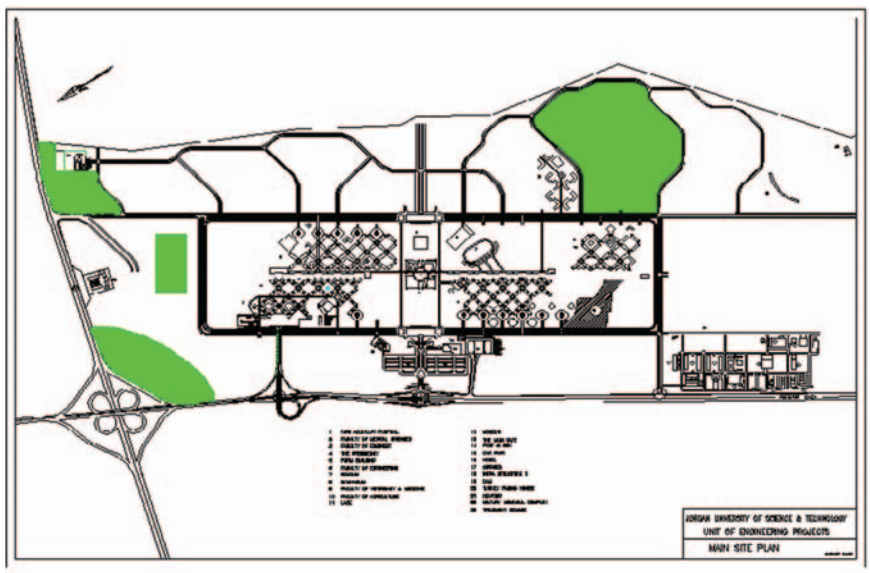


Figure 24.1 The four Reuse pilot project sites located within Jordan University of Science and Technology campus

24.4.1. Floppy Sprinklers

Floppy sprinklers are a new concept in irrigation. Also called overhead sprinkler, it is manufactured from stable engineering plastic and silicon tubing that pops up with incoming water pressure when spraying action starts and pops down immediately as the water pressure is released (see Figure 24.2). Floppy sprinklers provide a desirable rain-like irrigation effect with relatively large, droplet-size water, thus reducing wind drifting and enhancing uniformity. The freedom that the silicon tubing provides this unique irrigation method ensures full circle spraying patterns that prevent dry patches. These sprinklers can be installed in two ways; mounted on risers or dangled from wires, thus no moving parts, which eliminate maintenance.

Floppy sprinklers are ideally suitable for landscape and turf irrigation. They are especially recommended for irregular undulating terrain and hilly areas. Compared to conventional sprinklers, floppy sprinklers are believed to reduce evaporation losses and mitigate mist creation; thus protect farmers from air-borne pathogens. While the Jordanian standards (JS 893) do not allow for sprinkler use in effluent irrigation; floppy sprinkler system was allowed for research purposes in a controlled pilot.

24.4.2. Click Emitters

Click emitters are a new adjustable type of emitters as well. These emitters enable a wide variety of discharges by adjusting the click openings while keeping the

Figure 24.2 Irrigation operation for field crops at site 1

pressure unchanged. An emitter consists of two parts that are a perfect fit, as one is screwed into the other it clicks indicating a quarter of a turn (see Figure 24.3). Ideally, if the farmer set all of the emitters to four clicks, for instance, they should each give the same discharge. However, that can not be the case if the click emitters are not well-manufactured.

24.4.3. Irrigation Systems Evaluation

Two terms which describe the performance of irrigation systems are uniformity and irrigation system application efficiency. Irrigation efficiency is a concept used extensively in system design and management. It can be divided into two components, uniformity of application and losses. If either uniformity is poor or losses are large, efficiency will be low. Low application efficiencies are related to water losses mainly due to runoff and/or deep percolation. This efficiency will be further reduced by non-uniform water application. In case of minimal water losses as with well managed trickle irrigation systems, emission uniformity is usually used as indicator of efficiency. It is critical that the irrigation system be designed for high uniformity and that this high uniformity maintained throughout the life of the system.

24.4.4. Distribution Uniformity (DU), Emission Uniformity (EU) and Coefficient of Uniformity (CU)

Uniformity of water application with sprinkler irrigation systems is often reported as DU. It is an indicator of how equal (or unequal) the application rates are in the field. A low DU (below 60%) indicates that application rates are very different, while a high DU (80% or higher) indicates that application rates over the area are similar in value and the water is distributed evenly to all the plants. DU is based on the low quarter of the irrigated area. The DU is termed as EU when referred to trickle irrigation systems.

The calculation of DU requires that the catch-can test be performed in the irrigation zone.

$$DU = \frac{avg.low\,qtr.depth}{overall\,avg.depth} * 100\%$$

The average low-quarter depth of water received is the average of the lowest one-quarter of the measured values, where each value represents an equal area.

The other parameter of focus in the evaluation process for sprinkle irrigation systems is the CU, developed by Christiansen (1942). Values of CU greater than 75% are acceptable for general field analysis.

$$CU = \left[1 - \frac{ave.deviation\ from\ avg.depth}{overall\ avg.depth}\right] * 100\%$$

24.4.4.1. The Catch-Can Test

A catch-can test is simply placing the catch-cans in such a way as to collect the intended amount of water from the emitters/sprinklers that are being tested. Whichever type the irrigation system, the catch-can test is the same; the only difference is its position with respect to the emitter/sprinkler itself. A catch-can is placed underneath an emitter in a line of drip emitters, but placed in the areas between the risers in floppy sprinkler irrigation systems. The number of catch-cans placed in the test block depends on the system being tested and the spacing of the sprinklers. The volumes held in the catch-cans are then, normally, measured using a graduated cylinder.

24.4.4.2. Evaluation of Floppy Sprinkler Irrigation System

Evaluation had been conducted for floppy sprinkler irrigation system at site 1. Results of catch can evaluation for floppy sprinkler irrigation system at site 1 indicates the good performance of the irrigation system with relatively high values of both DU and CU coefficients, thanks to the large droplet size and the specific pressure-discharge relationship characterizing this innovative sprinkle system.

24.4.4.3. Evaluation of Drip Irrigation Systems

Evaluation of Drip Irrigation Systems at all sites is normally conducted during every growing season. The field EU for each block of the drip irrigation system in each site is estimated. In each block, at least four laterals are chosen in such a way to represent the remaining laterals in the block. For each selected lateral, the flow rates in (L/h) of at least four emitters are measured. Then, the block EU is calculated as the percentage of the average low quarter to the total average of all emitters tested. EU percentages for almost all sites are usually in the upper 80s which reflect the benefits of the continuous onsite checking/maintenance efforts.

24.5. Crop Water Requirement (CWR)

Knowing the CWRs is an essential step for proper irrigation scheduling. The CWRs were estimated by multiplying the grass reference evapotranspiration by the crop coefficient following the Food and Agriculture Organization of the United Nations

(FAO) method described in the FAO-56 irrigation and drainage paper (1998). Maximum and minimum values of CWRs were determined based on the last recent 10-year climatic data available for the region.

24.6. Management of Wastewater Reuse in Irrigation

Crop yield reduction associated usually with wastewater reuse in irrigation may result from osmotic stress caused by the total soluble salt concentration (i.e., salinity), toxicities or nutrient imbalances created when specific solutes become excessive or a reduction in water penetration through the crop root zone caused by excess sodium inducing a deterioration of soil permeability. The key to managing salinity is leaching, the net downward movement of water through the crop root zone. Salts are leached whenever water applications, either rainfall or irrigation, exceed evapotranspiration, provided soil infiltration and drainage rates are adequate.

24.6.1. Irrigation Management for Salt Control

CWR and irrigation water quality are the primary parameters that have to be considered to ensure effective irrigation management for salt control.

24.6.2. Leaching Salts From the Root Zone

In arid climates, irrigation must supply all water requirements of the crop for the growing season. Additional water must be applied to remove the salts from the root zone to avoid a build-up of salts which will exceed the threshold level for a given crop and result in yield reduction. The amount of additional water is usually expressed as a leaching fraction which is a dimensionless number. The leaching requirement for sprinkler and surface irrigation can be expressed by the following equation:

$$LF = ECw$$
$$5\,(ECe) - ECw$$

Where, LF = minimum leaching requirement needed to control salts within the tolerance (ECe) of the crop with ordinary surface methods of irrigation, ECw = salinity of the applied irrigation water in dS/m and ECe = average soil salinity tolerated by the crop as measured on a soil saturation extract. It is recommended that the ECe value that can be expected to result in at least a 90% or greater yield be used in the calculation.

In case of frequent irrigation as the case when using trickle irrigation systems, the LF can be estimated using the following equation:

$$LF = ECw$$
$$2\ (ECe)max$$

Where, (ECe)max = zero yield salinity (i.e., the maximum salinity that causes zero crop yield as measured on a soil saturation extract).

24.6.3. Irrigation Requirements

The irrigation requirement of a crop can be defined as the total amount of water that must be supplied by irrigation to a disease-free crop that is growing in a large field with adequate water and fertility. Irrigation requirement includes the water used for crop consumptive use, maintaining favorable salt balance with the root zone, and overcoming the non-uniformity and the inefficiency in irrigation.

The irrigation requirement is estimated as:

$$IR = \frac{CWR - Pe}{Ea\,(1 - LF)}$$

Where, IR = irrigation requirement, CWR = crop water requirement, Pe = effective precipitation and Ea = irrigation application efficiency or emission uniformity (EU).

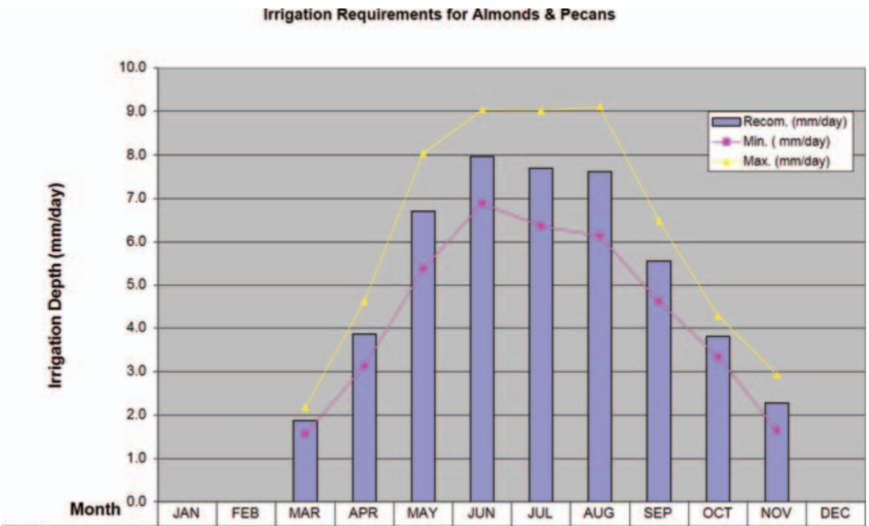
Based on the maximum and minimum CWR, Ea or EU and LF monthly recommended irrigation requirements for all crops were determined for better planning and easier/practical irrigation scheduling. For example, Figure 24.4 shows the maximum, minimum and recommended daily irrigation requirement depths on monthly basis for almond and pecans trees.

24.6.4. Irrigation Schedule

Irrigation scheduling is the term used to describe the procedure by which an irrigator determines the timing and quantity of water application. Accordingly, the two classical questions of irrigation scheduling are when to irrigate and how much water to apply.

A practical approach to scheduling of irrigation at JUST wastewater irrigation system while based on CWRs it also takes into account the need to have a simple and transparent operational approach that makes it easy for the system operator to follow the schedule. For that reason, water marks were introduced to act as indictors for soil moisture monitoring and hence help the operator to decide when to irrigate.

Figure 24.3 Click emitters



Month	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
Recom. (mm/day)			1.9	3.9	6.7	8.0	7.7	7.6	5.6	3.8	2.3	
Min. (mm/day)			1.6	3.1	5.4	6.9	6.4	6.1	4.6	3.3	1.7	
Max. (mm/day)			2.2	4.6	8.0	9.0	9.0	9.1	6.5	4.3	2.9	

Figure 24.4 Irrigation requirements for Almond and Pecan trees

Regarding how much water to apply, irrigation amounts were determined based on crop irrigation requirements as described previously, However, irrigation set times in hours for each crop were determined based on crop planting area and irrigation system flow rates to simplify the irrigation scheduling process.

Since the first operation of the irrigation system, regular records have been carried of the daily amounts of applied reclaimed water by the pumping system. Furthermore, water allocation of each plot has been recorded with the operation hours for that plot.

24.7. Plant Growth and Development

Plant growth and development were monitored periodically since the beginning of the project. All indicators of growth monitored indicate the well irrigation management and agricultural practices followed during the last four years since the beginning of the wastewater reuse project at JUST. Figures 24.5 and 24.6 are just two examples to show the good growth development of both pine trees and cactus plants which are originally suggested and introduced by project team committee as potential crops, as well as other crops as mentioned previously, which can be irrigated safely and successfully using non conventional irrigation water sources such as treated wastewater.

Similarly, and on regular basis, agronomy data is obtained for field crops demonstrated in the project. Figure 24.7 shows growth development for barley grown with treated effluent (supplemental irrigation) compared with barley grown on rain water only (rain-fed control).

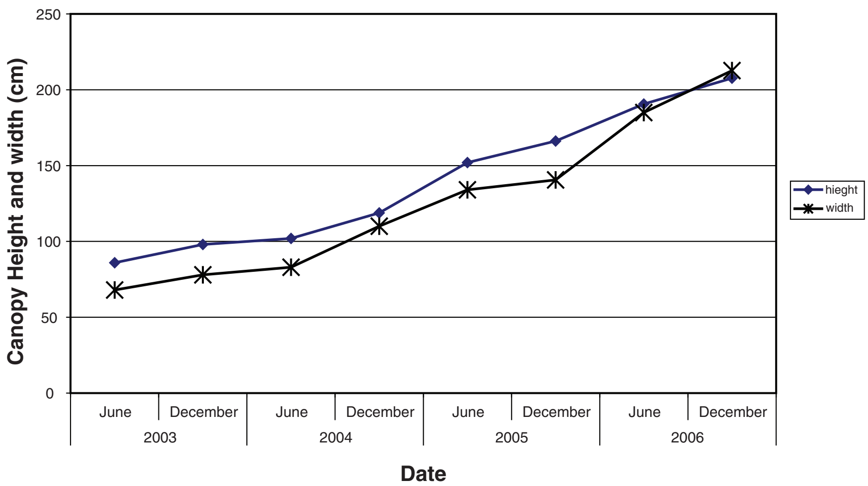


Figure 24.5 Pine Growth development of Pine trees in Site #1

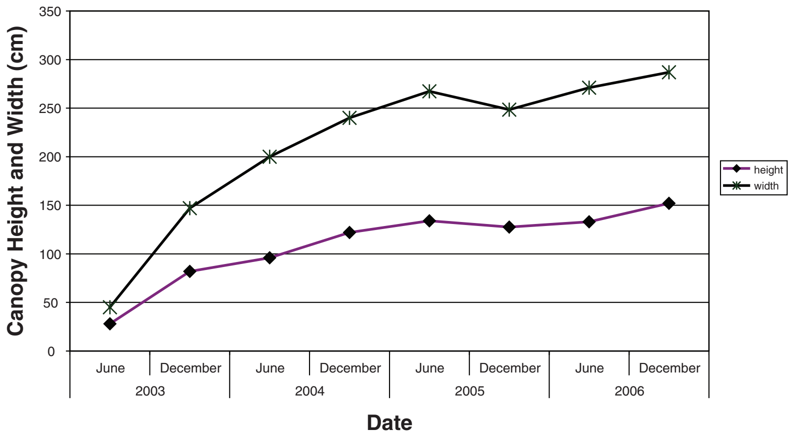


Figure 24.6 Growth development of Cactus trees in Site #1

Figure 24.7 Presentation for the growth development of the barley field crop at site 1 (December 2005)

24.8. Environmental Analyses

A comprehensive monitoring program has been in place since project inception to collect periodic environmental quality data on reclaimed water, soil as well as agricultural produce. This is essential to document and demonstrate the safety of waste water reuse and to mitigate impacts on public health and the environment as well. In addition, produce is analyzed for nutritional value as compared with same variety produced with fresh water. This gives rich data on possible impacts on crop quality, safety for consumption as well as marketability.

In this section, example analyses are presented for selected crops of the JUST water reuse project. In all analyses done so far in this project there have been no significant differences between crops irrigated with reclaimed waste water and crops irrigated with fresh water. This has been true for the nutritional value as well as the potential contamination criteria.

Some crop samples are occasionally tested in two different laboratories for cross-checking. Samples are analyzed in the laboratories of RSS (Royal Scientific Society), WAJ (Water Authority of Jordan) and/or the laboratories of JUST University.

24.8.1. Cactus, Alfalfa and Barley

Cactus fruits and pads were sampled and tested for chemical and microbiological contamination at two Labs. The results are presented in Tables 24.1 and 24.2. Sample results for the analysis of alfalfa and barley crops are also shown in Tables 24.3 and 24.4, respectively.

Table 24.1 Results for Cactus analysis by RSS Lab. Sampling date: August 17, 2005

Parameter	Unit	Cactus cortex	Cactus fruit	Cactus Pads
Pb	mg/kg	<0.05	<0.05	—
Cd	mg/kg	<0.015	<0.015	—
NO_3	mg/kg	69	8	—
E-Coli	MPN/g	—	—	<0.03
Salmonella	Present-Absent/10 g	—	—	Absent

Table 24.2 Results for Cactus analysis by WAJ laboratories. Sample date: August 17, 2005

		Cactus fruit		Cactus pads	
Parameter	Unit	Cortex	Inside	Cortex	Inside
Total coliform		<0.3	0.4	<0.3	0.9
E. coli	MPN/g	—	<0.3	—	<0.3
Salmonella	Present-Absent/10 g	—	—	Present	—
Helminth eggs		—	—	—	—

Table 24.3 Results for Alfalfa analysis by WAJ laboratories. Sample date: July 24, 2005

Parameter	Unit	Alfalfa
Total coliform	MPN/g	110
E. coli	MPN/g	<0.3
Salmonella	Present-Absent/10 g	Absent
Helminth eggs		—
Cd	<D.L.	
Pb	<D.L.	
NO_3-N%		0.45

Table 24.4 Results for Barley analysis by WAJ laboratories. Sample date: June 19, 2005

Parameter	Unit	Barley
Total coliform	MPN/g	4.3
E. coli	MPN/g	<0.3
Thermotolerant		
Helminth eggs		Pathogenic Helminth eggs were not seen

24.9. Educational Outreach Program for Students and Local Community

One of the main aims of JUST Pilot Project is the educational outreach for all stakeholders, especially students and the local community to raise public awareness about water reuse issues and concerns. Moreover, to demonstrate and train interested target groups who are involved in water reuse at all levels.

More than 45 groups of visitors from off campus were accommodated. Public and private schools, universities, national and international NGOs and agencies visited the project and toured its reuse sites. Many demonstrations regarding water reuse in agriculture have been conducted to various farmers groups.

Training and demonstration students enrolled in different courses in the faculties of agriculture, engineering and science have been implemented. Pamphlets and informational extension materials have been distributed to various groups. Training workshops where conducted to accommodate the training needs of national and international engineers and subject matter specialists.

In collaboration with the University of Arizona and USAID, an e-learning graduate course was offered in the Fall 2006/2007 semester. Eighteen graduate students from the schools of Civil Engineering and Agriculture at JUST have successfully finished this course for three credit hours. The course title was Management of Wastewater Reuse in Agriculture.

24.10. Concluding Remarks

The wastewater reuse project at JUST has been progressing since late 2002. The project tries to demonstrate and document safe water reuse in agriculture. Wide range of cash crop trees and field crops are demonstrated. Various irrigation systems are evaluated. Environmental analyses of agricultural produce have so far resulted in no reduction of the nutritional value of any crop due to the use of reclaimed water.

Reclaimed water used in the project does comply with Jordanian Standards for reuse in Agriculture (JS 893/2006). There has been no evidence thus far of any salt accumulation or contaminant build up in the soil of the various sites of the project. The Cash which is generated from selling fodder and other produce is donated by the Project to the Needy Student Fund at JUST.

Educating the public among other stakeholders about water scarcity in Jordan and the need for water reuse continues to be a major objective and outcome of the Reclaimed Water Reuse project.

References

Al-Ghazawi Z., Cardinalli R, and Others; Wastewater Reuse in Irrigation: USAID-WRIP program in Jordan, Proceedings of the first Water Reuse Conference, December 2003, Amman, Jordan.

CDM International. (2005). Wastewater Reuse in Agriculture: Three Pilot Projects, Final Report.

PA Consulting, Inc. (2004). Wastewater Reuse in Agriculture: Three Pilot Projects, Final Report.

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