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Removal mechanisms of nitrogen in waste stabilization ponds

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ABSTRACT

The aim of this research work was to determine the major nitrogen transformation and removal mechanisms in primary and maturation ponds. To accomplished this objective, nitrogen mass balance in waste stabilization pond system 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 nitrogen removal efficiency in a primary facultative pond unit was 13.2%, which was largely due to net loss of organic nitrogen to sediments (9.76%) and denitrification (3.42%). On the other hand, maturation pond removed 15.2% of nitrogen received in the influent with denitrification (13.55%) being the major pathway for nitrogen removal. Ammonia volatilization was not a predominant mechanism for nitrogen removal in both primary facultative and maturation ponds. The major nitrogen transformation routes were mineralization and ammonia uptake in the primary facultative pond, but ammonia uptake by microorganisms was a predominant nitrogen transformation mechanism in maturation pond.

1. Introduction

The presence of elevated concentrations of nitrogen in wastewater effluent is undesirable because it causes ecological imbalance and can affect public health (McCasland et al., 2012; http://ammoniabmp.colostate.edu, 2013). Ammonia is known to be extremely toxic to fish and many other aquatic organisms and it is also an oxygen consuming compound, which can deplete the concentration of dissolved oxygen in water (Joseph and Richard, 2006; Lee and Jones-Lee, 2007). The depletion of concentration of dissolved oxygen in water is a problem in aquatic ecosystem since maintenance of a high oxygen concentration is crucial for survival of the higher life forms in aquatic ecosystem (Orji et al., 2011; Saeed and Al-Nagaawy, 2013). 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 (Anderson et al., 2002; Showers et al., 2006), which may result in production of phycotoxins (Heisler et al., 2008; Ongore et al., 2013). Eutrophication contributes to the reduction of the oxygen level in water and disrupts natural balance of the ecosystem (Orji et al., 2011).

Excessive concentrations of Nitrate is a potential public health hazard in water consumed by infants (WHO, 2011; Kendall, 2014). In the body, nitrate is converted to nitrite, which can convert hemoglobin to *methemoglobin* by bacterial contaminants or bacteria present in the digestive system (Shearer et al., 1972). *Methemoglobin* binds oxygen less effectively than normal *hemoglobin*. The resulting decrease in oxygen levels in young children can cause *methemoglobinaemia* and can 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 becomes important before effluent is finally discharged into receiving water bodies.

To remove nitrogen from wastewater, it is important to identify different processes taking place in wastewater and how much they contribute to the transformation and removal of nitrogen. Numerous literature studies are available worldwide on nitrogen transformation and removal in aquatic systems and soil (Fritz et al., 1979; Reddy, 1983; Senzia et al., 2002; Vymazal, 2010; Mayo, 2013; Nakibuule, 2013; Muraza, 2013). These environments include algae-based waste stabilization ponds (Senzia et al., 2002, 2003; Zimmo et al., 2003; Valero and Mara, 2007; Yi et al., 2009; Mayo, 2013), attached growth systems (Shin and Polprasert, 1988), estuarine (Najarian, 1984; Dettmann, 2001; Palanisamy et al., 2007), constructed wetlands (Senzia, 2003; Senzia et al., 2004; Bigambo and Mayo, 2005; Vymazal, 2007; Nakibuule, 2013) and gravel bed systems (Manyama, 2005; Mutamba, 2002). Others are duckweed systems (Zimmo et al., 2003; Bal Krishna and Polprasert, 2008), natural wetlands (Muraza, 2013; Mayo et al., 2013; Zhao et al., 2013; Muraza et al., 2013), water hyacinth ponds (Dallah, 2001; Mutamba, 2002; Hanai, 2006) and high rate ponds (Mayo and Mutamba,

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2005; Hanai, 2006; Mayo and Hanai, 2014). However, environmental and ecological conditions in different ecosystems result in variable nitrogen transformation rates. Therefore, the objective of the research was to determine the major nitrogen transformation and removal mechanisms in primary and maturation ponds.

2. Methodology

2.1. 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/d mainly of domestic nature and also some chemical wastes flows from laboratories and workshops (Mayo, 2013). Fig. 1 shows a schematic layout of the pond system at the University of Dar es Salaam. The operational characteristics of waste stabilization ponds are shown in Table 1.

2.2. Data collection

In this study, samples were collected from the field and tested *in-situ* or in the laboratory in accordance with American Public Health Association et al. (2012). The data were collected daily over a period of 50 days. Samples for physico-chemical analysis were collected from primary facultative and tertiary (maturation) ponds of the University of Dar es Salaam waste stabilization ponds system. Water quality parameters measured include organic-nitrogen, ammonia-nitrogen and nitrate-nitrogen. Other parameters measured were flow rate, temperature, pH, dissolved oxygen (DO), and Total Kjeldahl Nitrogen (TKN). Temperature and pH were measured *in situ*. The pH was measured by a calibrated pH meter of Testo GmbH & Co. D-79849. Nitrate-nitrogen (NO_3-N) and ammonia-nitrogen (NH_3-N) were analyzed using Spectrophotometer with Cadmium Reduction and Turbidimetric methods, respectively. Nitra Ver.5 and Nessler reagents were used for NO_3^--N and NH_3-N analysis, respectively. NO_2^--N was analyzed using Calorimetric method with a spectrophotometry machine, UV-2001, Total Nitrogen (TN) was analyzed using the per sulfate digestion method and TKN was analyzed using the Semi-Micro Kjeldahl method.

2.3. Development of mathematical model

Nitrogen transformation in waste stabilization ponds was developed using conceptual model shown in Fig. 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. A complete nitrogen mass balance includes terms for substances produced or consumed in biochemical reactions, inflow, outflow and accumulation or depletion as shown by Eq. (1). This equation was used for all three forcing functions namely organic-nitrogen, nitrate-nitrogen and ammonia-nitrogen.

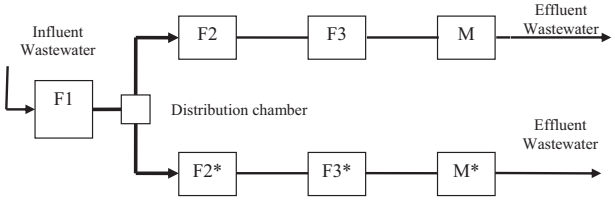

Fig. 1. The layout of waste stabilization pond system at the UDSM.

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

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/d; Q_o = Effluent flow rate in m^3/d.

With reference to the conceptual model Fig. 2, the mass balance equations for organic nitrogen (Org-N), ammonia nitrogen (NH_3-N) and nitrate nitrogen (NO_3-N) is given by Eqs. (2)–(4), respectively.

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

$$\frac{d(\text{NH}_3\text{-N})}{dt} = \frac{Q_i}{V}(\text{NH}_3\text{-N})_i - \frac{Q_e}{V}(\text{NH}_3\text{-N}) - r_1 + r_m - r_v - r_n \quad (3)$$

$$\frac{d(\text{NO}_3\text{-N})}{dt} = \frac{Q_i}{V}(\text{NO}_3\text{-N})_i - \frac{Q_e}{V}(\text{NO}_3\text{-N}) + r_n - r_2 - r_d \quad (4)$$

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 (Senzia et al., 2002). Mineralization process depends on temperature and concentration of organic nitrogen and may be computed from Eq. (5).

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

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 (Eq. (6)). It has been reported in the literature that *Nitrosomonas* yield coefficient varies from 0.03 to 0.13 mg VSS/mg N (Charley et al., 1980) and *Nitrosomonas* maximum growth rate of about 0.008 d^{-1} (Fritz et al., 1979). The oxygen *Nitrosomonas* half saturation K_2 was assumed to be varying between 0.13 to 1.5 mg/l in accordance with Jørgensen et al. (1991) and Mayo and Mutamba (2005).

$$r_n = \frac{U_n}{Y_n}\left(\frac{\text{NH}_4}{K1 + \text{NH}_4\text{-N}}\right) \times \left(\frac{\text{DO}}{K2 + \text{DO}}\right) \times C_T \times C_{\text{pH}} \quad (6)$$

In which C_{pH} is the *Nitrosomonas* growth limiting factor for pH. Downing (1966) reported that for pH $\geqslant$ 7.2 no significant inhibition occurs and therefore $C_{\text{pH}} = 1.0$. 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 Eq. (7).

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

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

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

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

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

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.