# Settling and Dewatering Characteristics of an A-stage Activated Sludge Process Proceeded by Shortcut Biological Nitrogen Removal

**Mark W Miller[1]\*, Jon De Armond[2], Matt Elliott[3], Maureen Kinyua[4], Dave Kinnear[5], Bernhard Wett[6], Sudhir Murthy[7] and Charles B Bott[8]**

[1]*Brown and Caldwell, Charlotte, North Carolina, USA*

[2]*Carollo Engineers Inc., Florida, USA*

[3]*Civil and Environmental Engineering Department, Old Dominion University, Norfolk, Virginia, USA*

[4]*Earth and Environmental Engineering Department, Columbia University, New York, USA*

[5]*HDR Inc., Charlotte, North Carolina, USA*

[6]*ARA Consult GmbH, Innsbruck, Austria*

[7]*DC Water and Sewer Authority, Washington DC, USA*

[8]*Hampton Roads Sanitation District, Virginia, USA*

**\*Corresponding author:** Mark W Miller, Brown and Caldwell, 309 East Morehead Street, Suite 160, Charlotte, North Carolina 28202, USA, Tel: +1-704-373-7131; **E-mail:** mmiller1@brwncald.com

## Abstract

The waste solids settling and dewatering characteristics of an A/B process pilot study were evaluated using standardized characterization methods, like sludge volume index and zone settling velocity, and recently developed characterization methods that measure settleability and dewaterability. The test results indicated that the A/B process had similar settling, thickening, and dewatering characteristics when compared to four full-scale water resource recovery facilities. Sludge volume index values of 85 ± 26 mL g$^{-1}$ over an 18-month operating period were obtained in the pilot A-stage. The centrifuge cake solids obtained from undigested A/B pilot waste activated sludge dewatered to an average value of 25.7% dry solids with a range of 23.2-28.0%. Settling, thickening, and dewatering performance of the undigested A/B waste sludge indicated that the A/B process is a viable process from a solids handling standpoint with similar characteristics to full-scale single-sludge activated sludge processes.

**Keywords:** A-stage; Carbon capture; Dewatering; High-rate Activated sludge; Settling

## Introduction

Many water resource recovery facilities (WRRFs) have shifted their focus from just treating wastewater to reducing their physical footprint and external resource consumption (i.e., supplemental carbon, energy, and alkalinity), all while striving to obtain energy neutral operation and meet stringent discharge standards. However, many WRRFs must also remove macronutrients (i.e., nitrogen and phosphorus), which is generally resource intensive and may require external carbon addition if sufficient influent organic carbon is not available for complete denitrification or biological phosphorus removal (BPR). In an effort to reduce the resource consumption of biological nitrogen removal (BNR) processes, several research groups have been developing shortcut BNR processes that attempt to maximize autotrophic nitrogen removal *via* the partial nitritation and anaerobic ammonia oxidation (anammox) pathways [1-5]. Unlike conventional BNR systems, these processes typically require low influent organic carbon loads thus providing the opportunity to maximize carbon capture upstream for energy recovery *via* biogas production. One carbon redirection process that has been successfully utilized for this purpose is the A-stage of the two-sludge adsorption/ bio-oxidation (A/B) process. The A-stage is a high-rate activated sludge (HRAS) process operated at a solids retention time (SRT) less than 1 day and approximately a 30-minute hydraulic residence time (HRT) resulting in 50-70% chemical oxygen demand (COD) capture and removal with low aeration energy input and minimal COD oxidation [6,7]. However, the low SRT operation of the A-stage results in the A/B process having approximately 10% higher overall sludge production when compared to single-sludge BNR processes with primary sedimentation [8]. Since solids handling and disposal represents a significant portion (10-30%) of WRRFs' operational and maintenance costs [9,10], it is critical that the solids handling characteristics, like settleability, thicken ability, and dewaterability, are well understood to reduce these costs.

While the A/B process has been well established in Europe since the 1980s, there is relatively little scientific literature available on the A-stage waste sludge characteristics. The majority of the data that has been published on the A/B process has been limited to overall performance, like COD and total suspended solids (TSS) removal efficiencies and general observations about sludge settleability [6,11,12]. For example, the A-stage has been reported to have low sludge volume index (SVI) values in the range of 40-90 mL g$^{-1}$ indicating good sludge settling characteristics [6]. However, A-stage processes are designed to remove only approximately 70% of the influent COD and therefore the effluent from these processes till contain slowly and nonsettleable solids that are not characterized by the SVI test since the SVI test is only a measure of settleability of the solids that actually settle. Poor settling sludge or undersized sedimentation processes can also pose a risk of solids escaping from the intermediate solids separation processes. The effluent solids of the A-stage can represent a significant organic load on the downstream shortcut BNR process and thus limit autotrophic nitrogen removal by increasing the heterotrophic bacterial population, which in turn compete with ammonia oxidizing bacteria (AOB) for dissolved oxygen and anammox bacteria for nitrite ultimately reducing nitritation rates and the potential for autotrophic nitrogen removal [13].

Settleability can also affect thickening processes limiting the obtainable thickened solids concentration and thus volume of sludge sent to dewatering or digestion processes, both of which typically perform more efficiently at higher solids concentrations. The volume of sludge also determines solids handling equipment and process sizing. Anaerobic digesters are sized according to the volume of sludge treated and therefore can be reduced in size if the solids thicken well resulting in significant capital and operating expenditure savings. Based on full-scale experience, Böhnke [6] stated that A-stage waste sludge easily thickened by gravitational settling to 6-8% total solids (TS). However, no indication as to why A-stage sludge thickened nearly as well as primary solids (5-10% TS) was given [14]. The dewaterability of A-stage sludge, both before and after anaerobic digestion, has not been reported in the literature. Presumably, A-stage waste sludge contains low concentrations of extracellular polymeric substances (EPS) that are known to negatively affect settleability, thicken ability, and dewaterability by increasing sludge viscosity and porosity resulting in decreased biofloc densities [7,15-17]. This is due in part to the hydrophilic nature of some EPS components and their ability to bind large quantities of water that is difficult to remove by mechanical dewatering [15]. Jimenez et al. [7] found that a high-rate activated sludge process operated below a 0.5-day SRT produced EPS concentrations less than 50 mg COD g$^{-1}$volatile suspended solids (VSS) while SRTs greater than 2 days resulted in greater than 100 mg COD g$^{-1}$VSS EPS concentrations. However, like other studies that have investigated the EPS content of HRAS processes operated below a 1-day SRT [18-20], the focus of the study was on COD capture and removal efficiencies and not the solids handling characteristics of the HRAS waste sludge.

The lack of literature and design standards for A-stage intermediate clarifiers and solids handling processes associated with A/B processes has created barriers to the full-scale implementation of the A-stage process, particularly in North America. Additionally, these barriers may also affect the adoption of shortcut BNR processes that rely on the A-stage process. Therefore, the objective of this research was to investigate the settleability, thicken ability, and dewaterability of waste sludge produced by apilot-scale A/B process treating municipal wastewater. Sludge settling and dewatering characteristics were evaluated using standardized characterization methods, like SVI and zone settle velocity (ZSV), in addition to novel settling and dewaterability tests. These results were then compared to the results obtained using the same methodologies from several full-scale activated sludge processes treating the same or similar municipal wastewater. These comparisons were used to determine the differences in sludge characteristics from established single-sludge activated sludge processes. This work represents the first study to document the settling, thickening, and dewatering characteristics of sludge produced by an A/B process and should provide guidance on designing the solids separation and solids handling portion of the A-stage process.

## Materials and Methods

### A/B pilot configuration and operation

The pilot continuously received municipal raw wastewater influent (RWI) from the Chesapeake-Elizabeth Wastewater Treatment Plant (CE WWTP) after mechanical screening (6 mm) and grit removal. Additional grit and scum removal and screening (2.4 mm openings) was performed in the pilot to prevent clogging and reduce maintenance issues in the pilot process. The RWI temperature was then adjusted to a user set point between 15-25°C using submersible heaters (OEM OTS, Minneapolis, MN) or a water chiller (Aqualogic MT-9, San Diego, CA).

The A-stage pilot consisted of three vertical, complete-mix bioreactors in series followed by an intermediate clarifier and an effluent storage tank that served as a sample collection point as shown in Figure 1. The total working volume of the A-stage bioreactors was 0.51 m³ and each

bioreactor had a side water depth of 3.4 m. The intermediate clarifier had a working volume of 1.7 m³ with a surface overflow rate of 17 m³m$^{-2}$d$^{-1}$ at the design influent flow of 17 L min$^{-1}$. The HRT of the bioreactors and clarifier was 30 minutes and 1.7 hours, respectively. Aeration was provided using compressed air through a single mechanically operated valve to fine-pore membrane disc diffusers (17.8 cm diameter) mounted on the bottom of each bioreactor as shown in Figure 1. Dissolved oxygen (DO) was maintained at 0.5 mg O₂ L$^{-1}$ using a DO set point controller and DO sensor (InsiteIG Model 10, Slidell, LA) mounted in the last bioreactor (tank 3). The RWI and return activated sludge (RAS) flows were flow-paced using progressive cavity pumps (Seepex BW5, Bottrop, Germany) with variable frequency drives and magnetic flow meters (Rosemount 8705, Houston, TX). A-stage and B-stage return activated sludge (RAS) flow rates were maintained at 100% of their respective influent flow. Waste activated sludge (WAS) was removed from the underflow of the clarifier using a digital, speed-controlled peristaltic pump (Master flex L/S, Vernon Hills, IL). The total SRT, considering only the mass of solids in the bioreactors and the effluent suspended solids, was maintained between 0.1-0.3 days.

The B-stage pilot consisted of four equal volume bioreactors in series with a combined working volume of 0.60 m³ followed by a clarifier with a working volume of 0.33 m³ and surface overflow rate of 6 m³m$^{-2}$d$^{-1}$ at the design flow of 1.9 Lmin$^{-1}$. To control the B-stage SRT, which varied between 4-10 days, hydraulic wasting from the last bioreactor was performed. Aeration was provided using compressed air through mechanically operated valves to fine-pore membrane disc diffusers (17.8 cm diameter) mounted on the bottom of each bioreactor as shown in Figure 1. A detailed description and overview of the instrumentation, automation, and control strategies for the B-stage pilot are covered in Regmi et al. [3,21].

### Full-scale WWTPs evaluated in this study

Sludge samples from various full-scale treatment facilities treating municipal wastewater were collected and subjected to the same measurements as the A/B pilot. This provided the ability to directly compare pilot to full-scale results using the same standardized methods. Brief descriptions of each full-scale facility included in this study are contained in Table 1. All of these facilities are owned and operated by the Hampton Roads Sanitation District (HRSD) located in Southeastern Virginia, USA. These facilities were selected because they had either very similar RWI characteristics or process configurations to the pilot study or to compare single-sludge processes with primary sedimentation to the two-sludge pilot process.

### Pilot study and full-scale WWTPRWI characteristics

The HRSD collection network is over 90% pressurized; therefore, the pilot RWI was characteristically septic and of moderate strength. The
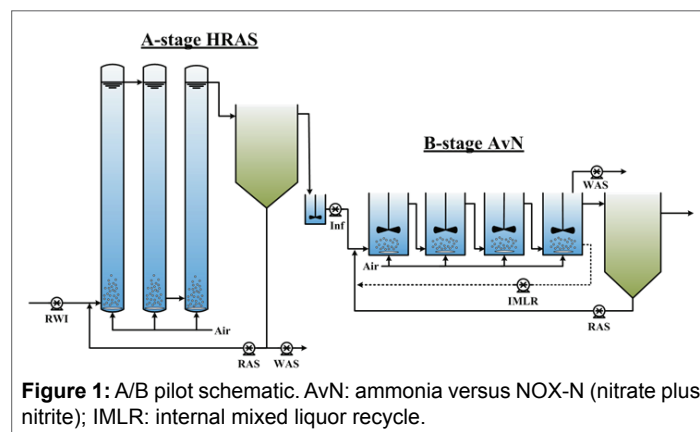


**Figure 1:** A/B pilot schematic. AvN: ammonia versus NOX-N (nitrate plus nitrite); IMLR: internal mixed liquor recycle.

average A-stage influent and effluent characteristics and the removal efficiency of each are included in Table 2. The A-stage effluent represents the influent of the B-stage pilot. Table 3 contains the annual average RWI characteristics, design flow, and annual average sludge age of the activated sludge processes for the full-scale facilities that were evaluated in this study. This data represents operational data collected over one year as part of each facilities sampling plan.

### Settling characterization methods

Settling performance was evaluated by a variety of methods. Standard Method 2710D [22] was used to measure the SVI. While this method is a commonly used measurement of settling performance at treatment facilities, the SVI results are affected by several factors, such as the mixed liquor suspended solids (MLSS) concentration, test apparatus depth and diameter, and intentional or inadvertent mixing during the test [23]. For this reason, Standard Method 2710E [22] for zone settling velocity (ZSV) was used as the benchmark measure of settleability in this work. A solids flux analysis (SFA) was performed on the CE, AT, and VIP facilities and the A-stage pilot. A SFA is conducted by performing multiple ZSV tests using the same sample of sludge diluted to different initial MLSS concentrations to yield a hindered settling velocity curve plotted as hindered settling velocity versus initial solids concentration. Using this curve, Vesilind settling parameters, V0 and k, were estimated by fitting the Vesilind exponential model (Equation 1) to the experimental data by minimizing the sum of the squared error between the model settling velocities and the measured settling velocities.

$$V = V_0 * e^{-k*X} \qquad (1)$$

Where $V_0$ (mhr$^{-1}$) is the initial hindered settling velocity, k (m$^3$ kg$^{-1}$) is the settling coefficient, X (kg m$^{-3}$) is the solids concentration, and V (m hr$^{-1}$) is the settling velocity at the solids concentration of interest. From these results, a solids flux curve due to gravity (SFg) was constructed by multiplying V and X and plotting against X.

### Settling column tests

To determine the effect of MLSS on COD and TSS removal the settling column method, described by Ramalingam et al. [24], was modified to include flocculent settling by using samples with higher MLSS concentrations. The settling column consisted of a clear cylinder (87 cm) with an Imhoff cone affixed to the bottom of the column. A drain port was located where the Imhoff cone and column met. The column was filled with effluent (5.8 L) from the process of interest and a 0.3 L sample of mixed liquor with a known TSS concentration was added to the top of the cylinder using an open bottom container. This container allowed the mixed liquor sample to be added to the top of the cylinder without inducing hydraulic currents that would result from pouring the sample into the column. The solids were allowed to settle for a predetermined amount of time and then the top of the cylinder was drained using the drain port. Settled solids in the Imhoff cone were removed by draining the cone and quantified by measuring TSS. The fraction of solids that settled during the test was subtracted from the total amount of solids added to the column at the start of the test accounting for the effluent TSS.

Three different settling times were used that corresponded to three distinct settling velocities. Large particles were defined as particles with a settling velocity >6 m hr$^{-1}$. Medium size particles were defined as particles that settled slower than 6 m hr$^{-1}$ but faster than 1.5 m hr$^{-1}$. Finally, small particles were defined as particles with settling velocities <1.5 m hr$^{-1}$. The fraction of solids that did not settle during the 1.5 m hr$^{-1}$ test was considered nonsettleable.

| WRRF Name | Abbreviation | Liquid Treatment | Solids Treatment |
|---|---|---|---|
| A/B Pilot Study | A/B | A/B process | None |
| Atlantic | AT | PST and HRAS with anaerobic selectors | Phased acid/gas AD |
| Boat Harbor | BH | CEPT and A/O | Incineration |
| Chesapeake-Elizabeth | CE | HRAS (no PST) | Incineration |
| James River | JR | PST and MLE IFAS (media in aerobic zones) | Mesophilic AD |
| Nansemond | NP | PST and 5-stage Bardenpho | Mesophilic AD |
| Virginia Initiative Plant | VIP | PST and VIP process | Incineration |

**Table 1:** Description of full-scale treatment facilities evaluated in this study PST: Primary settling tanks; AD: Anaerobic digestion; CEPT: Chemically enhanced primary treatment; A/O: Anaerobic/oxic; MLE: Modified Ludzack-Ettinger; IFAS: Integrated fixed-film activated sludge

| Parameter | Influent | Effluent | % Removal |
|---|---|---|---|
| Total COD (mg L$^{-1}$) | 551 ( ± 74) | 301 ( ± 64) | 45 (± 12) |
| Soluble COD; sCOD (mg L$^{-1}$) | 217 ( ± 30) | 144 ( ± 31) | 33 ( ± 12) |
| Particulate COD; pCOD (mg L$^{-1}$) | 335 ( ± 67) | 157 (± 45) | 52 ( ± 16) |
| Total Suspended Solids; TSS (mg L$^{-1}$) | 201 ( ± 41) | 98 ( ± 26) | 50 ( ± 16) |
| Volatile Suspended Solids; VSS (mg L$^{-1}$) | 178 ( ± 35) | 84 ( ± 21) | 50 ( ± 17) |
| Total Kjeldahl Nitrogen; TKN (mg-N L$^{-1}$) | 42 (± 5) | 38 ( ± 4) | 13 ( ± 7) |
| Total Phosphorus; TP (mg-P L$^{-1}$) | 5.7 ( ± 0.8) | 4.4 ( ± 0.8) | 23 ( ± 12) |

**Table 2:** Average measured (n>200) influent and effluent wastewater characteristics and removal efficiencies of the A-stage pilot. Mean ( ± standard deviation)

| Parameter | AT | BH | CE | JR | NP | VIP |
|---|---|---|---|---|---|---|
| COD (mg L$^{-1}$) | 507 ( ± 37) | 379 ( ± 70) | 497 ( ± 35) | 547 ( ± 18) | 469 ( ± 28) | 366 ( ± 50) |
| sCOD (mg L$^{-1}$) | 256 ( ± 21) | 180 (± 48) | 246 ( ± 16) | 281 ( ± 8) | 229 ( ± 18) | 191 ( ± 34) |
| pCOD (mg L$^{-1}$) | 251 ( ± 22) | 199 ( ± 30) | 251 ( ± 27) | 266 ( ± 27) | 241 ( ± 32) | 175 ( ± 26) |
| TSS (mg L$^{-1}$) | 168 ( ± 13) | 136 ( ± 20) | 161 ( ± 12) | 174 ( ± 17) | 161 ( ± 10) | 117 ( ± 13) |
| TKN (mg-N L$^{-1}$) | 44 ( ± 2) | 31 ( ± 5) | 41 ( ± 2) | 39 ( ± 4) | 41 ( ± 3) | 28 ( ± 4) |
| TP (mg-P L$^{-1}$) | 6.0 ( ± 0.5) | 3.8 ( ± 0.8) | 5.1 ( ± 0.3) | 5.1 ( ± 0.5) | 7.4 ( ± 0.8) | 4.6 ( ± 0.7) |
| Design Flow (m$^3$sec$^{-1}$) | 2.37 | 1.10 | 1.05 | 0.88 | 1.31 | 1.75 |
| SRT (days) | 2.4 ( ± 0.2) | 7.6 ( ± 4.9) | 3.2 ( ± 0.9) | 5.0 ( ± 0.6) | 12.5 ( ± 2.1) | 9.9 ( ± 1.7) |
| SVI (mL g$^{-1}$) | 93 ( ± 30) | 136 ( ± 39) | 107 ( ± 52) | 146 ( ± 21) | 76 ( ± 16) | 86 ( ± 15) |

**Table 3:** Average RWI (n>48) concentrations and activated sludge operational parameters of the full-scale facilities evaluated in this study. Mean ( ± standard deviation).

## Dewaterability characterization methods

To determine the obtainable dewatered cake solids from the pilot and full-scale treatment facility sludge samples, a standardized dewaterability test described by Higgins et al. [25] was used. This method provided the ability to compare different sludges using the same dewatering apparatus instead of relying on facility operational data, which varies based sludge conditioning and dewatering equipment. Sludge samples collected from biological phosphorus removal facilities were transported to the laboratory under aeration to prevent phosphorus release. First, the optimalcationic polymer (Zetag 7583, BASF) dose was determined by dosing increasing polymer concentrations and measuring capillary suction time (CST) according to Standard Method 2710G [22]. The optimal polymer dose was selected as the dose that resulted in the lowest average CST when ran in triplicate. The sludge sample with the optimal polymer dose was then partially dewatered by gravity on belt filter press fabric and then transferred to centrifuge cups. These cups were custom fabricated and contained an apparatus that supported a layer of belt filter press fabric and allowed filtrate to accumulate in the bottom of the cups. The samples were centrifuged at 3000X g for 10 minutes. The percent dry solids (DS) content of the dewatered cake was then measured according to Standard Method 2540G [22].

## Analytical methods and data analysis

Performance of the A/B pilot was assessed by collecting 24-hr flow-weighted composite samples of each influent and effluent and analyzing for total COD, soluble COD (1.5 µm glass microfiber filtered), TSS, VSS, TKN, and TP according to Standard Methods [22]. Particulate COD was calculated as the difference between total COD and sCOD. Filamentous bacteria were identified in mixed liquor samples by microscopic enumeration according to Jenkins et al. [26].

Statistical analyses, including the Pearson product moment coorelation (R), Shapiro-Wilk normality test, t-test, linear regression ($R^2$), mean, standard deviation (SD), standard error of the mean (SE), one-way analysis of variance (ANOVA), and confidence intervals (CI) were performed using Sigma Plot 12.5 (Systat Software Inc., Bangalore, India). Confidence intervals were calculated at a p-value of 0.05.

## Results and Discussion

### Average A-stage pilot operation and performance

The A-stage of the A/B pilot was operated at a 30-minute HRT and the SRT was maintained between 0.1-0.3 days accounting for effluent TSS and excluding solids present in the intermediate clarifier. As seen in Table 2, TSS and VSS removal averaged 50 ± 16% (mean ± standard deviation) and 50 ± 17%, respectively. Suspended solids removal was lower than what is achieved by primary sedimentation alone, which is typically 60-70% [14]. Although not explored in detail in this work, a reasonable explanation for this was that soluble substrate was consumed for microbial growth resulting in the production of particulate and colloidal biomass. Since the A-stage was operated at such low sludge ages, the newly produced biomass did not completely aggregate into settleable flocs resulting in lower TSS removal efficiencies. Bisogni and Lawrence [27] referred to this phenomenon as dispersed growth and found that it occurred at SRTs less than one day in complete-mix activated sludge processes.

Total COD removal efficiency of the A-stage pilot averaged 45 ± 12%, which was lower than the reported performance of full-scale A-stage processes that remove 55-75% of the influent COD [28]. The lower COD removal performance was intentional in the pilot A-stage and maintained via SRT manipulation since COD removal was controlled in order to optimize nitrogen removal in the B-stage pilot [3]. The B-stage was operated aggressively in terms of SRT in order to achieve shortcut

nitrogen removal and this required a relatively stable influent COD/N ratio between 6-7. As discussed in Regmi et al. [3] too much COD would reduce nitrification rates and too little would limit overall nitrogen removal.

Total phosphorus and TKN were removed by assimilation and sedimentation in the A-stage and averaged 23 ± 12% and 13 ± 7%, respectively. Total phosphorus was also removed by chemical precipitation since the CE facility dosed ferric chloride prior to preliminary treatment (i.e., upstream of pilot RWI intake) for most of the year to control odors.

### Impact of A-stage operation on mixed liquor settling

A-stage settling was quantified by routinely measuring the SVI of the mixed liquor. Mixed liquor samples were also observed under a microscope on a weekly basis to determine the presence of filamentous bacteria. The pilot A-stage mixed liquor exhibited an average SVI value of 85 ± 26 mL g$^{-1}$ (n=393) over the 600 days of continuous operation. This value was within the range of 38 to 93 mL g$^{-1}$ reported by Böhnke [6] for pilot- and full-scale A-stage processes. During the 600 days of operation, there were less than 10 days when the SVI was above 150 mL g$^{-1}$ and the maximum SVI observed was 217 mL g$^{-1}$. Unlike complete-mix HRAS processes that are known to settle poorly when operated at low sludge ages and high organic loadings [27,29,30], the A-stage pilot was configured in a plug-flow configuration to promote well settling sludge by selecting for fast growing flocculent bacteria or r-strategists over the slower growing filamentous bacteria or K-strategists. Additionally, operation at DO concentrations less than 1 mg $O_2$ L$^{-1}$ did not result in poor settling, which is counter to the classical bulking theory presented by Palm and Jenkins J. C. [31] where increased loadings necessitates an increased bulk DO concentration to avoid proliferation of filamentous bacteria.

During periods of bulking in the A-stage pilot, filamentous bacteria Type 1863 were observed in the mixed liquor samples. Thiothrix spp. types I and II were also present but never at an abundance that caused bulking sludge. Type 1863 filamentous bacteria are typically seen in activated sludge processes with high organic loading rates, short sludge ages, and low DO conditions, which are the operational conditions at which the A-stage pilot was operated [26]. Although a statistical analysis did not find a direct correlation between the bulk DO concentration and SVI (n=414, R=-0.05, p=0.24), bulking was generally observed when the DO was less than about 0.1 mg L$^{-1}$. The bulking was attributed to the proliferation of. Type 1863 bacteria, which are known to thrive under low, DO conditions [26]. However, t The SVI of the A-stage did correlate with SRT (n=381, R=0.32, p<0.001) and MLSS (n=242, R=0.24, p<0.001) since these parameters are interrelated and directly affect the bulk DO concentration. That is, as the SRT increases the MLSS concentration also increases resulting in an increase in the total oxygen demand. When the oxygen demand exceeded the oxygen transfer capability of the A-stage aeration system, the DO would fall below 0.1 mg L$^{-1}$ and bulking would soon onset if the bulk DO concentration remained low for periods longer than approximately one day.

### Characterization of settleability using ZSV and settling column tests

To further investigate and compare the settleability of A-stage mixed liquor to other full-scale activated sludge processes, ZSV and settling column tests were performed using mixed liquor samples from the A-stage and the AT, CE, and VIP treatment facilities. The CE facility was selected because its HRAS process was similar to the A-stage pilot (i.e., no PSTs) and received the same RWI as the A-stage pilot. The AT facility was also selected because of its HRAS process except it had PSTs and anaerobic selectors for improved settling by promoting biological phosphorus removal. The VIP facility also has pre-anaerobic zones for biological phosphorus removal but was operated at longer SRTs (9.9 ± 1.7 days) to achieve biological nitrogen removal.

Solids flux analyses were conducted using the ZSV test results to experimentally determine Vesilind settling parameters for the mixed liquor samples from the pilot A-stage, and the AT, CE, and VIP treatment facilities. The average Vesilind parameter results from these tests are shown in Figure 2. The pilot A-stage had a Vesilind $V_0$ of $15.0 \pm 6.3$ m hr$^{-1}$ (mean $\pm$ CI) and k of $0.70 \pm 0.06$ m$^3$ kg$^{-1}$. Comparing the pilot $V_0$ to the other facilities, the pilot A-stage was slightly higher than the CE ($11.3 \pm 3.4$ m hr$^{-1}$) and VIP ($11.2 \pm 6.1$ m hr$^{-1}$) facilities. However, the A-stage k was closer to that of the AT ($0.67 \pm 0.31$ m$^3$ kg$^{-1}$) facility. Apart from the AT facility, $V_0$ values for all of the other mixed liquor samples were within the typical ranges for activated sludge processes (k=0.2-1; $V_0$=5-15) [32]. The AT facility likely had an unusually high $V_0$ because the facility had anaerobic selectors that were specifically designed to improve settling. Using all of the ZSV test results for each process, a Vesilind settling velocity model curve (Equation 1) was fitted to each dataset using the sum of least squares method. The resultant curves are displayed in Figure 3a. Using the predicted $V_0$ and k for each model curve and the actual MLSS data, a solids flux due to gravity (SFg) curve was generated for each facility and the A-stage pilot (Figure 3b).

As indicated previously, the A-stage maximum settling velocity was similar (p=0.591) to CE and VIP. However, the pilot A-stage settling velocity decreased rapidly with increasing MLSS concentration, as was the case with AT mixed liquor. Interestingly, the maximum SFg for the pilot A-stage, CE, and VIP (Figure 3b) occurred at an initial MLSS concentration close to their average operating MLSS concentrations of $1592 \pm 615$, $2710 \pm 265$, $4145 \pm 450$ mg L$^{-1}$, respectively. This suggests that the MLSS concentration at which an activated sludge process is operated and the solids loading and surface overflow rates of the secondary clarifiers may play an important role in determining sludge settleability. That is, the solids separation process selects for solids that have settling properties that directly correlate to the operating conditions. This is analogous to the use of a high settling velocity to select for and retain granules in an aerobic granular sludge system [33]. Moreover, this is likely associated with the solids loading and surface overflow rates of the secondary clarifiers that results in the selection of solids that settle at certain velocities as dictated by the operation of the solids separation process.

One of the limitations of the ZSV test is that at low initial MLSS concentrations it is difficult to discern a discrete solids interface and therefore difficult to estimate settling velocities. Additionally, during the pilot A-stage ZSV tests, it appeared as if two types of solids existed and these solids settled at differential rates. To quantify this observation, settling column tests were performed on A-stage, CE, and VIP using mixed liquor samples with different TSS concentrations. The results from these tests are summarized in Figure 4. Samples from CE and VIP were used to compare the A-stage to a HRAS (short SRT) process and a BNR (long SRT) process for the same reasons discussed previously. For the full-scale facilities, when the settling tests were run at high (>1.5 g L$^{-1}$) initial MLSS concentrations, >80% of the solids were considered large particles that settled at a rate >6 m hr$^{-1}$ with only minor fractions (1-10%) of medium and small particles. However, the A-stage had a high nonsettleable fraction of 36%. This was as expected because the A-stage is bioflocculation limited due to its high-rate operation (i.e., SRT<0.5 days) and is reflected in the effluent as TSS, which averaged $98 \pm 26$ mg L$^{-1}$. Essentially, the high-rate operation precludes the accumulation of adequate quantities of biomass with available adsorption sites.

When comparing the results from the low initial MLSS to the high MLSS tests, the fraction of nonsettleable solids increases for all three mixed liquors. This can be explained by the phenomenon of orthokinetic flocculation where large particles settling at higher rates than smaller particles collide with the small particles resulting in the removal of both particles at a higher net settling velocity [14]. This also explains why

the fraction of large solids that settle at a rate >6 m hr$^{-1}$ increases with increasing MLSS. Another trend is that at longer SRTs the fraction of medium and small particles increases while the fraction of nonsettleable and large particles decreases. This was likely attributed to the conversion of denser primary particles to biological flocs that are more porous and settle at slower rates [27,34].
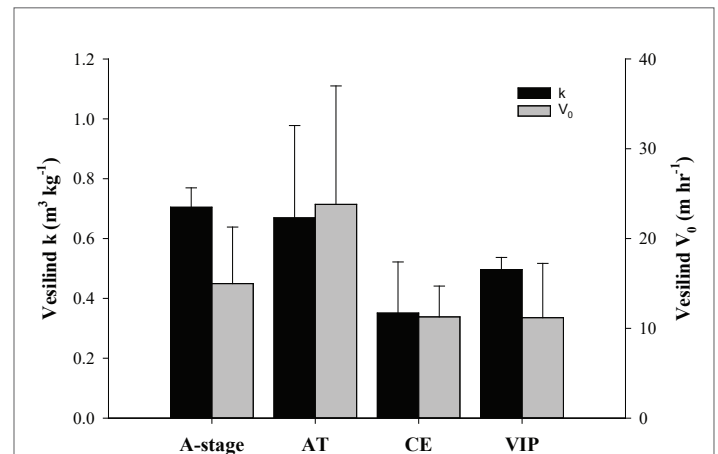


**Figure 2:** Comparison of measured Vesilind initial hindered settling velocity ($V_0$) and settling parameter (k) determined using ZSV tests on mixed liquor samples from the A-stage pilot and AT, CE, and VIP treatment facilities. Error bars represent 95% CI.
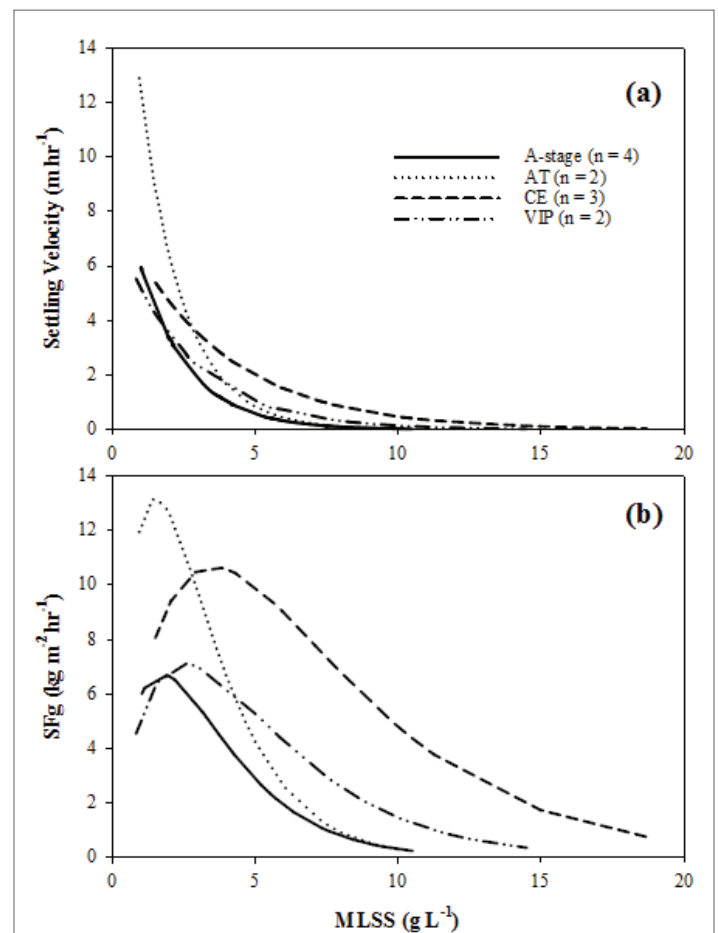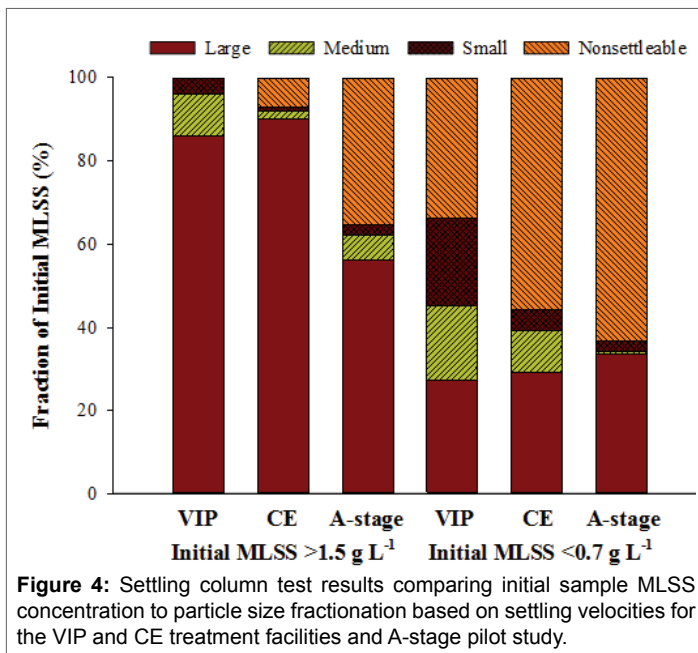


**Figure 3:** Comparison of (a) modeled settling velocity and (b) SFg to initial MLSS concentrations during ZSV tests on mixed liquor samples from the pilot A-stage and AT, CE, and VIP treatment facilities.

**Figure 4:** Settling column test results comparing initial sample MLSS concentration to particle size fractionation based on settling velocities for the VIP and CE treatment facilities and A-stage pilot study.

## Characterization of dewaterability using optimal polymer dose and centrifugation method

To characterize the sludge dewaterability of the A-stage pilot and full-scale facilities, capillary suction time tests at different polymer doses were performed. The optimal polymer dosage, indicated by the lowest CST, for all the sludge's are shown in Table 4. These tests were performed using primary and secondary sludge's from the BH, NP, and VIP treatment facilities. These facilities were selected because they all have primary sedimentation and each facility has different sludge characteristics because of different influent characteristics (Table 3) and process configuration (Table 1). Table 4 also contains the average fraction of primary to secondary solids produced based on one year of operational data for each facility and the A/B pilot. Although the pilot A-stage sludge was not sent to an anaerobic digestion system, analysing the dewaterability of anaerobically digested A-stage sludge is important to provide a full scope of solids handling characteristics. In addition, such a test would provide data for direct comparison with full-scale systems that have anaerobic digesters.

Capillary suction time tests consistently concluded that the A-stage sludge required an optimal polymer dose of 3.9 ± 1.1 g polymers kg$^{-1}$ DS at a CST of 10.1 ± 1.1 seconds. The test results were similar for the full-scale undigested solids streams tested (Table 4). All of the optimal polymer dosages were within the typically ranges for primary (1-4 g kg$^{-1}$) and secondary solids (3-10 g kg$^{-1}$) [14]. Using the optimal polymer doses determined during the CST tests, dewaterability at different blends of primary solids (PS) to secondary solids (SS) for each process was determined as shown in Figure 5. As expected, the results show that PS dewaters better than SS and that a higher blend of PS to SS increases the overall dewaterability of the combined sludge's. This is due to the fact the PS contain less EPS than SS and EPS is known to bind water resulting in increased sludge viscosities [7,15,17]. Houghton et al. [16] used the CST test to demonstrate that EPS benefits dewaterability until around 35 mg EPS g$^{-1}$ TSS for activated sludge. However, PS dewatered better at lower EPS concentrations and did not benefit from increasing EPS concentrations. Although the A-stage is a biological process and produces EPS, the A-stage WAS dewatered (34.3 ± 0.4%; mean ± SEM) nearly as well as the PS for the full-scale facilities (VIP=37.4 ± 1.1%; BH=36.3 ± 0.3%; NP=35.3 ± 0.01%). This was likely attributed to the low EPS content of the

A-stage sludge because of operation at short sludge ages [7] and the fact that lower EPS content typically results in better dewaterability [15-17]. However, further investigation of the EPS production and composition in A-stage process is needed.

The overall dewaterability of the A/B pilot, as indicated by the line of best fit in Figure 5, was less than that of the full-scale facilities. Although, the NP did not a have good linear fit (R2=0.90). When considering that the A/B pilot produced a higher fraction of A-stage to B-stage WAS (Table 4), the dewaterability of the A/B pilot compared well with the full-scale facilities in terms of obtainable cake solids. This can be seen in Figure 6, where dewatered cake solids for each facility were predicted based on the linear relationship (i.e., slope and intercept) between the PS to SS fractions to cake solids. Actual monthly averages for one year of the fraction of PS to SS for each facility were used to predict dewatered cake solids. For the A/B pilot a full year of data was used.

The predicted dewaterability of blended A/B solids was 26.8 ± 0.3% (mean ± CI), which is lower than the VIP and BH facilities that averaged 27.8 ± 0.5% and 29.4 ± 1.0%, respectively. The NP facility was the lowestat23.5 ± 1.4%. If the fraction of solids produced by the A-stage was closer to the typical split of 75% for full-scale A/B facilities [6], the predicted cakes solids would have been slightly higher around 28%. The actual lower fraction of PS to SS of 70 ± 15% was due to the lower COD and TSS removal efficiency of the A-stage pilot since COD removal was being controlled to optimize nitrogen removal in the B-stage pilot. The BH facility had the highest cake solids because of a higher PS to SS fraction

| Facility | Solids Type | CST (secs) | Optimal Polymer Dose (g polymer kg$^{-1}$ DS) | PS to SS Fraction (%) |
|---|---|---|---|---|
| A/B | A-stage | 10.1 ( ± 1.1) | 3.9 ( ± 1.1) | 70 ( ± 15) |
| | B-stage | 10.9 ( ± 1.7) | 4.1 ( ± 0.4) | |
| VIP | Primary | 11.0 ( ± 0.1) | 4.5 ( ± 0.3) | 67 ( ± 9) |
| | Secondary | 11.2 ( ± 0.3) | 4.4 ( ± 1.9) | |
| BH | Primary | 12.0 (± 1.5) | 3.8 ( ± 0.7) | 49 ( ± 5) |
| | Secondary | 11.5 ( ± 1.6) | 3.7 ( ± 1.3) | |
| NP | Primary | 11.1 ( ± 0.5) | 3.4 ( ± NA) | 45 ( ± 15) |
| | Secondary | 10.7 ( ± 0.2) | 2.5 ( ± NA) | |

**Table 4:** Average ( ± standard deviation) CST and optimal polymer dose for the A/B pilot and BH, VIP, and NP facilities.
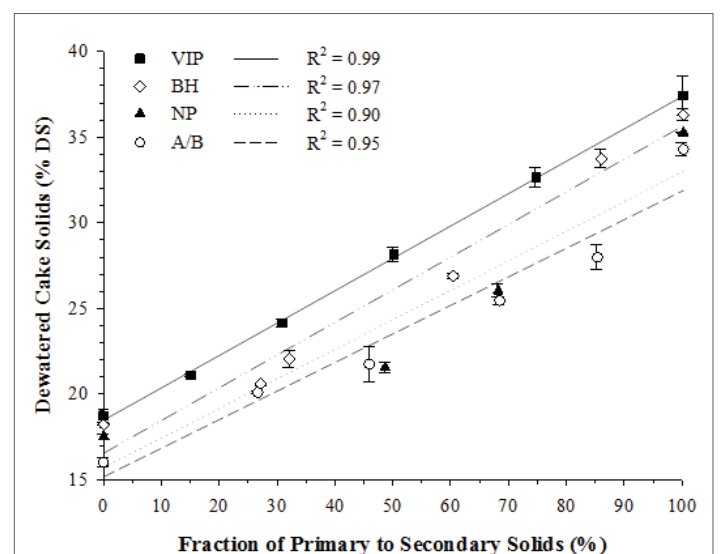


**Figure 5:** Dewatering test results comparing dewatered cake solids to fraction of primary to secondary waste solids on a dry solids basis. Error bars represent standard error of the mean.
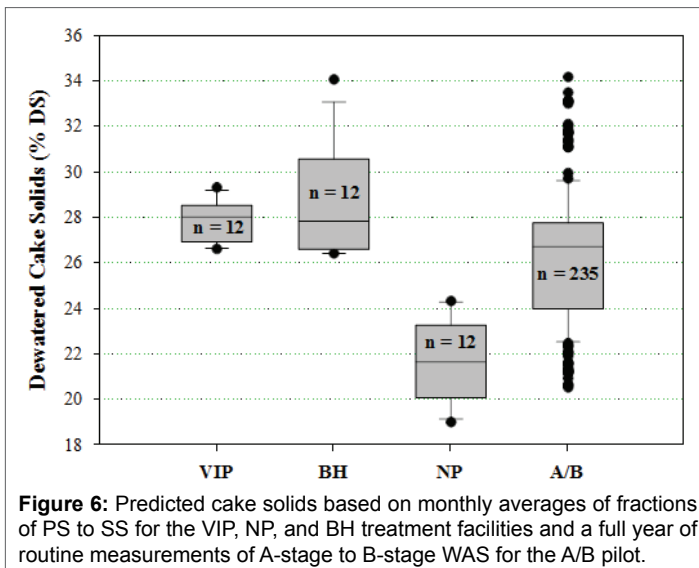
**Figure 6:** Predicted cake solids based on monthly averages of fractions of PS to SS for the VIP, NP, and BH treatment facilities and a full year of routine measurements of A-stage to B-stage WAS for the A/B pilot.

associated with higher solids removal by the chemically enhance primary treatment process and the production of metal precipitates resulting from iron salt addition. The NP had poor dewaterability likely because the facility performs biological phosphorus removal, which has been shown to decrease the dewaterability of digested solids [25].

## Conclusions

The solids produced by the A-stage process exhibited good settling and dewatering characteristics. Settling, thickening, and dewatering performance of undigested A-stage solids indicated that the A-stage activated sludge process is a viable process from a solids handling standpoint. These results are only applicable to undigested solids as they may not be predicative of digested solids since dewaterability changes with digestion. Further research is required to ascertain the dewaterability performance of digested A/B solids.

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