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Two-Phase Decay of Aerobic Sludge Shown by Online Fluorescence and Modeled with Interaction of Heterotrophs and Nitrifiers

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ABSTRACT

Both heterotrophs and autotrophic nitrifiers are present in aerobic sludge digestion systems. The endogenous heterotrophic metabolism releases ammonium that, via oxidation to nitrite and nitrate, serves as the energy substrate for growth and survival of nitrifiers. The population interaction was used to explain the two-phase decay behavior observed in this study for the biosolids collected from rotating biological contactors (RBC) as well as the mixed primary and secondary sludge. Separated by an intermediate transition, the two phases followed different first-order decay kinetics, with the first phase being much faster. The two-phase phenomenon was supported by the on-line fluorescence profile monitored by an NAD(P)H fluorometer. The fluorescence showed an initial rise to a high plateau, a sharp decline after staying at the plateau for 30–50 h, and a trailing very slow decrease. The initial plateau corresponded to the first-phase of rapid sludge digestion; the trailing fluorescence decline corresponded to the second phase of slow digestion. The transition between the two phases occurred during the sharp decline from the plateau. The fluorescence profile was consistent with our earlier observation of negligible NAD(P)H fluorescence from nitrifiers compared to that from heterotrophs. The VSS reduction was, therefore, modeled by considering the decay and interaction of heterotrophs and nitrifiers. The model described the experimental results very well. The best-fit average decay constants were 0.067 d$^{-1}$ for nitrifiers and 0.95 d$^{-1}$ for heterotrophs in the RBC biosolids. The apparent heterotrophic decay constants (in the presence of nonbiomass VSS) were found to be 0.48–0.55 d$^{-1}$, consistent among different runs of mixed sludge digestion.

Key words: aerobic sludge digestion; kinetic model; volatile suspended solids; NAD(P)H fluorescence; heterotroph; nitrifier; oxygen uptake rate

INTRODUCTION

AEROBIC SLUDGE DIGESTION is a common method used to reduce solids and destroy pathogenic substances in the wasted sludge. Compared to its anaerobic counterpart, aerobic digestion is easier to operate and less expensive in the capital cost. Other advantages include the lower BOD (biological oxygen demand) in the super-
natant liquor; recovery of more fertilizer values; and production of an odorless, humus-like, biologically stable product (Metcalf & Eddy, 1991). Aerobic digestion, however, requires high power cost for aeration, especially at the early stage of digestion when the biological activity is high. Knowledge of its decay kinetics is important to the process economy for aeration design and operation.

The solids reduction in aerobic digestion is typically described by a first-order kinetic model (Metcalf & Eddy, 1991):

$$\frac{d(VSS)}{dt} = -k_d(VSS)$$  \hspace{1cm} (1)

where VSS is the concentration of volatile suspended solids in the sludge (mg/L), $t$ is time (d), and $k_d$ is the first-order decay constant (d$^{-1}$). It, however, works well only for a short initial phase (Koers, 1979; Urbain et al., 1993). For long digestion (e.g., 60 d), VSS in Equation (1) is replaced by $VSS_b$ (biodegradable VSS, $VSS - VSS_i$) because VSS does not reach zero but tends to an inert organic residue ($VSS_i$) (Krishnamoorthy and Loehr, 1989). The “good fitting” with the modified first-order decay in the latter case could, however, be misleading. Because solids reduction was minimal after about 20 days, $r^2$ values larger than 0.95 were obtained even when there were significant deviations in the early stage, such as those for the digestion of primary sludge reported by Krishnamoorthy and Loehr (1989).

The use of constant $k_d$ in the above cases neglects the dependency of digestion rate on the varying microbial population (composition and physiology) and physical/chemical properties (digestibility and efficiency as energy source to microorganisms) of the sludge during the process. For improvement, it is essential to adjust the rate constant $k_d$ according to the varying microbial composition of the sludge along the digestion process. In this work, an on-line NAD(P)H fluorometer was applied as a tool for monitoring such changes. NAD(P)H are the reduced form of coenzymes NAD and NADP, that is, nicotinamide adenine dinucleotide (phosphate). Universally present in living cells, coenzymes NAD(P) are the major intermediate electron and hydrogen carriers coupling catabolism and respiration/anabolism. For example, for heterotrophs, NAD(P)H are formed from the oxidized counterparts, NAD(P)$^+$, during the catabolism of organic substrates. NADH is then cycled (oxidized) back to NAD$^+$ via the respiration chain for energy (ATP) generation, while NADPH via the anabolism to NADP$^+$ for providing the reducing power required in biosynthesis. NAD(P)H are fluorescent (peak excitation: 340 nm, peak emission: 460 nm) but NAD(P)$^+$ are not (BioChem Technology, Inc., 1983). The NAD(P)H fluorescence thus depends on the kinetic balance of its generation by catabolism and consumption by respiration and anabolism. The fluorometer has been applied to biological nutrient removal processes (Ju et al., 1995), microbial fermentations (e.g., Groom et al., 1988; Li and Humphrey, 1989; Kwong and Rao, 1994), and animal cell cultures (MacMichael et al., 1987; Siano and Mutharasan, 1990), for process monitoring and/or control as well as studying cellular metabolism.

For the current work, the fluorometer was envisioned as being particularly useful for following the interactions between, and changes in, the heterotrophic and autotrophic (especially the nitrifiers) populations in the sludge. The presence of nitrifiers in aerobic sludge digestion processes is well known (e.g., Hao et al., 1991; Anderson and Mavinic, 1993). The endogenous metabolism of cellular proteins releases ammonium (NH$_4^+$), which, via its oxidation to nitrite (NO$_2^-$) and, subsequently, to nitrate (NO$_3^-$), can serve as the energy substrate for the growth and/or long-term survival of nitrifiers (Madigan et al., 1997). The interaction between these two groups would offer a good starting point for considering the change of microbial population in the digestion process. The rationale of using the NAD(P)H fluorescence for monitoring such interaction is briefly described below:

The electrons donated from nitrification (i.e., oxidation of NH$_3$ and NO$_2^-$) enter the respiration chain at a more oxidized level (cytochrome $a_1$) than NAD(P)H (Bock et al., 1986). The critical role of NAD(P)H for heterotrophs as the intermediate electron carriers associated with energy generation is therefore totally absent for nitrifiers. Instead, an energy-consuming reversed electron flow has to be used to form the NAD(P)H that are required for biosynthetic purposes (especially the CO$_2$ fixation via the Calvin cycle) (Madigan et al., 1997) and for priming the ammonia monooxygenase in the initial ammonia oxidation to hydroxylamine (NH$_2$OH) (Wood, 1986). The dramatically different energy-related functions of NAD(P)H lead to much lower NAD(P)H levels in nitrifiers than in heterotrophs. For the nitrifiers enriched from the sludge of a wastewater treatment plant at the cell concentrations up to 5 g/L, only the background fluorescence from non-NAD(P)H sources was detected by the fluorometer (Ju and Nallagatla, 2001). On the other hand, the fluorescence from NAD(P)H was clearly observed with <0.5 g/L of heterotrophs, for example, Escherichia coli (Trivedi and Ju, 1994) and Pseudomonas aeruginosa (Ju and Trivedi, 1992). The different NAD(P)H levels between heterotrophs and nitrifiers thus provide a good opportunity for monitoring the varying population of these groups during the aerobic sludge digestion.
MATERIALS AND METHODS

Fresh samples of raw sludge (the mixed primary and secondary sludge in a waste pit) and plant effluent were taken from the Fishcreek wastewater treatment plant near Akron, OH. Having an average flow rate of 4 million gallons per day (i.e., 0.18 m³/s), the plant uses rotating biological contactors (RBC) for secondary treatment and chemical precipitation for primary treatment. The combined primary and secondary sludge is added, in three shifts a day, to four aerobic digesters where the sludge is treated for an average detention time of 20 days. The digested sludge is then concentrated by belt-filter presses to form sludge cake, and discharged to the PPG Lime Lakes Reclamation Area in Barberton, OH.

The sludge samples taken from the plant was centrifuged under 14,500 g for 10 min. The solids collected were resuspended in plant effluent to make the total suspended solids concentration (TSS) in the range of 1,500–2,700 mg/L, with VSS of 1,200–2,100 mg/L. Seven hundred milliliters of such prepared sludge was put in a double-sidearm glass reactor, mixed with a magnetic stirrer, and aerated with prehumidified air. The use of diluted sludge in this study was to ensure proper aeration and, especially, mixing so that the sludge fluorescence monitored on-line would be representative. The fluorescence profile was followed continuously using the BioGuide system (BioChem Technology, Inc., King of Prussia, PA) designed to measure the fluorescence of intracellular NAD(P)H. The fluorometer accessed the sludge through an optical well mounted on the reactor wall. The fluorescence intensity was expressed as a manufacturer-calibrated Normalized Fluorescence Unit (NUF). The reactor was also equipped with dissolved oxygen (DO) probe (YSI 5739) and pH electrode (Mettler Toledo 465). The whole setup was placed in a chamber coated with opaque black polymer to avoid any interference from outside light. The pH was controlled at 6.8–7.1 by automatic acid/base addition. The temperature of each batch started from the room temperature (25 ± 2°C), increased to 31 ± 2°C within 5 h, and then kept around that level until the end of the experiment. Samples of 10 mL were withdrawn periodically from one of the side arms for analyses of TSS and VSS. Error analysis of the sampling was within ±5%, assessed by averaging multiple samples taken at the same time.

TSS and VSS analyses

The sample was centrifuged under 14,000 g for 10 min. The supernatant was carefully withdrawn. The pellet was resuspended in deionized water and poured into a preweighed aluminum pan. The residues in the centrifuge tube were rinsed with deionized water and also collected in the pan. The solids suspension was dried first at 80°C and then at 105°C to constant weight, cooled in desiccator, and weighed. It was then placed in an oven at 550°C for 20 min and weighed after being cooled partially in air and then in desiccator. TSS and VSS were calculated accordingly. The imprecision of TSS and VSS data was about ±10 mg/L. The easier centrifugation procedure was checked to give consistent (±7%) results with the standard filtration method (APHA, 1995).

Oxygen uptake rate analysis

The dynamic, in situ measurement was achieved using a DO probe. With the aeration being turned off, DO decreased as a result of the microbial respiration. The DO profile was recorded with computer data acquisition software. The aeration was turned back on before DO dropped below 10% of air saturation, to prevent damage to the sludge population caused by low DO. The oxygen uptake rate (OUR) was determined as the absolute value of the slope of decreasing DO (Sundararajan and Ju, 1995).

RESULTS AND DISCUSSION

Typical profiles of fluorescence, OUR, and VSS reduction

Four batch runs of aerobic sludge digestion were made with the mixed primary and secondary sludge. The profiles of fluorescence, specific OUR, and VSS reduction of a typical run are shown in Fig. 1. Not shown is the profile of TSS, which is very similar to that of VSS.

The fluorescence increased rapidly at the initial stage, which lasted 10–20 h in different runs. The fluorescence signal (recorded at one point per minute), for unknown reason(s), became increasingly fluctuating as it reached the highest level. The time-averaged fluorescence, however, remained relatively constant for a period of 30–50 h before starting to decline sharply to a level typically lower than the initial fluorescence level. The rapid fluorescence drop was followed by a much slower decrease until the end of the batch process. The overall fluorescence profile of the batch digestion process was thus an early hump trailed by a slow decline.

Both VSS and SOUR decreased as the digestion proceeded and the decreasing rates were especially high initially. From the regulatory point of view, for vector attraction reduction the EPA regulation requires greater than 38% of VS reduction or, among other allowable al-
ternative tests, SOUR < 1.5 mg O₂/g TS-h (Farrell et al., 1996). As indicated in Fig. 1, these criteria were met when the on-line fluorescence completed the initial hump. (VS and VSS were expected to be similar in aerobic digestion systems because of the low concentrations of dissolved organics.) The potential of monitoring aerobic sludge digestion using the online fluorometer was indicated, although future study is needed to examine its applicability to plants of different wastewater treatment designs.

Two-phase phenomenon of aerobic sludge decay

Plotted in Fig. 1 with a semi-logarithmic scale, the VSS reduction exhibited a clear two-phase behavior. Each phase of the digestion followed different first-order kinetics, with an intermediate transition. Koers (1979) observed similar phenomenon and approximated the digestion profiles with two subsequent first-order kinetic equations, without giving fundamental basis for the observation.

The two-phase decay is also revealed by the fluorescence profile: the initial fluorescence hump corresponds to the first phase of rapid sludge digestion; the trailing fluorescence decline corresponds to the second phase of slow digestion. The transition between the two phases occurs during the sharp decline of the hump. Another indication of the two-phase behavior came from the acid/base addition used for pH control in the digestion experiments. It was observed that the acid was added initially during the fluorescence rise and plateau stages (i.e., before about 50 h in Fig. 1), while the base addition began with the sharp decline of the fluorescence.

The two-phase phenomenon was not (at least not solely) caused by the use of mixed primary and secondary sludge. It was observed also in a run made with the biosolids collected directly from the surface of the rotating biological contactors (results shown later). The phenomenon may be explained by the significantly different metabolism of the heterotrophs and autotrophic nitrifiers present in the sludge. During sludge digestion, the endogenous heterotrophic metabolism oxidizes cellular biopolymers for the energy to extend survival. Ammonium is released in the process. The ammonia serves as the energy substrate for growth and survival of nitrifiers. The aerobic digestion therefore proceeded with a rapid
first-phase decay of heterotrophs, during which the ammonium concentration accumulated initially, necessitating the acid addition for pH control. Nitrifiers may grow on the accumulated ammonia, the extent depending on the initial population ratio between the two groups. The growth of nitrifiers and/or the decay of heterotrophs eventually caused the rate of ammonium consumption by the nitrifiers to exceed the rate of generation from heterotrophic digestion. The pH would then begin to drop, and the base addition became required for pH control. After the accumulated ammonium was depleted, the second-phase decay of nitrifiers took place. The mechanism proposed above is consistent with the hump-and-tail fluorescence profile, considering our earlier observation of negligible NAD(P)H fluorescence from nitrifiers compared to that from heterotrophs (Ju et al., 1995; Ju and Nallagatla, 2001). The fluorescence observed in the second phase was predominantly from some non-NAD(P)H background materials. The fluorescence characteristics of the digestion process are described in more detail elsewhere (Li and Ju, 1999).

The proposed changes of heterotrophic and nitrifying population in aerobic digestion are also supported by the results of Hao et al. (1991), who used repeated cycles of alternating aerobic/anoxic operations for sludge digestion (results replotted in Fig. 2). The zigzag profile of nitrate concentration indicated that nitrification occurred during the aerobic period (generating nitrate from ammonia) while the denitrification by heterotrophic microorganisms occurred during the anoxic period (consuming nitrate for anaerobic respiration). The rates of nitrification and denitrification (mg NO$_3^-$-N/g VSS·h, produced or consumed) for these periods are shown in Fig. 2. The heterotrophic denitrification rate dropped rapidly initially and leveled off after about 75 h. On the other hand, the nitrification rate remained relatively constant (or increased slightly) initially and decreased slowly in the later stage. The changes of denitrification and nitrification
rates agreed well with the interaction of heterotrophs and nitrifiers proposed in the current work. Note especially that the rapid drop of heterotrophic denitrification rate corresponded to the rapid VSS reduction (in time profiles) and the much slower decline of nitrification rate in the later stage corresponded to the second phase of slow VSS reduction.

**Modeling aerobic decay with interaction between heterotrophs and nitrifiers**

The first-order kinetics given in Equation (1) is the same as that used to describe the cell death of pure microbial culture (Bailey and Ollis, 1986), that is,

\[ \frac{dX}{dt} = -k_d X \]

where \( X \) is the live cell concentration (g/L). The success of Equation (1) in modeling the initial sludge digestion process can be attributed to the correspondence of VSS to cell concentration. The decay constant

\[ k_d \]

is the decay constant of nitrifiers (\( k_d' \)), indicating the digestion rate per unit \( X \), expected to vary with the species. A comprehensive model considering every species present is impossible for complex processes of sludge digestion. The two-phase decay behavior observed in this study has thus been modeled by considering the change and interaction of the two groups, heterotrophs and autotrophic nitrifiers, during the process.

For the fresh biosolids collected from the RBC surface, the VSS is considered to consist of those from the heterotrophs (\( \text{VSS}_h \)) and the nitrifiers (\( \text{VSS}_n \)), that is,

\[ \text{VSS} = \text{VSS}_h + \text{VSS}_n \]  
(2)

The endogenous decay of heterotrophs follows the typical first-order kinetics, with a decay constant \( k_h \) (d\(^{-1}\)):

\[ \frac{d\text{VSS}_h}{dt} = -k_h \text{VSS}_h \]  
(3)

As described earlier, the ammonium released from the heterotrophic decay may support the growth of the autotrophic nitrifiers. The change of nitrifying population, thus, contains an additional growth term that is assumed proportional to the digestion rate of heterotrophs:

\[ \frac{d\text{VSS}_n}{dt} = -k_n \text{VSS}_n + k_i \left( \frac{d\text{VSS}_h}{dt} \right) = -k_n \text{VSS}_n + k_i \text{VSS}_h \]  
(4)

where \( k_n \) is the decay constant of nitrifiers (d\(^{-1}\)), \( k_i \) is the interaction (proportionality) constant (dimensionless), and \( k = k_i \cdot k_n \), resulting from the substitution of Equation (3) into Equation (4).

Equations (3) and (4) can be solved to give

\[ \text{VSS}_h = \text{VSS}_h^0 e^{-k_h t} \]  
(5)

\[ \text{VSS}_n = \left( \text{VSS}_n^0 + \frac{k}{k_h - k_n} \right) e^{-k_n t} - \text{VSS}_n^0 \frac{k}{k_h - k_n} e^{-k_h t} \]  
(6)

where \( \text{VSS}_h^0 \) and \( \text{VSS}_n^0 \) are the initial VSS of the heterotrophs and nitrifiers, respectively. The time profile of the total VSS digestion is then given by combining Equations (5) and (6):

\[ \text{VSS} = \text{VSS}_h^0 \left( 1 - \frac{k}{k_h - k_n} \right) e^{-k_h t} + \left( \text{VSS}_n^0 + \text{VSS}_n^0 \frac{k}{k_h - k_n} \right) e^{-k_n t} \]  
(7)

By taking \( f_h \) as the fraction of heterotrophic VSS in the initial sludge (at \( t = 0 \)), Equation (7) can be simplified to

\[ \frac{\text{VSS}}{\text{VSS}^0} = f_h \left( 1 - \frac{k}{k_h - k_n} \right) e^{-k_h t} + \left( 1 - f_h \left( 1 - \frac{k}{k_h - k_n} \right) \right) e^{-k_n t} \]

\[ = Fe^{-k_h t} + \left( 1 - F \right) e^{-k_n t} \]  
(8)

where

\[ F = f_h \left( 1 - \frac{k}{k_h - k_n} \right) = f_h \left( 1 - \frac{k k_h}{k_h - k_n} \right) \]  
(9)

The above derivation does not consider the inert, non-biodegradable organics in the VSS. If desirable, Equation (8) can be modified to

\[ \frac{\text{VSS}}{\text{VSS}^0} = f_i + \left( 1 - f_i \right) \cdot \left[ Fe^{-k_h t} + \left( 1 - F \right) e^{-k_n t} \right] \]  
(10)

by assuming a same inert fraction (\( f_i \)) for the initial heterotrophic and nitrifying VSS, that is,

\[ f_i = \frac{\text{VSS}_h^i}{\text{VSS}_h^0} = \frac{\text{VSS}_n^i}{\text{VSS}_n^0} = \frac{\text{VSS}_i}{\text{VSS}_i^0} \cdot \]

where the subscript \( i \) denotes the inert fraction of the corresponding VSS.

**Model fitting**

The primary objective of this study was to examine the changes in microbial populations during the aerobic sludge digestion process. As shown in Fig. 1, such changes occurred in relatively early stage of digestion. The experiments made in this study were, therefore, not very long (mostly 5–10 days, as shown in Fig. 3). The VSS reduction did not reach the constant level of inert residue (\( \text{VSS}^i \)) to allow for reliable, independent determination of \( f_i \) in Equation (10). Consequently, Equation (8) has been used to fit the VSS reduction results obtained in this study.

The experimental data and best-fit curve (using
SigmaPlot 4.0, SPSS Inc., Chicago, IL) for the run with biosolids from the RBC surface are shown in Fig. 3. The model describes the experimental results well ($r^2 = 0.98$), with the following parameters: $F = 0.64 \pm 0.03$, $k_h = 0.95 \pm 0.19$ d$^{-1}$, and $k_n = 0.067$ d$^{-1}$. Assuming the same average decay constant for the nitrifying population in the same plant, $k_n$ has been pre-estimated from the slope of the second-phase digestion in the run shown in Fig. 1. The longer second phase there gives a more reliable determination. The pre-assignment of $k_n$ also decreased the number of parameters (VSS$^0$, $F$, $k_h$, and $k_n$) to be fitted in the nonlinear regression, reducing the parametric dependencies of the fitting.

As shown in Equation (9), $F$ depends on the initial population fractions, the decay constants of heterotrophs and nitrifiers present, and the interaction constant ($k_i$) between the two groups. Inserting the values of $k_h$ and $k_n$ in Equation (9),

$$F = f_h(1 - 1.08k_i)$$

$f_h$ and $k_i$ cannot be evaluated separately from the experimental results. Nonetheless, the value of $F$ approaches the lower limit of the heterotrophic fraction ($f_h$) in the initial sludge. A value of $F = 0.64$ suggests the existence of significant nitrifying population (up to 36%) in the plant’s RBC biofilm. Although the quantitative measure

**Table 1.** Values of $F$ and the decay constants for heterotrophs and nitrifiers.

<table>
<thead>
<tr>
<th>Run No.</th>
<th>Sludge</th>
<th>$F$</th>
<th>$k_h$ (d$^{-1}$)</th>
<th>$k_n$ (d$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>RBC biofilm</td>
<td>0.64 ± 0.03</td>
<td>0.95 ± 0.19</td>
<td>0.067</td>
</tr>
<tr>
<td>2</td>
<td>Mixed sludge</td>
<td>0.39 ± 0.04</td>
<td>0.55 ± 0.10</td>
<td>0.067 ± 0.007</td>
</tr>
<tr>
<td>3</td>
<td>Mixed primary</td>
<td>0.56 ± 0.10</td>
<td>0.48 ± 0.17</td>
<td>0.067</td>
</tr>
<tr>
<td>4</td>
<td>Mixed and secondary</td>
<td>0.59 ± 0.10</td>
<td>0.50 ± 0.17</td>
<td>0.067</td>
</tr>
<tr>
<td>5</td>
<td>Mixed sludge</td>
<td>0.74 ± 0.19</td>
<td>0.48 ± 0.29</td>
<td>0.067</td>
</tr>
</tbody>
</table>

The decay constants of heterotrophs ($k_h$) obtained for runs 2–5 with the mixed sludge were apparent values in the presence of nonbiomass VSS.

The decay constant of nitrifiers ($k_n$) was obtained from run 2 with much longer second-phase decay. The same value was assumed in other runs for evaluation of $k_h$. 
requires further examination, the finding was consistent with the good nitrifying activity observed in the plant operation (Hall, personal communication, 1998).

Also shown in Fig. 3 are the experimental results and best-fit curves, based on Equation (8), of VSS reduction for the four runs made with mixed primary and secondary sludge. The two-phase digestion of the mixed sludge appeared to be fitted well ($r^2 \geq 0.98$, except one run with $r^2 = 0.86$ due to rather scattering early data), despite the fact that Equation (8) is derived without considering the presence of nonbiomass VSS or their support of growth/maintenance of heterotrophs and nitrifiers. Krishnamoorthy and Loehr (1989) also observed similar digestion profiles between the secondary and the mixed sludge, differing only by the smaller decay constants of the mixed sludge. (On the other hand, the digestion of primary sludge alone proceeded very differently, with a much slower, relatively linear initial decay followed by the typical exponential profile.)

The best-fit parameters obtained in this study are summarized in Table 1. As explained earlier, the value of $k_n = 0.067$ d$^{-1}$ has been assigned to all the runs. The best-fit values of apparent $k_h$ (in the presence of nonbiomass VSS) obtained from different runs of mixed sludge were very comparable (0.48–0.55 d$^{-1}$), supporting the applicability of this model. They are smaller than the $k_h$ for the biosolids from RBC biofilm, consistent with the literature report of slower digestion of mixed sludge than the secondary sludge (Krishnamoorthy and Loehr, 1989).

The (apparent) decay constant for heterotrophs ($0.5–1.0$ d$^{-1}$) was found an order of magnitude larger than that for nitrifiers (0.067 d$^{-1}$). This may explain the literature findings of far larger decay constants for sludge from low sludge-age processes (e.g., 0.2–0.5 d$^{-1}$ for sludge age of 3–4 d) than those from long nitrifying processes (e.g., 0.025–0.065 d$^{-1}$ for sludge age of 15–20 d) (Krishnamoorthy and Loehr, 1989; Kim and Hao, 1990). The latter are expected to have much higher nitrifier population than the former.

$F$ varied from 0.39 to 0.74 for different runs of mixed sludge. Because of the presence of nonbiomass VSS, the values cannot be easily correlated with the population factions as in Equation (9). Nonetheless, the variation of $F$ probably reflects the different initial properties of the sludge used in the laboratory experiments. The plant mixes the primary and secondary sludge in a pit and transfers the mixed sludge to the digester in three shifts every day. To simulate the plant’s operation, the samples for this study were all taken from the pit right before the sludge transfer in the morning (typically around 8:30 A.M.). However, the sludge properties would not be constant, especially the ratio of primary to secondary sludge as well as the sludge retention times (and, thus, the extent of digestion already occurred) in the clarifiers and the waste pit.

CONCLUSIONS

The aerobic decay of the RBC biosolids as well as the mixed primary and secondary sludge exhibited a two-phase behavior. Each phase followed different first-order kinetics, with an intermediate transition. The two-phase phenomenon can be successfully modeled by considering the decay and interaction of the heterotrophs and nitrifiers present in the sludge. The best-fit average decay constants were 0.067 d$^{-1}$ for nitrifiers and 0.95 d$^{-1}$ for heterotrophs in the RBC biosolids. The apparent heterotrophic decay constants (in the presence of nonbiomass VSS) were found 0.48–0.55 d$^{-1}$, consistent among different runs of mixed sludge digestion.

The two-phase digestion was also consistent with the on-line fluorescence profile monitored by the NAD(P)H fluorometer. The fluorescence showed an initial rise to a high plateau, a sharp decline after staying at the plateau for 30–50 h, and a trailing very slow decrease. The initial plateau corresponded to the first-phase of rapid sludge digestion; the trailing fluorescence decline corresponded to the second phase of slow digestion. The transition between the two phases occurred during the sharp decline from the plateau. The fluorescence profile was consistent with our earlier observation of much lower NAD(P)H fluorescence from nitrifiers than that from heterotrophs.

For vector attraction reduction, the EPA regulation requires greater than 38% of VS reduction or, among other alternative tests, SOUR $<1.5$ mg O$_2$/g TS-h. As indicated in Fig. 1, these criteria were met when the on-line fluorescence completed the initial plateau stage, suggesting a clear potential of using the fluorometer to monitor the aerobic sludge digestion process for regulatory purposes.

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ONLINE FLUORESCENCE SHOWING TWO-PHASE SLUDGE DECAY


