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Review article

BIOLOGICAL NITRIFICATION OF WASTE WATER

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ABSTRACT: Nitrification has been studied extensively as a result of its significance within the biological process and at intervals the necessity for treatment of waste water. In the last decade, the treatment of high ammonical concentration effluents has become a matter of nice interest. Many effluents will contain some hundred milligrams of nitrogen per liter (supernatants from anaerobic digestion, lechates from municipal water, etc.) may have specific treatment before utilization them to the plant recycling process. Sometimes this reaction is applied by maintaining robust ammonical concentrations which have the role of inhibiting the nitrite – oxidizing population responsible for the reaction of nitrites into nitrates (final stage of nitrification). However the nitrification methods served as a very important basis for the development of today understands and mathematical models for several waste treatment processes (activated sludge process using biofilm reactors) and self – purification in rivers. Often nitrogen removal from sea wastewater is problematic due to the low rate of bacteria concerned. Immobilization is an economical technique to retain slow growing organisms in continuous flow reactors. Immobilized cells can be classified into "naturally" attached cells (biofilms) and "artificially" immobilized cells. The simultaneous nitrification and denitrification within the step feeding biological nitrogen removal method were investigated below different inflowing substrate and aeration flow rates. The experimental results showed that there was additionally linear relationship between simultaneous nitrification and denitrification and DO concentration below the conditions of low and high aeration rate.

Key words: Nitrification; Biofilms; Wastewater; Immobilization; *Nitrosomonas*; *Nitrobacter*.

INTRODUCTION

Now a day's ammonia is that the most typically occurring nitrogenous waste product in wastewater. Ammonical nitrogen reaches surface and causes pollution. Sources of ammonical water will be underground water from sewage, agricultural and industrial sources. The industrial waste of this include: oil refineries, coal gasification plants, dairy plants, distilleries, fertilizer plants, pharmaceutical plants, glass production plants, cellulose and paper production plants. Ammonia exists in solution in two forms: NH_3 and NH_4^+. Though both forms are also harmful, unionized ammonia (NH_3) is of most harmful at low concentrations of the aqueous solution (Meade.J.W, 1985). For given total ammonia (NH_3 and NH_4^+) concentration in an aqueous solution depends on temperature, pH and salinity (Trusell.R.P, 1972). Lower pH and temperatures increase the percentage of ionized ammonia (Huguenin.J.E, and Colt.J, 1989). Thus, the total ammonia nitrogen (TAN) concentration instead of ammonia nitrogen is often used as a key limiting factor of water quality parameter in intensive aquaculture systems design and operation (Losordo.T.M, and Westers.H, 1994). The possible solution to the problem of ammonia removal is nitrification, which is a component of biological wastewater treatment.

Nitrification

In nitrification process, ammonia is first oxidized into nitrite (NO_2^-) by several genera of autotrophic bacteria, the most important being *Nitrosomonas*. Nitrite is then oxidized to the much less toxic nitrate (NO_3^-) by several other genera of bacteria, the most important of which is *Nitrobacter*. Eqs. (1) and (2) show the basic chemical conversions occurring in a nitrification process (WPCF, 1983; USEPA, 1984).

$$NH_4^+ + 1.5\ O_2 \longrightarrow 2H^+ + H_2O + NO_2^- \quad \text{-------- Eq (1)}$$

$$NO_2^- + 0.5\ O_2 \longrightarrow NO_3^- \quad \text{-------- Eq (2)}$$

Energy released from the above conversions is used by *Nitrosomonas* and *Nitrobacter* to drive their life processes. In addition, these reactions require oxygen, produce hydrogen ions (lowering pH) and produce nitrite as an intermediate product. According to USEPA, (1984), the complete nitrification process can be expressed as:

$$NH_4^+ + 1.83\ O_2 + 1.98\ HCO_3^- \longrightarrow 0.021\ C_5H_7O_2N + 0.98\ NO_3^- + 1.041\ H_2O + 1.88\ H_2CO_3^- \quad \text{----- Eq (3)}$$

For every gram of TAN oxidized to nitrate nitrogen, approximately 4.18 g of oxygen (or 4.57 g according to Losordo.T.M, Westers.H, (1994)) and 7.07 g of alkalinity (as $CaCO_3$) are consumed and 0.17 g of bacteria biomass is produced.

Before parameters like BOD, COD and organic carbon were used to choose the potency of wastewater treatment system, a high degree of nitrification during a secondary effluent was assumed to be an indicator of a well – treated waste material (Gujer.W, 1974). The introduction of the chemostat by Monod.J, (1950), and Novick.A, Szilard.L, (1950) set ground for the understanding and mathematical modeling of microbial culture systems.

Garrett.M.T, (1958) seems to be the first author who related microbial growth to the activated sludge process. A substantial step in understanding nitrification in the activated sludge process is due to a research group at the British Water Pollution Research Laboratory (Water Pollution Research, 1964). For the treatment of wastewater, nitrification with autotrophic bacteria has received most attention. It has been shown that under these conditions autotrophic nitrification is favorable and rates are orders of magnitude higher than those of heterotrophic bacteria.

Biological nitrification

Biological nitrification will be accomplished in two types of systems: suspended and attached growth. Under a suspended growth environment, the organism is freely mobile within the liquid providing direct contact between the microorganism cells and therefore the bulk water. In attached growth system, microorganisms had grown up on a visco – elastic layer of biofilm that are attached on the surface of a solid support medium. Thus, this process is termed a fixed film process in which the individual microorganisms are immobilized. attached growth on a fixed biofilm system offers many benefits when compared to suspended growth processes, such as handling convenience, increasing process stability in terms of withstanding shock loading and preventing the microorganism population from being washed off (Fitch.M.W, 1998; Nogueira.R, et.al., 1988) and handling convience.

Immobilization is an efficient method to retain slow growing organisms in continuous – flow reactors. Immobilized cells can be divided into “naturally” attached cells (biofilms) (Denac.M, 1983; Harremoes.P, 1982; LaMotta.E.J, 1976) and “artificially” immobilized cells (Dalili.M, and Chau.P.C, 9187). The residence time of the liquid phase in these systems may be chosen independently of the specific growth rate; as a result, immobilized – cell reactors are compact in comparison to activated – sludge plants.

Naturally attached cells (biofilms)

The trickling filter is the mostly widely applied immobilized cell system. A trickling filter is a percolating filter consisting of a bed with a porous support, on which a biofilm develops. In the first instance, the support materials used were lava and stone, which have relatively limited external specific areas (100 $m^2\ m^{-3}$). In the second generation of trickling filters, plastic media were used with a specific area of 100 – 300 $m^2\ m^{-3}$. Although the specific area of trickling filters is still small, the system is widely used because in practice there is much experience with them.

A system with comparable specific surface area is the rotating biologic contactor, in which the biofilm is attached to discs. A large number of these discs are closely arranged and mounted on a shaft, which rotates in the reactor. The discs are partly submerged in the sewage. As the shaft rotates slowly, attached biomass is alternatively exposed to air and sewage. Both trickling filters and rotating biological contactors have a limited capacity because of the relatively low specific surface area of the support. In the case of fluidized – bed reactors, increased oxygen transfer rates can be reached by sparging air in the column, creating a three – phase system in the reactor (Black.G.M, 1986; Denac.M, et.al., 1983; Focht.D.D and Verstraeta.W, 1977; Tanaka.H, et.al., 1981; Venkatasubramanian.K, et.al., 1983). Similar capacities can be reached in fluidized – bed systems with a draft loop reactors which have a more defined liquid flow (Woodward.J, 1988).

Artificially immobilized cells

Immobilized – cell reactors with naturally attached biomass are controllable to a limited extent, although underlying mechanisms are not very well understood. A better defined and more controllable system is obtained by artificial immobilization of pure strains of bacteria. One of the most common techniques for artificial immobilization is gel entrapment. Materials that are widely applied for entrapment are alginate and carrageenan. For this, solutions of polymers with cells suspended in them are extruded drop wise into a solution in which gelation of drops into solid spheres is initiated. In the case of alginate and carrageenan, gelation is initiated by Ca^{+2} and K^{+} respectively (Woodward.J, 1988).

Nitrification Kinetics

The rate of ammonia or nitrite oxidation depends mainly on the concentrations of those substrates within the bulk solution. During a pure culture under a single limiting – substrate condition, the steady state kinetics of substrate removal is typically represented by the Monod – type expression (Drtil.M, et.al., 1993; Rittmann.B.E and McCarty.P.L, 1980; Srna.R.F and Baggaley.A, 1975):

$$R = \mu_{max} \frac{X}{Y_S} \frac{S}{K_S + S} \quad \text{---------- Eq (4)}$$

Where R = substrate removal rate (g m^{-3} day^{-1})
μ_{max} = maximum specific growth rate (day^{-1})
X = bacterial mass concentration (g cell m^{-3})
Y_S = yield of bacterial mass per unit of substrate used (g cell g^{-1} substrate $^{-1}$)
S = limiting substrate concentration (g m^{-3})
K_S = half saturation constant (g m^{-3}).

This equation may be used to describe nitrification kinetics when ammonia is assumed as the growth – limiting substrate for *Nitrosomonas* while nitrite as the limiting substrate for N*itrobacter*. It absolutely was reported that the growth rate of *Nitrobacter* is bigger than that of *Nitrosomonas* (WPCF, 1983) and oxidation of ammonia is typically the rate – limiting step within the conversion of ammonia to nitrate. Thus, in Eq (4), values for ammonia oxidation are the rate limiting parameters in describing nitrification (Wheaton.F.W, et.al., 1994). Both mathematical analysis and experimental information observed two major characteristics of Eq (4).

Nitrification in the bacterial film of the biofilter involves physical, chemical and biological processes that are governed by a variety of parameters such as substrate and dissolved oxygen concentrations, temperature, pH, alkalinity, salinity and turbulence level.

Substrate Concentration

The concentration of total ammonia nitrogen (TAN) as the substrate of nitrification is the most significant factor to consider within the design and operation. The best water quality, in terms of ammonia, is defined by a minimum substrate concentration that a biofilter will operate a sustainable basis, S_{min}. The concept of a minimum substrate concentration required to support a steady state biofilm was proposed and proved by Rittmann.B.E and McCarty.P.L, (1980) and Rittmann.B.E and Manem.J.A, (1992). Rittmann.B.E and McCarty.P.L, (1980), also mathematically defined the S_{min} for a biofilm as:

$$S_{min} = K_S \frac{b}{\mu_{max} - b} \quad \text{---------- Eq (5)}$$

Where S_{min} is the minimum substrate concentration (mg L^{-1}) and b is the specific bacterial decay rate (day^{-1}).

Zhu.S and Chen.S, (1999) evaluated the minimum TAN concentration for submerged nitrification biofilters in a reactor series system and therefore the mean value of the minimum TAN concentration was found to be 0.07 ± 0.05 mg L^{-1} at 27.2 ^{o}C. At low concentrations (S << K_S), the nitrification kinetics may be simplified into a first – order reaction model:

$$R = \frac{R_{max}}{K_S} (S - S_{min}) \quad \text{---------- Eq (6)}$$

Where R = substrate oxidation rate (g m^{-2} day^{-1})
R_{max} = maximum substrate oxidation rate (g m^{-2} day^{-1})
S = limiting substrate concentration (mg L^{-1} or g m^{-3})
K_S = half saturation constant (mg L^{-1} or g m^{-1}).

Eq (6) shows that nitrification rates increase linearly with increase of TAN substrate concentration. This relationship has been confirmed with experimental aquaculture systems. Ester.C.C, et.al., (1994) studied the performance of three rotating biological contactor (RBC) systems used for RAS wherever water temperature was varied from 24 to 30 ^{o}C and determined first – order nitrification kinetics at low concentrations. Different researchers (Surampalli.R, et.al., 1989; Watanabe.Y, et.al., 1980) have also found that a first – order reaction can be developed for RBC reactors at very low ammonia concentrations and low organic loading rates. Liu.Y and Capdeville.B, (1994) also developed a linear relationship between influent ammonia concentration and ammonia removal rate in RBC.

Dissolved oxygen

The relationship between nitrification rate and DO concentration had major interest in nitrification. As demonstrated in Eq (3), oxygen may be a requirement in ammonia oxidation. The theoretical oxygen needs in step with the nitrification stoichiometric equations are: 3.43 mg for oxidation of 1 mg NH_3 – N and 1.14 mg for oxidation of 1 mg NO_2 – N, though a rather lower ratio of oxygen consumed to nitrogen oxidized in an experimental study was also reported by Sharma.B and Ahlert.R.C, (1977) and Wezernak.C.T and Gannon.J.J, (1967).

The effects of the DO concentration on the nitrification rates had been reviewed by many authors (Beccari.M, et.al., 1992; Painter. H.A, 1986; Sharma.B and Ahlert.R.C, 1977; Stenstrom.M and Poduska.R, 1980) in both attached and suspended growth systems. Wuhrman.K (1963) reported the optimum DO of 4 mg L^{-1} for max nitrification rate in activated sludge and most of the experiments considered to be the limitation of low DO on nitrification. Zhu.S and Chen.S, (2002) reported that it absolutely was more important to maintain sufficient do in the fixed film process than within the suspended growth process as a result of the character of diffusion transport with fixed film. DO concentration profiles at intervals biofilms were studied using a micro technique and a microslicing technique with heterotrophic biofilms, heterotrophic – autotrophic biofilms and nitrifying biofilms was reported by Zhang.T.C, et.al., (1995).

Temperature

It was well accepted that a better temperature enhances nitrification rate because the biochemical driven microorganism processes accelerate as temperature increases. This can be true in a suspended growth system. For fixed film filters, however, the results of temperature on nitrification kinetics are also influenced by different phenomena and parameters was explained by Fdz-Polanco.F, (1994), especially substrate diffusion and transport. A general conclusion on the relationship between nitrification rate and temperature must also include the effect of mass transfer and microorganism. However, the impacts of change on nitrification rate in fixed film biofilters were poorly understood by Okey.R.W and Albertson.O.E, 1989. Very little data is available to quantify the results of temperature on fixed film nitrification rate (Wheaton.F.W, et.al., 1994).

Zhu.S and Chen.S, (2002) studied the impact of temperature on nitrification rate through laboratory experiments, mathematical modeling and sensitivity analysis. They (Zhu.S and Chen.S, 2002) showed that in the case of oxygen limitation, temperatures from 14 to 27 ^{o}C had no significant impact on nitrification rate. A lower nitrification rate was observed only at the lowest temperature they tested, 8 ^{o}C. Temperature had a more significant effect on nitrification rate within the case of TAN limitation than within the case of DO limitation.

pH

A great deal of investigations conducted has demonstrated the ph effects on nitrification. However, poor agreement existed on how much, and what point, ph begins to effect nitrification rates (Biesterfeld.S, et.al., 2001). Based on the review provided by Sharma.B and Ahlert.R.C, (1977) and studies by other researchers, the optimal pH for the growth of nitrifying microorganism varies wide. The optimum ph for nitrification will vary from 7.0 to 9.0 with the optimum ph vary from 7.2 to 8.8 for Nitrosomonas and 7.2 to 9.0 for Nitrobacter. Based on the ability of free ammonia (NH_3) and free nitrous acid (HNO_2) to penetrate the nitrifying organism, Anthonisen. A.C, (1974) reported that both NH_3 and HNO_2 were inhibitory to nitrifying bacteria than ammonia and nitrite ions. Moreover, Suzuki.I, et.al., (1974) and Painter.H.A, (1986) recommended that free ammonia rather than ammonium ion is the substrate for ammonia – oxidizing bacteria (*Nitrosomonas*) supported the observation of an identical Monod saturation constant under variable free ammonia concentration.

Therefore, reduced nitrification activity at lower pH levels may result indirectly from substrate limitation since the fraction of NH_3 – N in the total ammonia nitrogen decreases with decrease of pH (Allison.S.M and Prosser.J.I, 1993). When a higher TAN is used, a higher non – limiting concentration of NH_3 may be maintained at lower pH values (Biesterfeld.S, et.al., 2001). Interestingly, to evaluate the pH effect on ammonia oxidation activity, Groeneweg.J, et al., (1994) measured ammonia oxidization rates at a constant NH_3 – N of 0.37 mg L^{-1} (varying TAN in accordance with pH) and a constant TAN of 5 mg L^{-1} (NH_3 – N varies with pH) over a wide pH range (5 – 11).

They found that the maximum ammonia oxidization rate was obtained between pH 6.7 and 7.0 (0.37 mg L^{-1} NH_3 – N) and pH 7.5 and 8.0 (5 mg L^{-1}TAN), while the ammonia oxidization rate decreased sharply outside the optimum pH ranges.

Alkalinity

Alkalinity effects the conversion of ammonia to nitrate as seen from Eq (3). In fact, pH alkalinity type of carbonate and hydrogen carbonate may be a nutrient component for nitrifying bacteria. Additionally, alkalinity provides the buffering capability that is necessary to prevent pH changes because of acid production within the nitrification process. Therefore, the impact of alkalinity on the nitrification rate is additionally related to that of the pH. During a study on the pH effect upon the efficiency in an upflow biofilter, it was reported that the nitrification efficiency showed a linear increase of 13 % per unit pH increase from pH 5.0 to 8.5 (Villaverde.S, et.al., 1997). The same authors additionally investigated the relationship between pH and alkalinity. They determined a linear correlation between the alkalinity (as mg $CaCO_3$ L^{-1}) and pH, with a stoichiometry coefficient of 7.1 mg $CaCO_3$ consumed/ mg NH_4^+ - N oxidized. Chen.G.H, et al., (1989) showed that the rate of nitrification would be reduced when pH was below 40 g m^{-3}. Gujer.W and Boller.M, (1986) according that in nitrifying biofilters utilized in municipal waste water treatment, an alkalinity level of at least 75 mg L^{-1} (g m^{-3} or 1.5 meq/L) was needed to maintain maximum nitrification rate. Considering possible stratification of alkalinity and pH in a biofilm, a better alkalinity concentration of 200 mg L^{-1} is suggested especially for the applications wherever the water exchange rate was minimum.

Salinity

Less information is available regarding the effect of salinity on nitrification kinetics. There are discrepancies within the reports, most likely due to different experimental conditions. Nijhof.M and Bovendeur.J, (1990) compared the nitrification characteristics of salt water with that of fresh water systems. The results indicated that the maximum nitrification capacity within the salt water systems was significantly less than in fresh water systems. At 24 ^oC, a maximum ammonia removal rate of 0.28 g m^{-2} day^{-1} NH_4^+ - N was determined versus 0.69 g m^{-2} day^{-1} NH_4^+ - N in comparable fresh water systems. During a separate laboratory study, Saucier.B, (1999) was able to get a sufficient nitrification rate that is comparable with the reported result in fresh water systems under similar conditions considered by Zhu.S and Chen.S, (2002).

Turbulence

The significance of the impact of turbulence on nitrification rate has been demonstrated by different researchers. Kugaprasatham.S, et al., (1991) studied the impact of hydraulic conditions on nitrifying biofilm grown under a low ammonia nitrogen concentration (about 1 gm-3) in a cylindrical reactor. When turbulence intensity was changed and kept at the new value for many days, filamentous – type biofilm with higher substrate flux was determined at high turbulent intensities, but colony – type biofilm under low turbulent intensities showed reduced mass transfer (Kugaprasatham.S, et al., 1991). Additionally, Chen.G.H and HuangJ.C, (1996) found that chemical oxygen demand reported higher nitrification rates in biofilters with high turbulence levels. These results are important for the design and optimal operation of biofilters, as they suggest that the nitrification rate could also be significantly improved through increasing turbulence.

Rasool.K, et al. (2014) reported the high removal efficiencies of organic matter of about 97% as total COD and more than 99% removal of ammonia-nitrogen with Synthetic wastewater with average loading rates of 0.53 $kgCOD/m^3.d$ and 0.067 $kgNH_4^+$-$N/m^3.d$ was fed to the reactor system at hydraulic residence times (HRT) of 24 and 18 h and operated for 100 days in a bench-scale anoxic–oxic activated sludge system for integrated removal of COD and nitrogen Wan.C, et al. (2014) reported on the partial nitrification performances for granules as nitrite accumulation rate >95% and chemical oxygen demand (COD) removal at >85% at salt concentration up to 50 $g.L^{-1}$ using aerobic granules to conduct partial nitrification reactions for wastewater with high NaCl concentrations in a continuous-flow reactor. Wang.L et al. (2014) reported that the results indicate partial nitrification of landfill leachate could be successfully achieved under the 1.0-2.0 $mg.L^{-1}$ dissolved oxygen (DO) condition after 118 d long-term operation, and that the effluent is suitable for an Anammox reactor. Further decreasing or increasing the DO concentration, however, would lead to a decay of nitrification performance on a coupled system of partial nitrification and anaerobic ammonium oxidation (Anammox) is efficient in nitrogen removal from wastewater. Wang.B et al. (2012) reported that the removal rates of COD, NH_4^+-N and TN were 88.2%, 95.7% and 86.4% respectively in a novel four-stage step-feed wastewater treatment system combined with a fluidized bed laboratory bioreactor to investigate on chemical oxygen demand (COD), NH_4^+-N and total nitrogen (TN) removal performance. Dong.Y, et al. (2011) reported on the effects of environmental changes, such as temperature, dissolved oxygen (DO) concentration and pH, on nitrification characteristics under conditions of low ammonia concentrations using Suspended and waterborne polyurethane immobilized nitrifying bacteria. They stated that rate of nitrification increases with increasing pH, DO and temperature.

Simultaneous Nitrification and Denitrification process:

Many researchers have put much attention to this process and drawn many valuable conclusions (Larrea.L, et.al., 2001; Zhu.G.B, et.al., 2005). Moreover, nitrogen loss and simultaneous nitrification and denitrification is step feeding process were also reported by researchers (Gorgun.E, et.al., 1996; Zhu.G.B, et.al., 2007). Simultaneous nitrification and denitrification (SND) implies that nitrification and denitrification occur concurrently in the same reaction vessel under identical overall operating conditions. SND is of particular interest in saving anoxic volume and in treatment wastewater with low C:N ratio (Zhu.G.B, et.al., 2008). The mechanism and explanation for SND can be divided into two broad categories. The physical and conventional explanation is that SND occurs as a consequence of DO concentration gradients within microbial flocs or biofilms due to diffusional limitations. The biological explanations for SND are in contrast to the traditional "engineering" conception of nitrification and denitrification. Microbiologists have reported the existence of aerobic denitrifiers as well as heterotrophic nitrifiers (Kim.J.K, et.al., 2005; Zhu.G.B, et.al., 2008). Radhika.K, et al. (2013) reported that around 98.9% ammonia removal was achieved with ammonia loading rate 0.35kgNH_4^+-N/m^3.day in the presence of 46.6 mg/LCOD at 2.31 days hydraulic retention time and ambient temperature of 30 ^oC in a simultaneous partial nitrification, anammox and denitrification (SNAD) process for the treatment of ammonia effluent of a fertilizer industry.

Heterotrophic Nitrification

Mainly autotrophic nitrifiers are suitable to be responsible for nitrification process. However, nitrification was also employed during heterotrophic growth of some bacteria, such as *Thiosphaera pantotropha*, *Alcaligenes faecalis*, *Pseudomonas stutzeri*, *Diaphorobacter sp*. and *Bacillus sp*. (Su.J.J, et.al., 2001; Joo.H.S, et.al., 2005; Kim.J.K, et.al., 2005; Khardenavis.A.A, et.al., 2007). Heterotrophic nitrification was thought to be performed in a similar way to the autotrophic process: NH_4^+ is firstly converted to NH_2OH by the enzyme ammonia monooxygenase, and followed by NH_2OH oxidation to NO_2^- by the enzyme hydroxylamine oxidoreductase (HAO), and then NO_2^- is further oxidized to NO_3^-. The coupling of heterotrophic nitrification and aerobic nitrite/nitrate denitrification has been widely accepted as the result of nitrogenous gas production under aerobic conditions. (Kim.J.K, et.al., 2005; Khardenavis.A.A, et.al., 2007; Wan.C, et.al., 2011; Zhang,J, et.al., 2011). Recent studies showed that most heterotrophic-nitrifying bacteria are capable of aerobic denitrification, including *Alcaligenes faecalis* (Joo.H.S, et.al., 2007), *Pseudomonas stutzeri* (Su.J.J, et.al., 2001), *Microvirgula aerodenitrificans* (Patureau.D, et.al., 2001), *P. putida* (Kim.M, et.al., 2008), *Acinetobacter calcoaceticus* (Zhao.B, et.al., 2010a,b) and *Rhodococcus species* (Zhang.G, et.al., 2003). Bacteria capable of combined heterotrophic nitrification and aerobic denitrification have drawn increasing attention for their potential application in biological nitrogen removal system.

By considering all parameters a brief review of operating conditions is given in table 1.

Table 1: A brief review of operating conditions for the nitrification process

S.No	pH	Temp.	DO	Substrate Conc.	Alkanity	Salinity	Turbulence or Air flow rate	Time	Reference
1	8 – 9	18–25^{0}C	3–4 mg/lit	100 – 300 mg/lit as N of $(NH_4)_2CO_3$ or NH_4Cl	1 mg/lit as P of KH_2PO_4			2 days	Sheintuch.M, et.al., (1995)
2	8 – 8.3	10–35^{0}C	> 1 mg/lit	80 – 100 mg/lit	> 7 mg/lit		30 – 40 l/hr	7 days	Fdz- Polanco.F, et.al., (1994)
3	7.5	30 ^{0}C		180 – 260 mg dm^{-3} as N			0.1 l/min	2 days	Kotlar.E, et.al., (1996)
4	7.1–7.3	12–24^{0}C		25 – 70 mg/lit as N	320 – 450 mg/lit as $CaCO_3$		20 – 70 m^3/hr		Lazarova.V, et.al., (1997)
5		28^{0}C		252 mg/lit of NH_4/Cl		50 gm/lit of NaCl		15 hrs	Rosa.M.F, et.al., (1998)
6		25^{0}C	6.5 g/m^3	23.8 g/m^3 as N of $(NH_4)_2SO_4$	4.6 g/m^3 as P of KH_2PO_4		42x10^{-3} m^3/hr	9 hrs	Xiaojing.X, et.al., (1998)
7		20^{0}C		450 mg/lit as N of NH_4Cl				80 days	Campos.J.L, et.al., (1999)
8	7 – 8.5	30^{0}C	5.5 mg/lit	10 g/lit as N of NH_4Cl	80 g/lit of $NaHCO_3$			175 days	Ruiz.G, et.al., (2003)
9		23–26^{0}C	85 mg/lit	700 mg/lit			1.7 Nm^3/hr	20 days	Delgoda.S, et.al., (2002)
10	7.5 – 9	10-35^{0}C		280 mg/lit as $(NH_4)_2CO_3$				5 hrs	Benyahia.F and Polomarkaki.R, (2005)
11	8.5	15–20^{0}C	8.5 mg/lit	38 mg/lit as NH_4Cl				5 days	Ling.J and Chen.S, (2005)
12				8.7 mg/lit as N of NH_4Cl	100 -200 mg/lit of $CaCO_3$	30 g/lit of NaCl	0.66 m^3/hr	90 days	Silapakul.S, et.al., (2005)
13	7.0	35^{0}C	8.2 mg/lit	60 mg/lit as N of NH_4Cl		0 – 30 g/lit of NaCl		45 days	Mosquera-Corral.A, et.al., (2005)
14	7.5	25^{0}C		100 g /lit as N	0.02 g/lit of KH_2PO_4		2 cm/sec		Terada.A, et.al., (2006)
15	6.8 – 7.6	30^{0}C	2.5 mg/lit	79 mg/lit			28 – 60 l/hr	20 days	Yongzhen.P, et.al., (2007)
16			3.5 – 4.5 mg/lit	35 mg/lit as N	200 – 230 mg/lit as $CaCO_3$		0.47 m^3/day	2 – 3 days	Li.B and Iruin.S, (2007)

CONCLUSION

In the present paper a brief review on biological nitrification was reported. Many industries particularly fertilizer, coking, refining, food processing and organic chemicals generate wastes containing high concentrations of ammonium compounds along with varying concentrations of arsenic, chromium and fluoride. To reduce the high ammonium concentrations in nitrogenous wastewater to ecologically acceptable levels, biological nitrification is a well established method. Biological aerobic or anoxic treatment processes are much simpler and cheaper than a sequence of combined chemical-physical treatments. However, they could not achieve high and reliable ammonium and COD removal efficiencies. Aerobic treatment processes are used for reduction of BOD and COD as well as nitrification. As observed from above review biological nitrification is seen to be an economical and ecofriendly process for removing ammonium from wastewater and treated wastewater can be used for many industries as inlet water for many purposes.

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