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Water Research 39 (2005) 3773–3780

WATER RESEARCH

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Comparative economic analysis of the impacts of mean cell retention time and denitrification on aeration systems

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Received 15 March 2005; received in revised form 29 June 2005; accepted 4 July 2005
Available online 18 August 2005

Abstract

Biological nutrient removal is practiced in various modifications of the activated sludge process (ASP) throughout the world. This paper compares conventional, nitrifying-only and combined nitrifying/denitrifying (NDN) processes. The authors performed 113 oxygen transfer efficiency measurements with the off-gas method over 20 years. This dataset was analysed and used to perform an economic analysis for three example scenarios, one for each layout (conventional, nitrifying-only and NDN). Field oxygen transfer efficiency and relevant plant operative costs and credits were considered (i.e., aeration cost, sludge disposal cost, methane production credit). The conclusion is that NDN operations always have lower aeration costs, and generally have the lowest combined operating cost. Reduced aeration costs result because of improved aeration efficiency at higher mean cell retention times and the use of nitrate as an electron acceptor. The improved aeration efficiency overcomes the increased oxygen required at higher cell retention time due to cell decay.

Keywords: Aeration; Denitrification; Economics; Nitrification; Off-gas

1. Introduction

The urban areas in California have mostly separate sewers, and with few exceptions, full secondary treatment or more advanced treatment is practiced. The plants that discharge directly to the Pacific Ocean are generally designed to remove only carbon and do not remove nutrients. The inland plants were originally designed to remove only carbon, but more recently have been required to upgrade for biological nutrient removal (BNR). The motivation for removing nutrients at the inland plants is not the traditional reason of avoiding eutrophication, but often relates to water reclamation. The insensitivity of inland waters to eutrophication is the historical reason why Southern California lags the many areas in constructing treatment plants that remove nutrients. A greater fraction of the effluents from wastewater treatment plants are now being recycled, since the need for potable water is increasing and wastewater reclamation is viewed as a new source. A strict requirement for water reclamation for indirect potable use or for groundwater recharge is nitrogen removal. Therefore, all treatment plants that are candidates for reclamation are being upgraded to remove nitrogen. Phosphorus removal at the inland plants is usually not required.

There are many methods for removing nitrogen in wastewater treatment. The processes range from simple

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0043-1354/$ - see front matter
doi:10.1016/j.watres.2005.07.002

(Ludzack–Ettinger, LE) to complex (Bardenpho). Newer techniques using simultaneous nitrification–denitrification (SND) are also being evaluated (Reardon et al., 2003). The treatment requirements for the Southern California needs are not so severe as to require Bardenpho, but are generally being met with the modified Ludzack–Ettinger (MLE) process. This is advantageous because conventional carbon-only plants can be converted to the MLE process. The conversion of conventional plants creates special aeration concerns, because they usually employ fine pore diffusers. Fine pore diffusers are sensitive to operating conditions such as mean cell retention time, aeration efficiency and fouling rates. This paper focuses on the application of fine pore diffusers to conventional and BNR processes, such as the MLE process.

Originally the cost of upgrading for N removal was thought to be high, but experience has shown that the cost is less than anticipated, and there are additional benefits to upgrading for N removal. An indirect benefit is improved oxygen transfer efficiency (Fisher and Boyle, 1999), which is in addition to the well known oxygen credit provided by anoxic removal of influent substrate. This improved transfer efficiency results because of higher mean cell retention time (MCRT) operation, which has been shown to be associated with increased alpha factors (Rosso et al., 2005), but has not been well recognized by design engineers. In addition to removing nitrogen, the nitrifying–denitrifying (NDN) processes provide additional stability because the built-in anoxic zone functions as a selector (Harper and Jenkins, 2003). Additionally, the high MCRT operation is more efficient in removing biodegradable organic carbon (Khan et al., 1998), and there is mounting evidence that they are more efficient in removing anthropogenic compounds, such as pharmaceuticals (Soliman et al., 2004).

It is the objective of this paper to show the impact of the improved aeration efficiency on plant economics for an NDN process over conventional and nitrifying-only processes. Three example plants are considered and balances that quantify the cost of aeration and sludge processing are presented. Differences in capital cost, such as the possible need for increased aeration tank volume, are not considered.

2. Background and methods

The following sections describe three typical treatment designs and the operational details that are important for the economic analysis. The oxygen transfer data were all collected using off-gas analysis at full-scale treatment plants, and the methods and results are described as a function of operating conditions. Finally, oxygen requirements are calculated, and the impact of sludge processing variables is shown.

2.1. Treatment plants

Fig. 1 shows the two process configurations for the treatment plants evaluated. The top shows a conventional "plug flow" treatment which, if operated at low MCRT, removes only carbon. These layout and mode were typical for plants in Southern California before nitrogen removal was required. This layout can also represent a nitrifying-only treatment plant, when operating at higher MCRT, and may require greater tank volume or clarifier surface area for the same flow rate. The lower part of Fig. 1 shows an NDN plant, using an anoxic zone, separated from the main aeration zone by a baffle. This configuration is commonly known as the MLE process (Metcalf & Eddy, Inc., 2003), and is currently the most popular method for NDN operations in Southern California. Only one digester is shown, because it is common to both process layouts.

Over the last 20 years, 22 treatment plants were tested using the off-gas procedure. A total of 113 tests were available for this analysis. Fourteen of the plants were conventional, while three plants were operating as nitrifying-only and five plants were operating in NDN mode. Eighty-four of the tests were for conventional

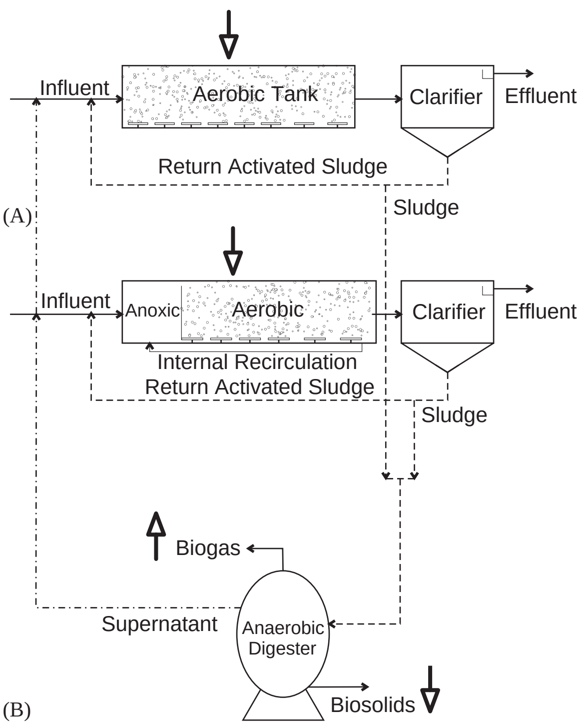

Fig. 1. Process schemes for: (A) conventional and (B) modified Ludzack–Ettinger (MLE) processes. Large arrows represent costs (downwards) and credits (upwards).

operation, nine tests were for nitrifying-only plants and 20 tests for NDN operation. The range of MCRTs for the conventional plants was 1.2 to 8.5 days. The range of MCRTs for nitrification-only was 12–21 days and for NDN was 4.7–22 days. The operating MCRTs in Southern California can be lower to achieve nitrification or NDN because wastewater temperatures are 26–28 °C in the summer and seldom below 20 °C during winter.

All plants were equipped with fine pore diffusers and included ceramic discs and domes, membrane discs, tubes and panels, and plastic discs and tubes. Equipment from 11 different manufacturers was included in our dataset. The range of diffuser ages varied from new (less than 1 month of operation) to used (within the first 24 months of operation) to old (greater than 24 months operation). Diffusers that were maintained by cleaning were classified as cleaned, if they had been cleaned within 1 month of testing. All cleaning methods were grouped together because the effects of different methods, at least within the present dataset, were too small to quantify.

2.2. Off-gas aeration efficiency testing

The technique used to test all plants, the off-gas method, was first developed by Redmon et al. (1983) in conjunction with the US Environmental Protection Agency (EPA)-sponsored American Society of Civil Engineers (ASCE) Oxygen Transfer Standards Committee. This committee produced a fine pore manual (United States Environmental Protection Agency—US EPA, 1985,1989), a clean water oxygen transfer standard (American Society of Civil Engineers—ASCE 1984,1991) and guidelines for process water testing (American Society of Civil Engineers—ASCE, 1997). Clean water testing and off-gas testing are described in detail in these publications. The increase in accuracy and precision in designing and quantifying aeration systems is the net result of improving this testing method. The off-gas technique provides accurate and precise oxygen transfer measurement for diffused aeration systems (coarse, fine, turbines, and jets) at virtually all process conditions. The dissolved oxygen concentration or oxygen uptake rate does not interfere with or limit the test procedure.

Clean water test results can be reported as standard oxygen transfer efficiency (SOTE, %), standard oxygen transfer rate (SOTR, kgO_2/h or lbO_2/h), or standard aeration efficiency (SAE, $kgO_2/kW/h$ or $lbO_2/HP/h$). The effect on transfer efficiency from contaminants is usually quantified by the α factor (ratio of process water to clean water mass transfer coefficients, or K_La_{pw}/K_La_{cw}). In the same way, the effects of process water salinity are usually quantified by the β factor, i.e. the ratio of oxygen saturation in process water to clean water, or $C^*_\infty/C^*_{\infty cw}$ (Stenstrom and Gilbert, 1981). Standard conditions are defined in a protocol (American Society of Civil Engineers—ASCE 1984,1991) and correspond to 20 °C, mean atmospheric pressure, zero dissolved oxygen, and zero effect of salinity or other contaminants (e.g., α factor = 1.0, β factor = 1.0). Because it is generally not possible or easy to measure α factors, the transfer efficiencies are generally reported as αSOTE, which includes all adjustments (DO, temperature, barometric pressure, salinity), except the α factor. We also use this approach, which is convenient because the other non-standard conditions are easily measured. The α factor can be calculated from off-gas results when clean water data are available. When it is desirable to differentiate the effects of wastewater contaminants and fouling, an αF factor is often used. In this work we do not make this differentiation and α represents both impacts. In order to exclude the effects of depth on mass transfer, all data in this paper were reported as αSOTE/Z, where Z is the diffuser submergence, and its units are %/m. When calculating SAE or power consumption, care is necessary because different power measurements may be used. In general, "wire" power is preferable, which includes inefficiencies from piping losses, blower, coupling, and motor. Wire power is used in this paper.

Different aeration methods (i.e., coarse bubble, fine pore, surface, etc.) show different ranges of α factors, as recorded by Kessener and Ribbius (1935) as early as in the 1930s. Previous off-gas investigations by Masutani and Stenstrom (1990) recorded a wide range of α factors for fine bubble diffusers, from as low as 0.17 to as high as 0.7. The general understanding that αSOTE and the α factor are functions of MCRT, or sludge age, of airflow rate, and of tank geometry (diffuser submergence, number, and surface area) was recently discussed and quantified (Rosso et al., 2005). As more data have become available, fine-bubble aeration systems in activated sludge processes operating at low loading rates (i.e., high MCRT or low food-to-microorganism ratio [F/M]) are generally associated with higher α factors. Fine-pore diffusers have an important disadvantage: the need for periodic cleaning. These devices are subject to fouling and scaling, and their efficiencies decrease over time (United States Environmental Protection Agency—US EPA, 1985). This decrease in efficiency and its economic impact have been recently quantified by the authors (Rosso and Stenstrom, 2005).

Fig. 2 shows typical examples of selected measurements for the three types of plants. The transfer efficiency varies according to diffuser condition and time in operation. The measurements were divided in four categories: NEW (within 1 month from installation), USED (between 1 and 24 months of operation), OLD (over 24 months in operation), and CLEANED (within 1 month from a cleaning event). We recorded in our dataset the different cleaning techniques

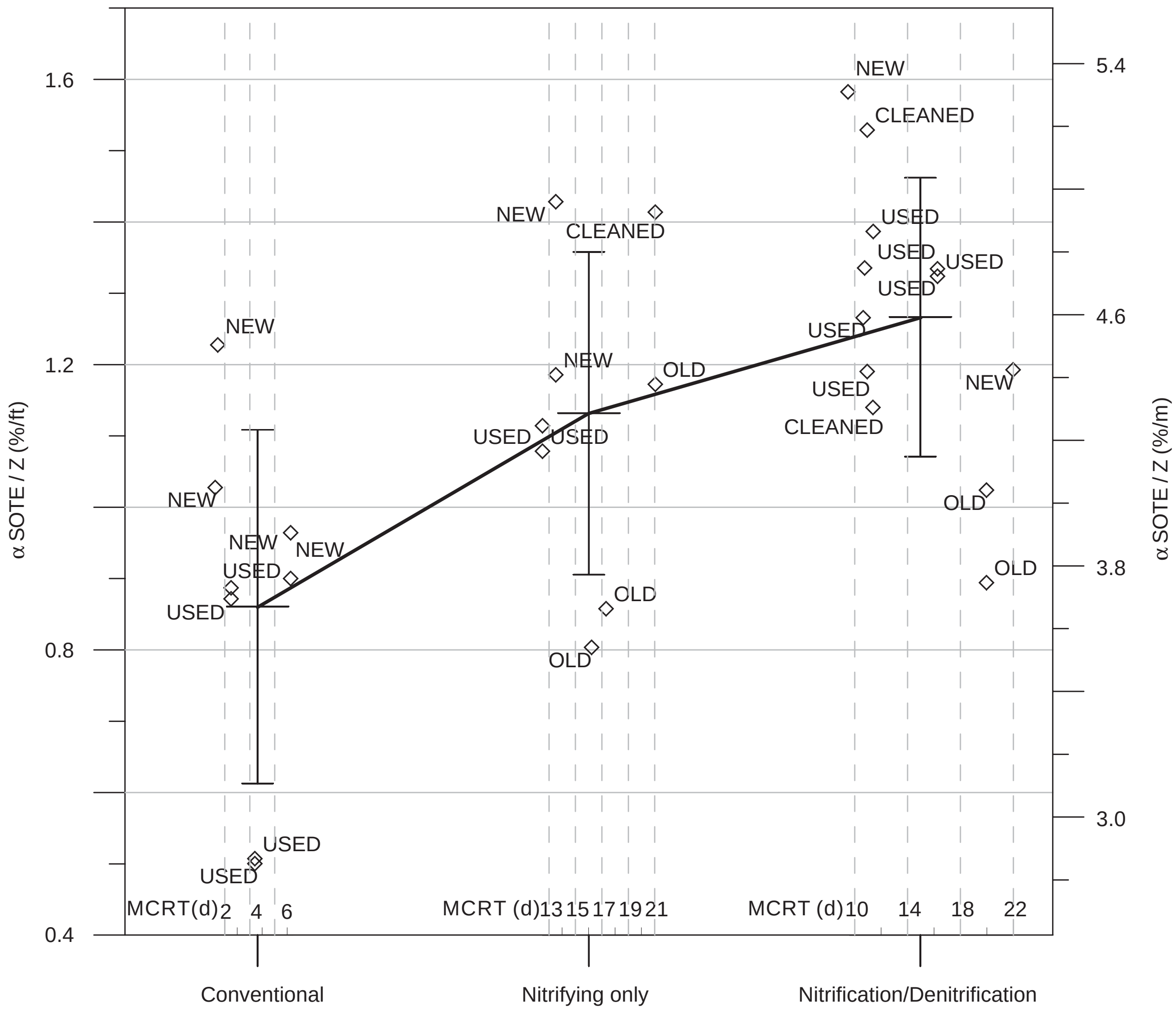

Fig. 2. Normalized standard oxygen transfer efficiency for selected plants operating at different sludge age and time in operation. Labels refer to the diffuser status: NEW (within 1 month from installation), USED (between 1 and 24 months of operation), OLD (over 24 months in operation), and CLEANED (within 1 month from cleaning).

implemented at each location, but the difference in efficiency restoration between different techniques was too varied to quantify. The error bars represent one standard deviation in the data and the mean oxygen transfer efficiencies (3.75%/m, 4.3%/m, and 4.6%/m for conventional, nitrifying-only, and NDN operations, respectively) are used for calculating the aeration requirements. The efficiency increases with increasing MCRT and is higher for NDN when compared to nitrifying-only for the same range of MCRTs. Both modes are higher than conventional plants.

2.3. Oxygen requirements

The oxygen requirements for the process are for heterotrophic (carbon) consumption and ammonia oxidation. The oxygen required for carbon oxidation will depend upon the removal efficiency as well as the sludge production. There are various models and methods to calculate oxygen requirements (Metcalf Eddy and Inc., 2003; IAWPRC, 1986), but a simplified approach is used here based upon cell yield and MCRT. The substrate balance is

$$S_{in} = S_{out} + S_{ox} + S_b, \tag{1}$$

where S_{in} is the influent substrate concentration, S_{out} the effluent substrate concentration, S_{ox} the substrate oxidized, S_b the substrate converted to biomass (synthesis).

Any convenient units for substrate concentration may be used and chemical oxygen demand (COD) is used here. For convenience, a constant influent COD concentration is assumed and the change in effluent COD with MCRT is ignored. The difference between influent and effluent CODs is consumed and the split between oxidation and synthesis can be described by an observed cell yield as follows:

$$Y_{obs} = Y_T(1 + K_d\,\mathrm{MCRT})^{-1}, \tag{2}$$

where Y_{obs} is the observed cell yield (mass cells/mass substrate), Y_T the theoretical cell yield (mass cells/mass substrate), K_d the decay coefficient (time^{-1}).

The oxygen demand for carbon removal in the aeration tank becomes:

$$\mathrm{OUR} = (S_{in} - S_{out})(1 - Y_{obs})Q, \tag{3}$$

where OUR is the oxygen uptake rate (mass time^{-1}), Q the plant flow rate (volume time^{-1}).

Ammonia oxidation requires oxygen based upon the following stoichiometry:

$$NH_4^+ + 2O_2 \rightarrow H_2O + NO_3^- + 2H^+. \tag{4}$$

The mass of oxygen required for nitrification is 4.5 parts of oxygen per part of nitrogen. The actual requirement will be slightly lower due to nitrifying cell synthesis, but this small difference is ignored (i.e., 4.5 in lieu of 4.3). For nitrification and denitrification, the oxygen used for nitrification is partially recovered because the produced nitrate is used as an electron acceptor, displacing oxygen. A portion of the influent ammonia is used for heterotrophic cell synthesis and is unavailable for nitrification/denitrification. This fraction is ignored because the major portion of it will be returned to the aeration basin in the digester supernatant as shown in Fig. 1. The stoichiometry for this reaction is shown in Eq. (5), if methanol is used as a carbon source model:

$$6NO_3 - + 5CH_3OH + 6H^+ \rightarrow 3N_2 + 5CO_2 + 13H_2O. \tag{5}$$

Eq. (5) provides an electron acceptor that is equivalent to 2.84 parts of oxygen per part of nitrogen. For the ammonia that is nitrified and denitrified, Eqs. (4) and (5) can be summed, as follows:

$$\tfrac{3}{4}O_2 + NH_4^+ \rightarrow \tfrac{1}{2}N_2 + H^+ + 1.5H_2O. \tag{6}$$

Therefore the net oxygen required for nitrification/denitrification is only 1.71 parts of oxygen per part of nitrogen. For the purposes of this analysis, it is assumed that 90% of the nitrate is denitrified. This value represents the amount of denitrification required to obtain the upper range of expected efficiency of a denitrifying activated sludge process (Metcalf and Eddy Inc., 2003).

2.4. Biomass production and disposal cost

The waste biomass produced can be directly calculated from the observed cell yield. The biomass produced from heterotrophic growth can be calculated as follows:

$$X_h = (S_{in} - S_{out})Y_{obs}Q, \tag{7}$$

where X_h is the heterotrophic biomass production (mass COD/d).

The biomass yield from denitrification is lower and must be calculated separately. The substrate required for denitrification can be calculated from Eq. (5), and is 0.833 moles methanol/mole N. The COD of methanol is 1.5 moles/mole N. Therefore the COD oxidized by the anoxic reactions, using methanol as a model is 1.25 moles COD/mole N or, in terms of mass, 2.85 parts of COD per part of N. This substrate must be subtracted from the heterotrophic biomass production (Eq. (7)). The biomass produced from anoxic reactions is:

$$X_{an} = \{[NH_3 - N]_{in} - [NH_3 - N]_{out} - [NO_3 - N]_{out}\}QY_{a\,obs}, \tag{8}$$

where X_{an} is the anoxic biomass production (mass COD time^{-1}), [NH_3-N], [NO3-N] the nitrogen mass concentrations, Y_{aobs} the observed yield for anoxic biomass production.

The observed yield is calculated from the theoretical yield using Eq. (2). The biomass productions are in units of COD, which can be converted to volatile suspended solids (VSS) by dividing by 1.42, which is the commonly used COD of waste activated sludge, assuming an empirical cell formula of $C_5H_7NO_2$. The biomass of the nitrifying population is ignored, since it is small.

Fig. 1 shows the plant process flows and the waste activated sludge is thickened and digested. It is assumed that the nitrogen in the waste biomass is returned in the supernatant. The mass of waste biomass and methane production will change as a function of MCRT, and therefore needs to be included in the economic analysis. It is assumed that 50% VSS destruction occurs in the digester and methane production ranges from 0.75 to 1.12 m^3/kg/VSS destroyed and the biogas composition is 65% methane on a dry basis (Metcalf and Eddy Inc., 2003). The higher methane production value was assumed for the conventional system and the lower value was assumed for other two modes. This is realistic because greater endogenous respiration occurs at longer MCRTs and the waste biomass will be more oxidized. A cost of ultimate disposal is assumed based upon current costs in Southern California.

3. Results and discussion

3.1. Operating costs

To demonstrate the impact of denitrification on plant economics, the related operating costs from the three types of plants were calculated. Oxygen requirements, aeration efficiency, sludge disposal cost and digester gas value were considered. Other operating costs, while important, were not considered because they do not vary significantly among the three modes. The cost-analysis results for the three scenarios are reported in Table 1.

Oxygen requirements were calculated using the stoichiometry summarized in Eqs. (3)–(6). In order to calculate the required airflow rate, it was assumed the process temperature is 20 °C and the dissolved oxygen (DO) concentration to be 2 mg/l for all three scenarios. This is an acceptable value for the NDN as well, since there is no aeration in the anoxic zone. DO values and depth are needed to adjust efficiency values to field conditions. Assuming a common depth of 5 m, the required airflow rate may then be calculated. Nitrifying-only operations, with higher MCRT, result in higher oxygen requirements. However, higher MCRT operations are associated with higher field transfer efficiencies. The results show a comparable airflow rate for conventional and NDN layout, this being due to the oxygen nitrification credit and to higher efficiencies at higher MCRTs, as discussed above. The highest required airflow rate is for the nitrification-only layout, although the increase is not as much as the increase in oxygen requirement, due to the increased efficiency with longer MCRT. The minimum oxygen requirement, and therefore the minimum aeration burden, occurs during NDN operation.

As previously mentioned, sludge production values vary amongst layouts and will be lowest for NDN operations. Therefore, assuming the digesters operating at the same retention time in all three cases, sludge disposal costs will be highest for conventional operations. Due to higher methane production yields, conventional operations will result in higher methane mass per unit time. Assuming the digester gas has a 65% methane composition, gas value at 5.4 USD /10^6 kJ, (current methane value in Southern California), and 75% discounting due to gas processing costs (scrubbing, adsorption, combustion, etc.), the final methane gas value is calculated as 0.06 USD/m^3. The net sum of costs minus credits is here referred to as total cost. Table 1 shows that NDN operations are the most cost-effective on the basis of total costs as well as aeration costs alone.

The top-half of Fig. 3 plots three combinations of plant costs and credits, and the total cost. Values in the top-half of Fig. 3 are reported as USD/1000 m^3, and costs are plotted as positive values, i.e., higher values correspond to higher costs. The combinations here proposed and discussed are: 1. sludge disposal + aeration costs; 2. sludge disposal cost—methane credit; 3. aeration cost—methane credit.

Case 1. This may be the case of a treatment plant where the methane is not recovered and burned for energy and/or heat production. This is often the case of small plants, where the biogas is usually flared because the methane production is too small to justify the installation of a power generation facility. In this case, conventional operations have the highest costs.

Case 2. This is the case of a treatment plant whose electrical expenditure is recorded by a different entity than operations. Therefore, sludge cost—methane credit is the solids processing facility net operative cost. Fig. 3 shows the comparability of the three costs.

Case 3. This case may represent a treatment facility with negligible sludge disposal costs. In this case nitrifying-only operations have the highest operating cost, followed by NDN.

Costs, credits and their case combinations were normalized to dimensionless ratios in order to eliminate local currency effects. The lower half of Fig. 3 shows the relative cost of nitrifying-only and NDN operations when compared to conventional operations. The costs for conventional operations were used as baseline; therefore, all dimensionless costs and credits related to such operations amount to 1.0. Thus, the plot profiles will be different from the top-half of Fig. 3, since they

Table 1
Comparative summary of operational cost for activated sludge practices

Cost-analysis results for activated sludge plants			
	Conventional	Nitrifying-only	Nitrifying/denitrifying
Sludge disposal cost (USD/d)	140	78	69
Oxygen requirement (kgO_2/d)	3800	5034	3469
Field transfer efficiency (%)	15.3	17.6	18.8
Aeration cost (USD/d)	39	52	36
CH_4 production credit (USD/d)	96	36	32
Total cost (USD/d)	83	94	73

Plant flow: 20000 m^3/d; influent concentration: 350 mg_{BOD}/l; effluent concentration: 20 mg_{BOD}/l; theoretical yield: 0.7/d (for carbonaceous growth), 0.15/d (for nitrifying growth); decay coefficient: 0.13/d (for carbonaceous growth), 0.05/d (for nitrifying growth); power cost: 0.15 USD/kW/h; sludge disposal cost: 20 USD/wet t; methane gas value: 0.25 USD/m^3; final methane resale value: 0.06 USD/m^3; required blower energy: 1.17 kW/m^3.

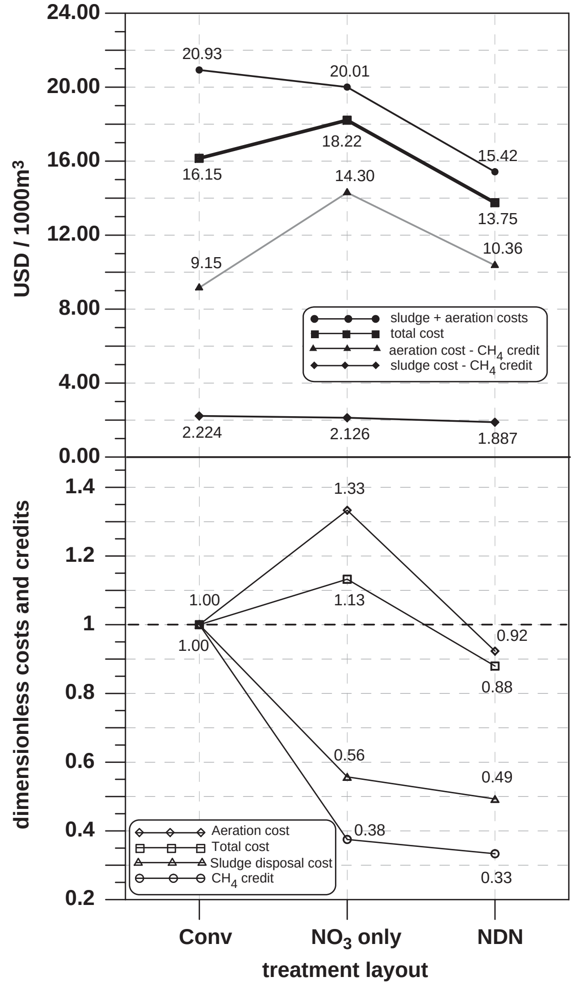

Fig. 3. Dimensional and dimensionless costs and credits for different layouts.

are ratios of all values for nitrifying-only and NDN to conventional operations. In terms of total cost, nitrifying-only operations will total 1.13 times (113%) the total cost of conventional operations, while NDN will total 0.88 (88%) of conventional total cost. Aeration costs are highest for nitrifying-only operations, due to their higher oxygen requirements. Costs and credits associated with solid-handling (i.e., sludge disposal cost and methane credit) have lower magnitude for nitrifying-only and NDN operations, due to the smaller amounts of sludge produced per unit time.

Fig. 4 shows the sensitivity of results to the decay coefficient for all cost scenarios. Different K_d values were considered because different sources used in this paper used wide ranges of K_d. The reasons are varied but are partially due to different temperatures and the way different process models represent decay. For the range of K_d values (0.026–0.25 d^{-1}), the NDN case always has

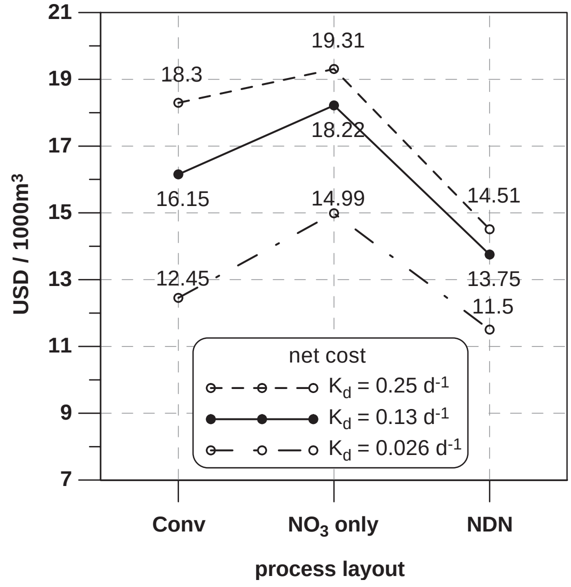

Fig. 4. Sensitivity analysis for net cost of different layouts.

lowest operating cost; the conclusions are unaffected by the magnitude of decay coefficients.

3.2. Need for capital improvements

The analysis has been restricted to operating cost and a detailed analysis of potential changes in process sizes is beyond the scope of this paper. The change from conventional to NDN mode will require an increased MCRT, perhaps a three fold increase, which will require supporting higher mixed-liquor suspended solids (MLSS) mass and DO concentration in the aeration tank (Stenstrom and Song, 1991). This may be partially achieved with higher MLSS concentrations, which will result in higher solids flux to the final clarifiers. Usually conversion to NDN operation creates better settling sludge (Harper and Jenkins 2003), which will increase the limiting secondary clarifier solids flux. The amount of the increase will depend upon site-specific considerations, including the condition of the existing clarifiers and the loading of the plant relative to its design basis. The best way of supporting higher MLSS mass will be different for each plant, and the required capital improvements will be site specific.

The MLE process requires internal recirculation pumps as shown in Fig. 1. We have not considered this capital cost, nor have we quantified the operating cost in our analysis. This cost will again be site specific. The pumping cost will ordinarily be low, since there is usually no elevation change (static head). Commercially available low head pumps sized for this application at

200% internal recycle rate have energy costs less than 5% of the blower cost. A more detailed analysis should be performed for each site. Also, since the air requirements for conventional and NDN operations are comparable, it may not be necessary to require an upgrade in blower capacity.

4. Conclusions

This analysis shows that NDN operation has a lower total operating cost (aeration + sludge disposal costs—methane credit) than a nitrifying-only or a conventional treatment plant, for the conditions of this analysis. In relative terms, if conventional operations have a total cost of 1.00, nitrifying-only will have a total cost of 1.13, and NDN operations a total cost of 0.88. NDN operation offers an oxygen credit due to its process nature, and higher oxygen transfer efficiencies associated with the higher MCRT. The two factors overcome the additional oxygen demand that occurs at longer MCRT. Additional potential advantages exist, such as higher removal efficiency of more complex organic chemicals (such as pharmaceuticals or synthetic hydrocarbons), and lower sludge production.

NDN operation should always be evaluated as an alternative to conventional treatment. The lower operating cost and environmental advantages may more than compensate for the increased size of aeration tanks or clarifiers. In addition, the choice of NDN versus conventional operations may require increased aeration tank volume or clarifier area, but may not require an upgrade in blower capacity. The commonly accepted assumption that NDN is a more expensive type of operation should be abandoned.

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