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Nitrogen mass balance in waste stabilization ponds at the University of Dar es Salaam, Tanzania

Aloyce W. Mayo

Department of Water Resources Engineering, University of Dar es Salaam, P.O. Box 35131, Dar es Salaam, Tanzania.

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Nitrogen mass balance in waste stabilization pond system at the University of Dar es Salaam was determined using a dynamic mathematical model in order to elucidate the biological nitrogen transformation mechanisms that are effective for removal of nitrogen in this pond system. Results show that the pond system removed 4741 g/day of nitrogen from an influent load of 8036 kg/day, which is equivalent to 59% removal efficiency. The overall dominant nitrogen removal mechanism was denitrification, which was responsible for 77.5% of the removed nitrogen. Other permanent nitrogen removal mechanisms were net loss of nitrogen to sediments and volatilization, which contributed 18.2 and 4.3% of the removed nitrogen, respectively. However, sedimentation was the major nitrogen removal mechanism in primary facultative pond, which was responsible for 73.7% of the total nitrogen removed in that pond. On the other hand, denitrification was the major nitrogen removal mechanism in secondary facultative ponds (F2 and F3) and maturation pond, M, which contributed about 95.0, 89.4 and 89.1% of the total nitrogen removed from these ponds, respectively. The major nitrogen transformation routes were mineralization and ammonia uptake in the primary facultative pond F1. In secondary facultative pond F2, nitrification and denitrification were the dominant nitrogen transformation mechanisms, while in secondary facultative pond F3 and maturation pond M, ammonia uptake was the dominant transformation route. The results obtained in this work may be used as a management tool in assessing the levels of nitrogen compounds in waste stabilization ponds and thus protect the water bodies downstream.

Key words: Nitrogen dynamics, waste stabilization ponds, nitrogen removal, mathematical modeling.

INTRODUCTION

Wastewater stabilization pond treatment technology has been used for removal of organic matter (Zimmo et al., 2005; Mugasa, 2005; Barrie, 2002), pathogenic organisms (Mayo, 1995; Kalibbala, 2001; Osman, 1998) and nutrients particularly nitrogen (Hanai, 2006; Alahmady et al., 2013; Rockne and Brezonik, 2006). This technology can be used for treatment of wastewater from domestic, industrial and agricultural sources (Hanai, 2006; Tadesse et al., 2004; Al-Sa'ed, 2007; Sedlack, 1991). Waste stabilization ponds are largely used in tropical climates (Mayo, 1995; Mara, 2005) although their application in temperate climatic region with temperatures as low as 7°C is not uncommon (Rockne and Brezonik, 2006). However, in cold climates, wastewater is applied at very low organic loading rates of less than 60 kgBOD$_5$/ha/day (Faleschini et al., 2012) and wastewater may be retained for a duration of up to one year (Krkosek et al., 2012).

The major advantages of waste stabilization pond technology include removal of pathogenic organisms without addition of chemicals, low sensitivity to hydraulic and organic shock loads, low construction, operation and maintenance costs, minimum or no mechanical equip-

E-mail: aloyce.mayo@yahoo.com.

ment is required for their operation and low requirement of external source of energy (Mayo, 1995; Shilton and Walmsley, 2005; Isosaari et al., 2010). However, pond technology suffers stability problems with effluent concentrations because of large amount of biomass in the effluent and large land requirement (Isosaari et al., 2010; Vera et al., 2013).

As a result, improvement of effluent from waste stabilization ponds is desirable through rock filters and constructed wetland systems (Manyama, 2005; Mara and Johnson, 2006; Al-Sa'ed et al., 2011). Other efforts to improve nutrient removal and reduce land requirements include introduction of floating elements to improve hydraulic characteristics and attachment of algae and bacteria such as duckweed and water hyacinths systems (Bigambo, 2003; Senzia, 2003; Bal Krishna and Polprasert, 2008).

Nitrogen is becoming increasingly important in wastewater management because nitrogen can have many effects on the environment (Halling-Sørensen and Jørgensen, 1993). Excessive discharge of nitrogen in aquatic ecosystem can contribute to degradation of water quality (Showers et al., 2006), can cause adverse ecological impacts and can affect public health (Sedlak, 1991; http://ammoniabmp.colostate.edu, 2013). For instance, ammonia is extremely toxic to fish and many other aquatic organisms such as algae and it is also an oxygen consuming compound, which can deplete the dissolved oxygen in water (Konig et al., 1987). The depletion of dissolved oxygen in water causes ecological imbalance since maintenance of a high oxygen concentration is crucial for survival of the higher life forms in aquatic ecosystem. Excessive concentration of nitrate and ammonia can cause changes in ecosystem, which may disrupt natural balance of the system. As a result fragile plants and animal species can be replaced by nitrogen-responsive species, which may lead to disruption or even extinction of the ecosystem (http://ammoniabmp.colostate.edu, 2013). Orji et al. (2011) observed that excessive concentration of nitrogen from abattoir wastes was capable of causing shifts in microbial community structure and altering aquatic nitrogen cycle.

Another ecological impact is eutrophication caused by the excessive growth of bacteria and algae due to the increase of the amount of nitrogen discharged into water. Eutrophication contributes to the reduction of the oxygen level in water. Nitrite is a potential public health hazard in water consumed by infants (Sedlak, 1991). In the body, nitrite can oxidize the iron (II) and cause methamoglobinaemia, which binds oxygen less effectively than normal haemoglobin. The resulting decrease in oxygen levels in young children leads to diarrhea, vomiting, and in extreme cases even death (Kelter et al., 1997). The problems that all these incidents have posed are a clear indication that nitrogen removal in wastewater is important before effluent is finally discharged into receiving water body.

In an attempt to understand nitrogen transformation mechanisms researches have been carried out worldwide and models developed in various environments. These environments include activated sludge treatment plants (Charley et al., 1980), algae-based waste stabilization ponds (Ferrara and Hermann, 1980; Senzia et al., 2002; Senzia et al., 2003), attached growth systems (Shin and Polprasert, 1988; Mutamba, 2002, Mkama, 2005), river deltas and estuarines (Di Toro et al., 1971; Najarin, 1984) constructed wetlands (Senzia et al., 2004; Mayo and Bigambo, 2005), natural wetlands (Muraza, 2013), duckweed systems (Bal Krishna and Polpersert, 2008), water hyacinths ponds (Dallah, 2001; Mayo and Bigambo, 2005) and high rate ponds (Mayo and Mutamba, 2005). Some research studies have established the performance of individual waste stabilization ponds (Senzia, 1999; Mayo and Mutamba, 2005), but no attempt has been made to establish the overall performance of the whole system. Therefore, to achieve the performance of the entire system mass balance became a prominent tool to tackle the problem. The objective of this research was to identify major nitrogen transformation and removal mechanisms in different types of ponds.

METHODOLOGY

Description of the study area

The University of Dar es Salaam (UDSM) pond system is located at latitude 6° 48' S and longitude 39° 13' E. It has one primary facultative pond (F1), two lines each with two secondary facultative ponds (F2, F3, F2* and F3*) and one maturation pond (M and M*). They receive wastewater flow of about 840 m^3/day mainly of domestic nature and also some chemical wastes flows from laboratories and workshops. Figure 1 shows a schematic layout of the pond system at the University of Dar es Salaam. Operational characteristics and wastewater flows are shown in Table 1.

Samples were collected both in the influent and effluent of each pond for examination of organic-nitrogen, ammonia-nitrogen and nitrate-nitrogen in water quality laboratory at the University of Dar es Salaam in accordance with the Standard methods (1996). Wastewater flow rate, temperature, pH, dissolved oxygen (DO) were measured *in situ*.

Model development

Nitrogen transformation in waste stabilization ponds was developed using conceptual model shown in Figure 2. A simplified and appropriate nitrogen cycle was developed paying particular attention to mineralization, nitrification/denitrification, uptake by micro-organisms (algae and bacteria), permanent sedimentation (net loss) and ammonia volatilization, as dominant nitrogen pathways.

For mathematical simplicity, either a first order plug flow regime (Reed, 1985; Mayo, 1995; Thirumurthi, 1969) or completely mixed flow (Ferrara and Hermann, 1980; Shin and Polprasert, 1988; Fritz et al., 1979; Somiya and Fujii, 1984) models were adopted. Assuming completely mixed flow regime, a complete materials

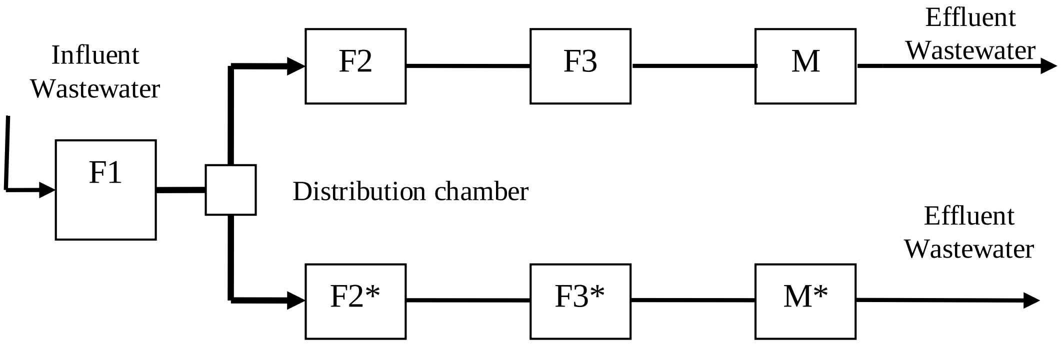

Figure 1. The layout of waste stabilization pond system at the UDSM.

Table 1. Hydraulic elements and wastewater flows (Senzia, 1999).

Pond type	F1	F2	F3	M
Retention time (days)$^+$	11.5	12.2	12.5	7.6
Flow rate (m^3/d)$^+$	528.23	252.41	245.61	241.41
Volume (m^3)	6080*	3070	3070	1824
Water depth (m)	1.33*	1.51	1.51	1.22
Mid pond surface area (m^2)	4065	2100	2100	1425

+ These are mean values; * these are estimated values, which took into account sludge accumulation (910 m^3) since the pond was desludged in 1986. In 1986, the depth and volume of the pond were 1.82 m and 6990 m^3, respectively.

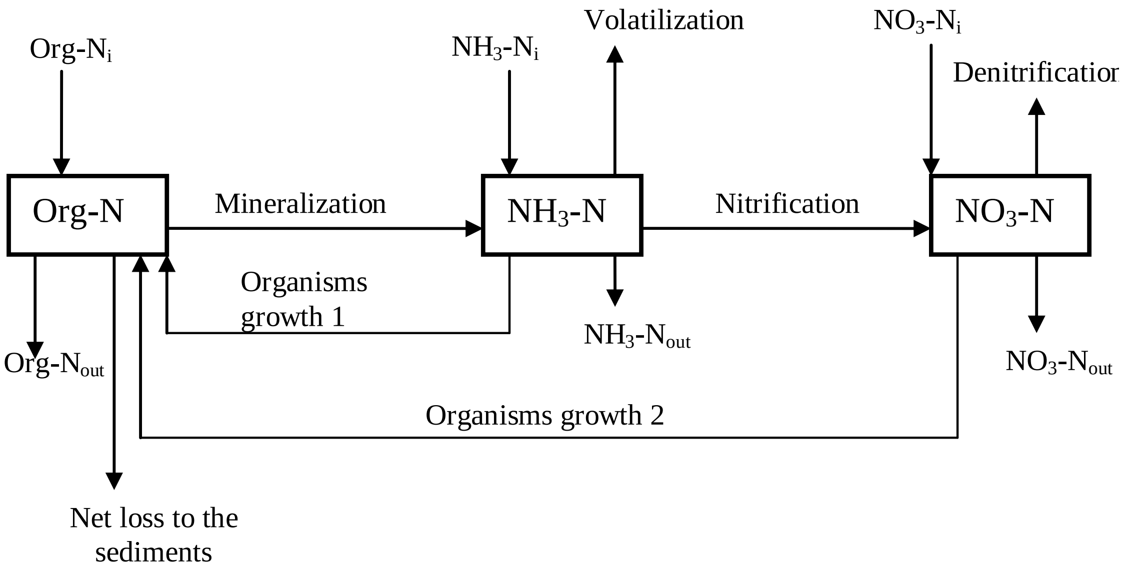

Figure 2. Nitrogen transformation and removal in WSP.

balance includes terms for substances produced or consumed in biochemical reactions, inflow, outflow and accumulation or depletion as shown in Equation 1.

$$V\sum_{j=1}^{m}\left(r_c\right)_j + Q_iC_i = Q_oC + V\frac{dC}{dt} \qquad (1)$$

Where, *C* = Effluent concentration of the substance in mg/l (e.g. NH_3-N from the pond); *V* = Reactor volume in m^3; r_c = Volumetric reaction rate of the state variable(mg/l.d); C_i = Influent concentration of the substance in mg/l (e.g. NH_3-N from raw wastewater); *m* = Number of reactions that involve the substance; *V (dC/dt)* = Volumetric rate of change of substance in the reactor (mg/l.d); Q_i = Influent flow rate in m^3/day; Q_o= Effluent flow rate in m^3/day.

With reference to the conceptual model in Figure 2, the mass balance equations for organic nitrogen (Org-N), ammonia nitrogen (NH_3-N) and nitrate nitrogen (NO_3-N) was given by Equations 2, 3 and 4, respectively.

$$\frac{d(Org-N)}{dt}=\frac{Q_i}{V}(Org-N)_i-\frac{Q_e}{V}(Org-N)-r_m-r_s+r_1+r_2 \tag{2}$$

$$\frac{d(NH_3-N)}{dt}=\frac{Q_i}{V}(NH_3-N)_i-\frac{Q_e}{V}(NH_3-N)-r_1+r_m-r_v-r_n \tag{3}$$

$$\frac{d(NO_3-N)}{dt}=\frac{Q_i}{V}(NO_3-N)_i-\frac{Q_e}{V}(NO_3-N)+r_n-r_2-r_d \tag{4}$$

Where, r_n = Nitrification rate, (mg/l.d); r_d = Denitrification rate, (mg/l.d); r_m= Mineralization rate, (mg/l.d); r_s = Net loss of organic nitrogen, (mg/l.d); r_v = Volatilization rate, (mg/l.d); r_1 = Uptake rate of NH_3-N by micro-organisms, (mg /l.d); r_2 = Uptake rate of NO_3-N by micro-organisms (mg /l.d).

Mineralization of organic nitrogen was modeled using first order kinetics with respect to organic nitrogen concentrations (Di Toro et al., 1971). Mineralization process depends on temperature and concentration of organic nitrogen and may be computed from Equation 5.

$$r_m=0.002T\times(Org-N) \tag{5}$$

The rate of nitrification, r_n which is governed by the growth of chemoautotrophic nitrifying bacteria, depends on the pH, temperature and concentration of ammonia and dissolved oxygen (Equation 6).

$$r_n=\frac{U_n}{Y_n}\left(\frac{NH_4}{K1+NH_4\_N}\right)\times\left(\frac{DO}{K2+DO}\right)\times C_T\times C_{pH} \tag{6}$$

In which C_{pH} is the *nitrosomonas* growth limiting factor for pH. Downing (1966) reported that for pH ≥ 7.2 no significant inhibition occurs and therefore C_{pH} = 1.0. When pH falls below 7.2, the existence of free ammonia inhibits growth of nitrifying bacteria. Therefore, the nitrification rate is corrected in accordance with Equation 7.

$$C_{pH}=1-0.833(7.2-pH) \tag{7}$$

The term K_1, which is half saturation constant for *nitrosomonas* is temperature-dependent (Downing, 1966) in accordance with Equation 8.

$$K_1=10^{(0.051(T-1.58))} \tag{8}$$

Nitrification is also temperature-dependent. Over the range of 5 to 30°C, the exponential model shown by Equation 9 describes the temperature correction factor.

$$C_T=e^{\alpha(T-T_o)} \tag{9}$$

Where, T_o is the reference temperature and *α* is an empirical constant. The values of T_o and *α* were 15°C and 0.098/°C, respectively.

The maximum growth rate and yield coefficient of nitrosomonas were assumed to be 0.008 day^{-1} and 0.13, respectively (Charley et al., 1980). The oxygen nitrosomonas half saturation K_2 was assumed to be 1.3 mg/l in accordance with Downing (1966).

Denitrification rate, r_d in ponds is a function of temperature and NO_3-N concentration in wastewater it is given by Equation 10 (Fritz et al., 1979).

$$r_d=R2_{20}\theta^{(T-20)}NO_3-N \tag{10}$$

The Arrhenius constant *θ* varies from 1.02 to 1.09 and denitrification constant $R2_{20}$ may vary from 0 to 1 (Bacca and Arnett, 1976).

The rate of NH_3-N volatilization depends on the concentration of ammonia gas in the liquid, temperature, depth of the pond and pH of the water. NH_3-N in water exists as dissolved ammonia gas (NH_3-$N_{(g)}$) or ammonium ions (NH_4^+).

The concentration of NH_3-$N_{(g)}$ is pH and temperature-dependent according to the Equation 11.

$$NH_3-N(g)_{conc}=\frac{NH_3-N}{1+10^{(10.5-0.032T-pH)}} \tag{11}$$

The rate of ammonia volatilization is influenced by mass transfer coefficient *KI* as shown by Equation 12 (Stratton, 1969):

$$KI=0.0566\times Exp(0.13(T-20)) \tag{12}$$

Where, *T* = water temperature in °C

The rate of NH_3-N volatilization is given by Equation 13.

$$r_v=0.0566\times Exp(0.13(T-20))\left[\frac{NH_3-N}{10^{(10.05-0.032T-pH)}}\right] \tag{13}$$

The net loss of nitrogen to the sediments depends on the concentration of organic nitrogen according to Equation 14.

Net loss, $r_s=R1(Org-N)$ (14)

Non-biodegradable organic nitrogen was assumed to settle in the sediments and thus contributed to the permanent removal of nitrogen. The coefficient *R*1 was obtained from the model calibration.

The microbial uptake was modeled using Monod kinetics. The preference factors P1 and P2 for NH_3-N and NO_3-N, respectively were introduced since NH_3-N must be depleted before nitrate can be utilized for cell synthesis (Fritz et al., 1979; Ferrara and Avci, 1982; EPA, 1985). It was assumed that each time an organism produces a new biomass, a certain amount of that material

Table 2. Optimized parameters for wastewater stabilization pond.

Parameter	Range	Reference	Optimum			
			F1	F2	F3	M
Settling rate	0.001-0.1/day	Jørgensen et al. (1991)	0.024	0.001	0.001	0.001
Nitrosomonas yield coefficient (Yn)	0.03-0.13 mg VSS/mgN	Charle et al. (1980)	0.13	0.10	0.13	0.10
Nitrosomonas maximum growth rate (Un)	0.0-0.008	Fritz et al. (1979)	0.008	0.008	0.008	0.008
Arrhenius constant θ	1.02-1.08	Najarian (1984)	1.02	1.02	1.08	1.08
Nitrosomonas half saturation constant	0.3-1.3 mg/l	Fritz et al. (1979)	1.3	0.13	1.3	0.3
Denitrification rate at 20°C (DR_20)	0.0-1.0/day	Bacca and Arnett (1976)	0.07	0.2	0.06	0.07
Microorganism maximum growth rate at 20°C ($\mu_{max, 20}$)	0.18-0.77	Ferrara and Hermann (1980)	0.18	0.18	0.45	0.77

becomes locked into a non-biodegradable form that will settle and remain in the pond sediments. NH_3-N and NO_3-N is converted to biomass in accordance with Equations 15 and 16, respectively.

Microorganisms growth 1

$$r_1 = \mu_{\max 20}\theta^{(T-20)}\left[\frac{NH_3 - N}{K_3 + NH_3 - N}\right](Org - N)\times P_1 \tag{15}$$

Microorganisms growth 2

$$r_2 = \mu_{\max 20}\theta^{(T-20)}\left[\frac{NO_3 - N}{K_4 + NO_3 - N}\right](Org - N)\times P_2 \tag{16}$$

The modeling process was carried out by using Stella II software, which was used to determine the best values of unknown coefficients, used in the model. This software has four main components that include forcing functions, state variables, mathematical equation and parameters. The model was validated using different set of data collected at the University of Dar es Salaam waste stabilization ponds (WSP).

Mass balance was done only to the ponds in route F1, F2, F3 and M. In each pond, data were processed using fourth-order Runge-Kutta approximation incorporated in Stella II software. Each pond was optimized differently. The effluent concentrations of Org-N, NH_3-N and NO_3-N was taken to be the inflow of another pond, that is, effluent concentration of pond F1 was the influent concentration of pond F2, effluent concentration of pond F2 was the influent concentration of pond F3, and effluent concentration of pond F3 was the influent concentration of pond M. The aim of considering concentration was to incorporate the loss of flow from one pond to another.

RESULTS AND DISCUSSION

Model simulation

Influent wastewater characteristics, environmental conditions and kinetic coefficients were used for simulation. The model simulation was done using data collected for 50 days. The simulated values were then compared with the observed values. A number of iterations were done to determine the optimum parameters, which would reasonably simulate the wastewater quality response of the ponds. The optimized parameter values determined from the model simulations on University of Dar es Salaam wastewater stabilization pond system are listed in Table 2.

To demonstrate successful simulation, the observed and the predicted values for organic-nitrogen and ammonia-nitrogen for primary facultative pond F1 are shown in Figures 3 and 4, respectively. The predicted values for the simulated organic nitrogen, ammonia nitrogen and nitrate nitrogen agree well with the observed values as shown by correlation coefficient ranging from 0.59 to 0.77. Therefore, the agreement between the model output and the observed effluent concentration is reasonably good given that data was collected in the field where environmental, chemical and physical factors were not controlled.

Nitrogen mass balance

Figure 5 shows daily average nitrogen mass balance in waste stabilization ponds system at the University of Dar es Salaam.

The major nitrogen transformation mechanisms in the primary pond were mineralization and uptake of ammonia, which were accounting for 39.1 and 35.4% of the total nitrogen transformed, respectively (Table 3). High rate of mineralization in the primary pond was caused by decomposition of large mass of organic material received in this pond particularly in the benthic layer.

This also explains the increase in concentration of ammonia in pond F1. It is worth mentioning that net sedimentation loss of nitrogen in the primary pond of 1413 g/day was not matched by any of the subsequent pond. In secondary facultative pond F3 and maturation pond uptake of ammonia was slightly more dominant than mineralization owing to reduced mass of organic nitrogen. Uptake of ammonia was 33.9 and 35.9% of the

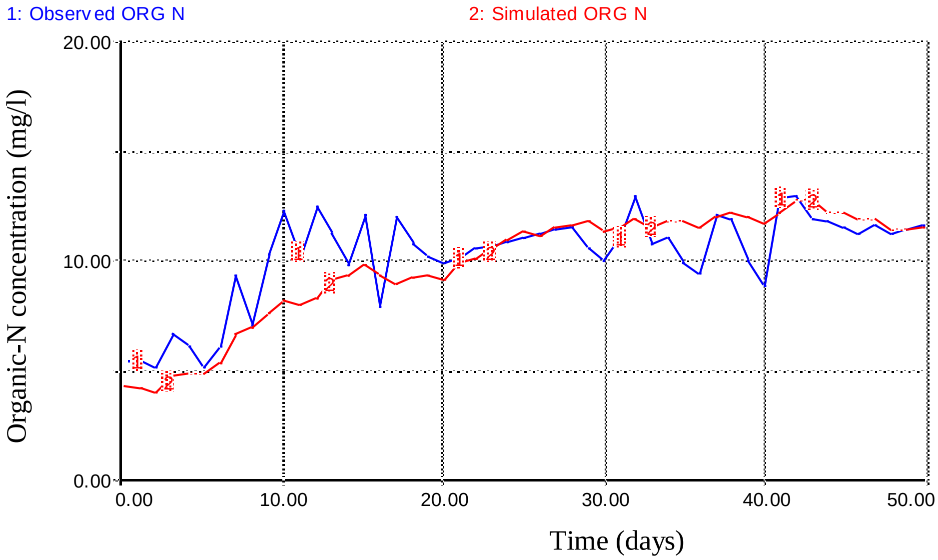

Figure 3. Organic nitrogen concentration variation with time in pond F1.

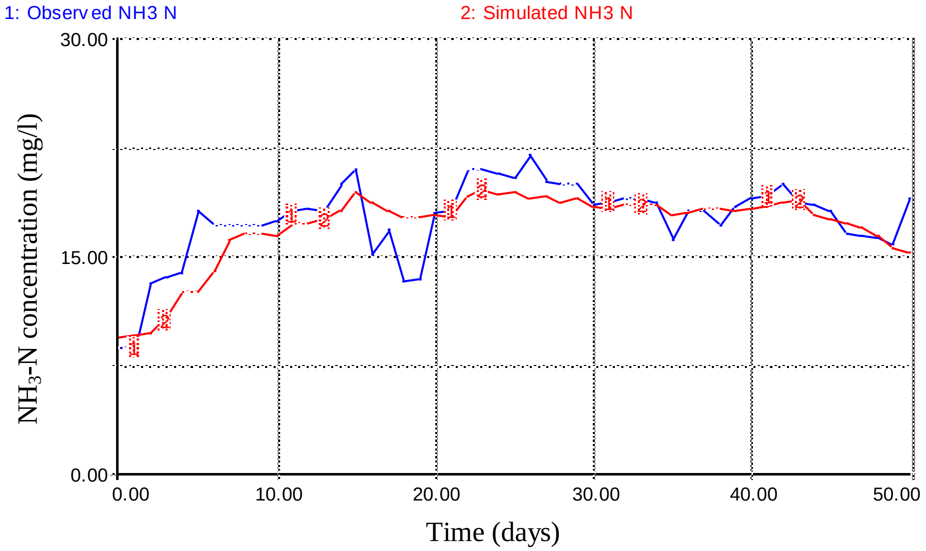

Figure 4. Ammonia nitrogen concentration variation with time in pond F1.

total nitrogen transformed in the secondary facultative pond F3 and maturation pond, respectively. These transformation rates were higher than 31.7 and 34.7% for mineralization in the same ponds, respectively.

However, in secondary facultative pond F2, nitrification and denitrification were the dominant nitrogen transformation mechanisms and were responsible for 31.3 and 26.2% of total nitrogen transformed. Increased rate of nitrification was a result of increased concentration of ammonia concentration following mineralization of organic nitrogen in primary pond F1 and high concentration of oxygen in secondary pond F2. Low nitrification rate in pond F1 was caused by the low concentration of dissolved oxygen.

Permanent nitrogen removal

It is worth mentioning that wastewater received in the primary pond F1 is divided in two series of three ponds each (Figure 1). Flow measurements indicate that pond F2 received 55.5% of the total flow and therefore the same proportion of nitrogen mass was assumed to flow in this pond. It may therefore be assumed that the mass load of nitrogen in the influent of primary pond F1

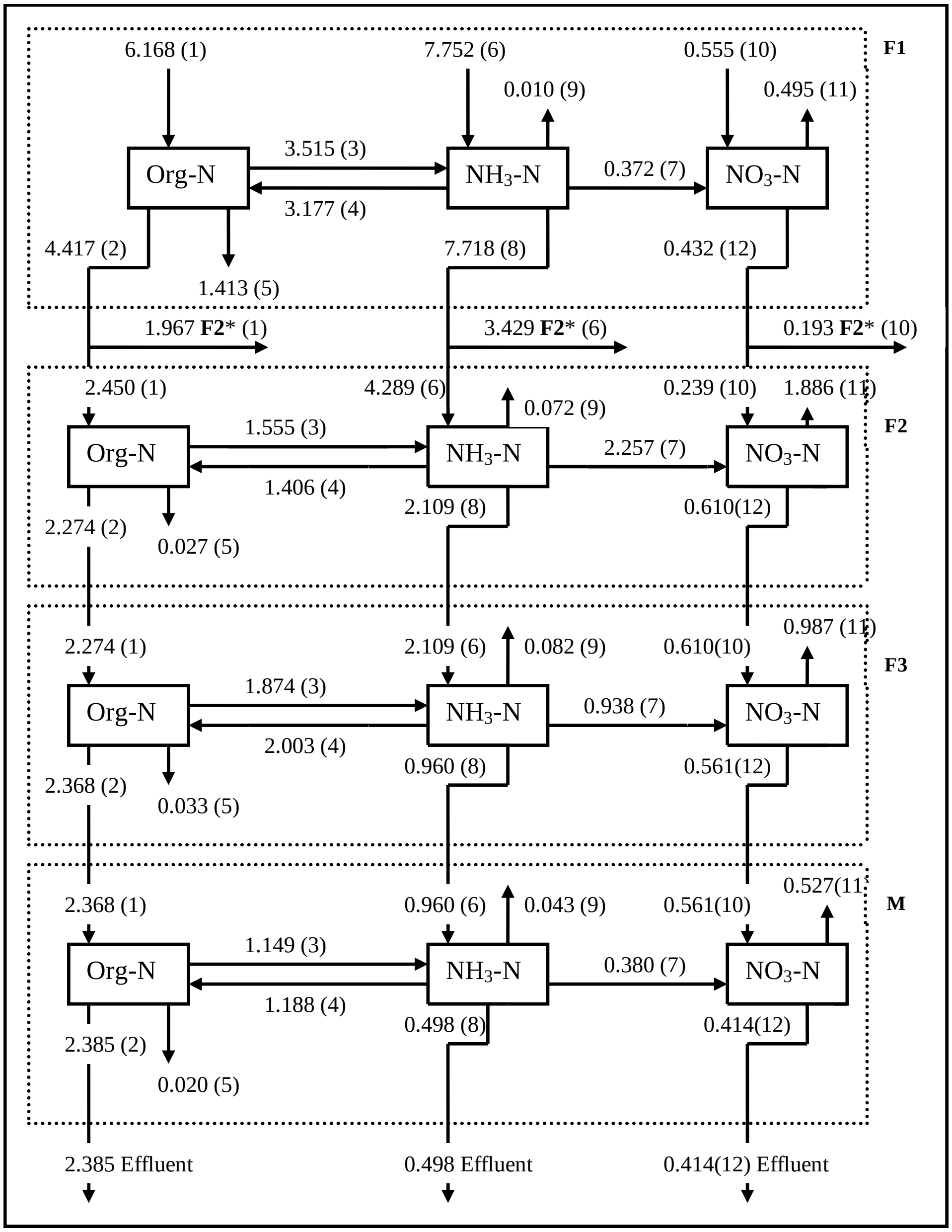

Figure 5. Nitrogen mass balance in waste stabilization ponds system. Numbers in parenthesis represents: (1) Organic-nitrogen flowing to the pond, (2) organic-nitrogen flowing out of the pond, (3) mineralization of organic-nitrogen to NH_3-N, (4) NH_3-N uptake by micro-organisms for their growth, (5) organic-nitrogen lost to the sediment by settling, (6) NH_3-N flowing to the pond, (7) nitrification of NH_3-N to NO_3-N, (8) NH_3-N flowing out of the pond, (9) volatilization of NH_3-N to NH_3-N(g), (10) NO_3-N flowing to the pond, (11) denitrification of NO_3-N to N_2(g) and (12) NO_3-N flowing out of the pond.

Table 3. Nitrogen removal mechanisms (kg/day).

Pond	Denitrification	Sedimentation	Volatilization
Primary facultative pond F1	0.274	0.784	0.006
Secondary facultative pond F2	1.886	0.027	0.072
Secondary facultative pond F3	0.987	0.033	0.082
Maturation pond M	0.527	0.020	0.043
Total	3.674	0.864	0.203

contributing to series F2, F3 and M were 3.421 kg/day of organic nitrogen, 4.308 kg/day of ammonia-nitrogen and 0.307 kg/day of nitrate-nitrogen. With the same reasoning, the proportion of net sedimentation, ammonia volatilization and denitrification will be 0.784, 0.006 and 0.274 kg/day, respectively.

In accordance with mass balance presented in Figure 5 as summarized in Table 3, the pond system was able to remove 4741 g/day of nitrogen in from inflow load of 8036 g/day, which is equivalent to nitrogen removal efficiency of 59%. This removal efficiency is within typical range of total nitrogen removal efficiency of 20 to 80% in pond systems reported elsewhere (Rockne and Brezonik, 2006; Isosaari et al., 2010). However, it is worth to note that a wide range of removal efficiency is a result of multi-factors responsible for removal of nitrogen in pond systems. These factors include hydraulic retention time, water depth (Alahmady et al., 2013), high pH levels which is responsible for removal of unionized ammonia (Rockne and Brezonik, 2006), temperature and dissolved oxygen (Fritz et al., 1979; Faleschini et al., 2012).

The overall dominant removal mechanism is denitrify-cation, which was responsible for 77.5% of the removed nitrogen followed by net loss of nitrogen to sediments (18.2%) and volatilization (4.3%). Volatilization of un-ionized ammonia was insignificant in this study, although in few studies in cold climates indicate that this mechanism was a dominant nitrogen removal mechanism during late spring when pH was above 8 and ammonia levels were still high (Rockne and Brezonik, 2006). However, other studies have reported that nitrification-denitrification is the dominant nitrogen removal mechanism (Faleschini et al., 2012; Senzia, 2003; Mkama, 2005, Muraza, 2013), which is in agreement with the results of this study. However, net loss of nitrogen to sediments was the dominant removal of nitrogen in primary facultative pond F1 accounting for 73.6% of the total nitrogen removed in that pond. It is worth mentioning that the primary pond receives large mass of suspended solids from the University community, which settled down and subsequently decomposed in the bottom benthic layer. Senzia (1999) and later Mkama (2005) observed the dominance of sedimentation in nitrogen removal in primary pond systems. In the subsequent secondary facultative ponds and maturation ponds, denitrification was the dominant nitrogen removal mechanism owing to increased nitrate content in these ponds. The dominance of denitrification as the dominant mechanism for nitrogen removal in maturation ponds has also been reported by Mtweve (1999).

Sedimentation

Owing to large quantity of particulate organic matter in the primary facultative pond F1, large quantity of organic matter was settled and removed in the primary pond as compared to other ponds. As a result 1.413 kg/day of organic-nitrogen was removed in pond F1 but only 0.027, 0.033 and 0.020 kg/day were removed in ponds F2, F3 and M, respectively because of decreased concentration of organic-nitrogen.

Volatilization

Volatilization was the weakest transformation route of nitrogen in the pond system. Facultative pond F3 had the best volatilization rate of 0.082 kg/day of ammonia-nitrogen. However, only 0.072 and 0.043 kg/day of ammonia-nitrogen were removed in secondary facultative pond F2 and maturation pond M while primary facultative pond F1 removed the lowest rate of 0.010 kg/day of ammonia-nitrogen. Volatilization rate was higher in pond F3 as compared to other ponds because of increased pH range of 8.01 - 10.03. Ammonia in water exists in equilibrium between the ionized (NH_4^+) and free (NH_3) forms. The free NH_3 form is volatile and may be lost to the atmosphere as a gas in a physical process. The equilibrium between the two forms depends on temperature and pH. At pH above 6.6, NH_4^+ starts to convert to the volatile NH_3 form; at pH 9.2 the two forms are equal in concentration, and at pH 12 all the ammonia is in NH_3 form (Reed, 1985). Consequently, loss by volatilization is more likely at higher pH. It is worth mentioning that volatilization rate was not a major removal factor in ponds at the University of Dar es Salaam because the range of 6.82 to 10.03 did not allow sufficient volatilization.

Denitrification

Denitrification rate of 1.886 kg/day of nitrate-nitrogen in the secondary facultative pond, F2 was the highest as compared to 0.987, 0.527 and 0.495 kg/day of nitrate-nitrogen in ponds F3, M and F1, respectively. Denitrification is normally limited to the amount of nitrate produced and dissolved oxygen present in wastewater. In a case like this denitrification should take place especially in the sediment zone, anaerobic layer or during the night when dissolved oxygen has been consumed for respiration by algae and bacteria (Ferrara and Avci, 1982).

Mineralization

Mineralization rate of organic-nitrogen was 3.515 kg/day in the primary facultative pond F1, which is higher than 1.555, 1.874 and 1.149 kg/day observed in ponds F2, F3 and M, respectively. High mineralization rate in pond F1 was due to decomposition of organic nitrogen introduced via wastewater influent which is readily converted to ammonia. This explains the increase of ammonia concentration in the primary facultative pond F1.

Nitrification

Nitrification rate in the secondary facultative pond F2 was the highest transformation route as compared to the other ponds, which transformed 2.257 kg/day of ammonia-nitrogen in the pond. This pond had a favorable nitrification rate because of high concentration of dissolved oxygen, optimum temperature for the growth of nitrifying bacteria, which was ranging between 27.0 and 32.2°C and favorable pH value ranging between 7.0 and 7.95, which is optimum for nitrosomonas bacteria. Nitrification rate decreased to 0.938 and 0.380 kg/day in secondary facultative pond F3 and maturation pond M, respectively because of decrease in concentration of ammonia and increase of pH, although dissolved oxygen remained high. Low nitrification rate in pond F1 was caused by low concentration of dissolved oxygen and inadequate time for *nitrosomonas* and *nitrobacter* to act upon ammonia-nitrogen.

Ammonia uptake

The uptake of ammonia by microorganisms was highest in the primary facultative pond F1 and 3.177 kg/day was consumed. This was because of preference for ammonia as a source of nitrogen for cellular growth by microorganisms (Konig et al., 1987) and high concentration of ammonia in the pond. Secondary facultative ponds (F2 and F3) and maturation pond M transformed through this route: 1.406, 2.003 and 1.188 kg/day of ammonia-nitrogen, respectively. Algal nitrogen uptake is known to be a significant contributor to nitrogen removal in waste stabilization ponds (Faleschini et al., 2012; Senzia et al., 2002). However, in the absence of algal removal systems such as sub-surface flow wetland systems and gravel bed filters (Senzia et al., 2004; Mayo and Mutamba, 2005), nitrogen stored in algal biomass will be discharged back to the environment.

Conclusions

Based on the results presented, it is concluded that:

1. In a primary facultative pond, mineralization and ammonia uptake were the major forms of transformation, which was responsible for 39.1 and 35.4%, respectively, of the total nitrogen transformed. However, in secondary facultative pond (F2), contribution of nitrification and denitrification were the dominant transformation mechanisms accounting for 31.3 and 26.2% of the total nitrogen transformed. In secondary facultative pond F3 and maturation pond uptake of ammonia became the major transformation route responsible for 33.9 and 35.9% of the total nitrogen transformed. In these ponds, mineralization was the second major transformation route.
2. The major permanent nitrogen removal mechanism in primary facultative pond F1 was sedimentation, which was responsible for 73.7% of total nitrogen removal. Denitrification was the major removal mechanism in secondary facultative ponds (F2 and F3) and maturation pond (M), which accounted for 95.0, 89.4 and 89.1%, respectively of total nitrogen removed in the ponds.
3. The effluent in maturation pond, M contains high concentration of organic nitrogen associated with particulate matter such as algae. It is recommended to remove suspended solids in the final effluent using gravel bed media or subsurface flow constructed wetlands.

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