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Review

A review of secondary sludge reduction technologies for the pulp and paper industry

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ABSTRACT

The broader application of the activated sludge process in pulp and paper mills, together with increased production, have amplified sludge management problems. With sludge management costs as high as 60% of the total wastewater treatment plant operating costs, and increasingly stringent environmental regulations, it is economically advantageous for pulp and paper mills to reduce their biosolids production. In order to provide a state-of-the-art review of secondary sludge reduction technologies, we have considered the scenarios of lower sludge production through process modifications, and waste-activated sludge reduction through post-treatment. A critical evaluation of all candidate reduction technologies indicates that sludge reduction through treatment process changes appears more appealing than post-treatment alternatives. The former approach offers a clear advantage over the latter in that the treatment process changes reduce sludge production in the first place, thus decreasing sludge management cost. Although it is technically feasible to eliminate the need for sludge disposal, it is unlikely to be economically feasible at this time.

Contents

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1. Introduction

The broader application of the activated sludge process (ASP) in pulp and paper mills together with increased production has amplified sludge management problems. While the yearly residue production has been and will continue to increase in the foreseeable future, the conventional management methods such as landfilling, incineration and beneficial uses have come under stronger public opposition and stricter regulatory pressure. Landfilling, which is the most common disposal method in current practice (Reid, 1998; Stein et al., 1989), is becoming increasingly more difficult to implement because of rapidly shrinking landfill space, public opposition to opening new landfill sites (Rouleau and Sasseville, 1996), leachate related issues and, above all, poor economics. In many cases the costs of opening a new landfill are prohibitive due simply to legal restrictions (Lagacé et al., 1998). Incineration, which is second to landfill in popularity, suffers from its own drawbacks. Rising supplemental fuel costs, high capital costs and air pollution concerns have historically limited the use of this method. Land application of pulp and paper residues only accounts for less than 5% of the total residues generated (Reid, 1998). Public opposition to this disposition method has limited the widespread use of this method as well. It is apparent that the conventional sludge management methods, which might have been acceptable in the past, can no longer be considered optimal to meet present and future requirements.

With sludge management costs as high as 60% of the total wastewater treatment plant operating costs (Canales et al., 1994) and increasingly stringent environmental regulations, it is to the economic advantage of the pulp and paper mills that they reduce their sludge production. Although the industry has almost eliminated bark waste and significantly decreased wood waste production through process recycling (Stein et al., 1989), the disposal of water treatment residuals continues to be the major solid waste issue at most pulp and paper mills (Nichols, 1992).

Water treatment residues are produced as a result of primary and secondary wastewater treatment. Primary treatment residue is mainly composed of fibres, fines and fillers lost because of incomplete solid/liquid separations at various stages of pulp and paper production. The lost materials, which vary from 3% to 4% for pulp mills to 15–30% for the waste paper mills (Trutschler, 1999), is captured through gravity settling in the primary clarifier and is referred to as primary sludge. The primary sludge generation rate mainly depends upon efficiency of the fibre recovery system installed at a mill. Driven by economics, a large number of mills are optimizing their fibre recovery systems, which will increase their raw material yields and will reduce operating costs, and primary sludge production (Trutschler, 1999). On the other hand, the production of secondary sludge, which is a by-product of biological treatment, is expected to increase for two reasons: (1) increased BOD loading as a result of increased production and (2) improved BOD and suspended solids (SS) removal required by future regulations. The secondary sludge is far more difficult to dewater than the primary sludge and most pulp and paper facilities dewater a mixture of primary and secondary sludges. In general, as the primary to secondary (P/S) ratio decreases, so does the ease of dewaterability. Increased secondary sludge production coupled with increased fibre conservation is expected to worsen the already challenging secondary sludge dewatering process. It seems inevitable from a dewatering standpoint alone that less secondary sludge should be a goal for the near future. A decreased secondary sludge production rate will help to improve or at least maintain the current P/S ratio. Also, a decreased amount of sludge produced will require less polymer addition to aid dewatering. Savings towards polymer costs can be substantial since conditioning chemicals constitute the largest part of a wastewater treatment plant's operating budget (Kenny et al., 1995).

1.1. *Objective*

The objective of this report is to provide a state-of-the-art review of secondary sludge reduction approaches and technologies. The scenarios of lower sludge production through process changes and waste activated sludge (WAS) reduction through post treatment have been considered. The review covers sludge reduction technologies developed not only within the pulp and paper industry but also those developed in the municipal and other industrial sectors. A schematic diagram that outlines many of the available technologies can be found in Fig. 1.

2. Sludge reduction alternatives

Various approaches relying on either one or a combination of physical, chemical and biological principles have been exploited for producing less sludge. While the underlying principles vary widely, the reduction endeavours have mainly targeted either of the following two approaches:

- *Process changes* to lower sludge production by the biological treatment system.
- *Post treatment* of excess WAS to reduce the amount for disposal.

In the former approach, the wastewater treatment plant design and/or operation is engineered to achieve minimized sludge yield (kg VSS produced/kg BOD stabilized). An extended aeration version of the ASP is one such example. On the other hand, excess WAS reduction through post treatment involves sludge oxidation by physical, chemical or biological means. A choice between the two strategies can be situation specific and could require diligent techno-economic evaluations.

2.1. *Sludge reduction through process changes*

The conventional ASP can be modified to a low sludge producing process by making changes in the design and/or operating parameters. The growth rates of bacteria in long sludge-aged systems are generally believed to be lower than those found in short sludge-aged systems. This can be explained by the differences in the allocation of cellular

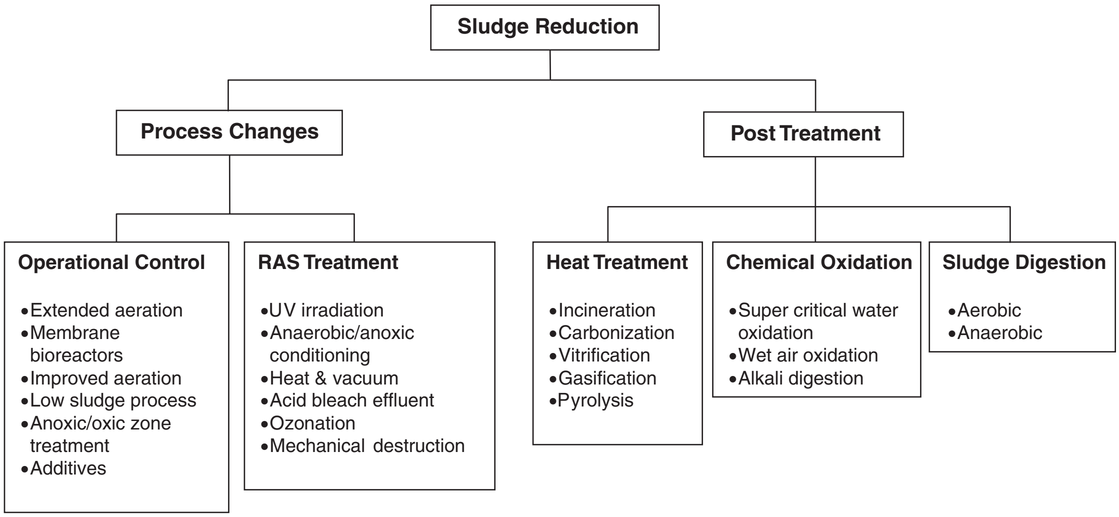

Fig. 1 – Outline of sludge reduction technologies.

energy for cell maintenance, as opposed to cellular reproduction. In the conventional approach for lower sludge production, the mixed liquor suspended solids (MLSS) are simply retained in the aeration basin under endogenous respiration for extended periods of time. One drawback of this approach is that cell solubilization proceeds slowly and thus long solids residence times are required. This limitation could be overcome by using a relatively modern approach in which the return activated sludge (RAS) is treated by physical and/or chemical means before return to the aeration basin. Low and Chase (1999) demonstrated how sludge yield in a dispersed growth ASP could be reduced by as much as 44% by increasing the biomass concentration from 1.7 to 10 g/L while maintaining other operating conditions similar.

2.1.1. *Sludge reduction through operational control*

2.1.1.1. Extended aeration. The extended aeration process is similar to the conventional plug-flow ASP except that it operates in the endogenous respiration mode, which requires long aeration time and a low organic loading (0.16–0.4 kg BOD/m^3 vs. 0.3–0.6 kg BOD/m^3 for the conventional ASP). Theoretically, no excess sludge should be produced in the extended aeration process as the growth rate of the new cells equals the decay rate of the existing cells (Corbitt, 1989). In practice, however, excess sludge is produced but in significantly lower quantities than that in the conventional ASP (Corbitt, 1989). Design data for the conventional ASP and the extended aeration processes treating municipal wastewaters presented in Table 1 delineate differences between the two processes.

A much longer cell residence time and low food/microorganism (F/M) ratio maintains the extended aeration culture in endogenous respiration producing less sludge. However, despite low organic loading, low F/M ratio, and longer hydraulic residence time (HRT) in the aeration basin, the extended aeration systems are typically 5–10% less efficient in BOD removal than the conventional ASP (Table 1), which in certain cases could be undesirable in view of stringent environmental regulations. However, when treating pulp and paper wastewaters, extended aeration systems have been shown to remove up to 99% of incoming BOD (Ericsson et al., 1999; Carlson et al., 2000).

The Springfield Water and Sewer Commission, operating a 67 mgd domestic wastewater treatment facility in Agawam, Massachusetts, converted their conventional ASP to the extended aeration process for reduced sludge production (Borgatti et al., 2000). In this upgrade, the existing sludge reduction Zimpro process (described later) was discontinued. The extended aeration operation reduced sludge production from 715 to 504 dry tonne/month—a 30% decrease. At the current landfill hauling and disposal contract cost of U.S. $310 per dry tonne, savings of U.S.$785,000 per year were realized in addition to savings in sludge dewatering polymer and electricity costs. Moreover, sludge dewatering improved from 21% to 27% solids due mainly to improved P/S sludge ratio. The extended aeration sludge had lower odour potential and better stability because of low respiration rate (Borgatti et al., 2000). The increase in aeration system electricity costs was estimated as U.S. $51,000 per year.

2.1.1.2. Membrane bioreactors (MBRs). An MBR is an alternate approach to achieving reduced sludge production through increased sludge age. In this version of the ASP, the secondary clarifier is eliminated altogether and the sludge/supernatant separation is achieved by using membrane processes, mainly ultrafiltration (UF), directly from the aeration basin. A schematic of the MBR process is shown in Fig. 2.

Nearly perfect solid/liquid separation is achieved in MBRs where the UF completely decouples hydraulic and biomass retention times. This unique character allows MBRs to be operated at very high biomass concentrations and sludge ages approaching infinity can be obtained. While the viability of biomass decreases with an increase in the sludge age (Goma et al., 1997), long sludge residence time (SRT) in MBRs allows proliferation of microorganisms higher in the food chain, which prey on bacterial cells thereby reducing excess biomass production (Ghyoot and Verstraete, 2000). Also, high MLSS in MBRs together with a low F/M ratio promotes cell lysis and cryptic growth and finally low sludge production.

Table 1 – Design parameters for activated sludge and extended aeration processes (Metcalf and Eddy Inc., 1991)

Process	Mean cell residence time (day)	F/M (kg BOD/ kg VSS)	HRT (h)	BOD removal efficiency (%)	Waste sludge[a] (kg/kg BOD removed)	Air supplied[b] (m^3/kg BOD)
Conventional ASP	5–15	0.20–0.40	4–8	85–95	0.40–0.60	45–90
Extended aeration process	20–30	0.05–0.15	18–36	75–90	0.15–0.30	90–125

[a] Corbitt, 1989.
[b] Peavy et al., 1986.

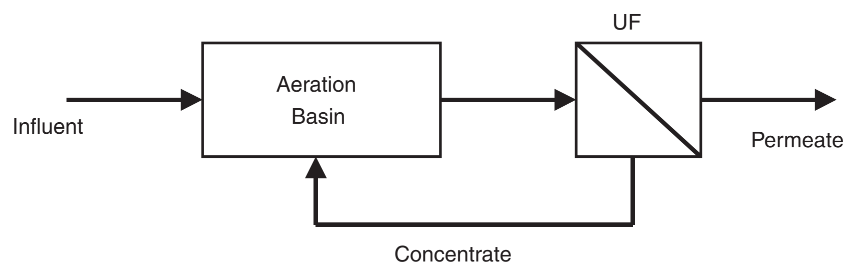

Fig. 2 – Schematic of a membrane bioreactor.

Based on their studies on viability of cells and phenomena of cell death and lysis and growth on intra-cellular products released, Mason and Hamer (1987) concluded that the biodegradation of the cell wall is the rate-limiting step, and to increase it physical and/or chemical treatment methods can be used. Canales et al. (1994) thermally conditioned biomass (90 °C for 3 h) cultured on a synthetic wastewater simulating municipal wastewater and recycled hydrolysates to the MBR, achieving a 2.5-fold decrease in the substrate/ biomass conversion yield.

Ghyoot and Verstraete (2000) combined the MBR concept with the two-trophic level approach (described later) for enhanced sludge reduction. The first stage of the process was a once through completely mixed reactor for bacterial growth. The second (predator) stage was an ASP equipped with 0.2 μm polypropylene hollow fibre membranes for solid/liquid separation. The control reactor was similar to a two-stage membrane-assisted bioreactor with the exception that solid/ liquid separation was realised by sedimentation. Some 80% of the skimmed milk-based feed COD was removed in the bacterial stage at volumetric loading rates up to 1.6 g COD/ L day. Most of the nitrogen and phosphate in the first stage was used for cell synthesis. The membrane-assisted process produced 20–30% less sludge than the control reactor for similar conditions of SRT and organic loading. Reduced sludge production by the membrane bio-reactor was attributed to two main factors: (1) a relatively larger proportion of protozoa (ciliates and flagellates) and metazoa (rotifers and nematodes) retained by the membranes, which prey on the bacteria and (2) a membrane-assisted reactor completely retained bacteria which can be grazed on by predators, which is consistent with the lab-scale findings of Mahmood (1988), who detected no bacteria in the permeate of an MBR using polypropylene hollow fibre membranes of 0.1 μm nominal pore size. In case of gravity based solid–liquid separation, dispersed bacteria could be washed out with the effluent. A high degree of predation by protozoa and metazoa in the membrane-assisted reactor produced higher dissolved nitrogen and phosphate concentrations in the membrane permeate. Excessive higher life forms grazing on nitrifiers decreased the nitrifying capacity of the membrane-assisted system, which can raise ammonia toxicity issues.

A membrane-based ASP (the Kubota process) has been implemented at over 950 industrial facilities worldwide including some within the pulp and paper sector (Churchouse, 2004). In addition to reducing sludge production, it renders a treated effluent that is of sufficient quality that it can be reused for many mill applications.

2.1.1.3. *Improved aeration*. One approach to sludge reduction is to improve aeration to allow for a high MLSS concentration in the aeration basin thus increasing the SRT. Oxygen transfer limitations of the conventional aeration equipment have often restricted the use of this approach as there must be sufficient dissolved oxygen in the system to meet the needs of the bacteria. Larger bacterial flocs, in particular, offer high resistance to oxygen transfer. In order to understand the direct effect of dissolved oxygen concentration in the bulk liquid on excess sludge production, Abbassi et al. (2000) developed a mathematical model taking into account mass transfer of oxygen, biological reactions within flocs and the endogenous respiration process. Model predictions supported with lab experiments showed that the oxygen concentration in the bulk liquid has a significant effect on the amount of excess sludge production. They hypothesized that an increased oxygen concentration in the bulk liquid leads to a

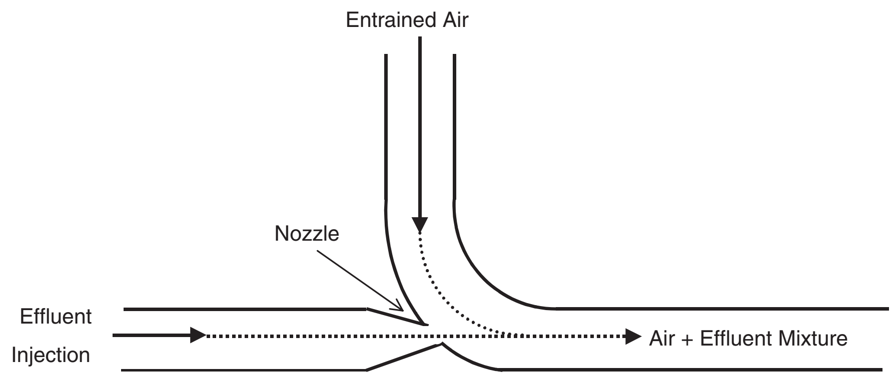

Fig. 3 – Conceptual diagram of a Venturi injector (modified from Young and Barlow, 1998).

deeper diffusion of oxygen into the floc, which enlarges the aerobic volume inside the floc leading to a deficient situation regarding the organic substrate. According to Monod kinetics, a decreased substrate concentration puts more emphasis on cellular maintenance leading to an inferior growth rate and a lower sludge production. The formation of larger bacterial aggregates also negatively impacts sludge reduction in mixed cultures where prey and predators co-exist (Lee and Welander, 1996a).

Relatively recently, modified designs of the aeration equipment have been proposed (Young and Barlow, 1998; Grom, 1997). Such devices create high shear in the air-mixing zone, which reduces the air bubble size and deflocculates sludge flocs. Decreased bubble and floc sizes create high surface area to volume ratio for more efficient oxygen and nutrient uptake (Young and Barlow, 1998). A conceptual diagram of a Venturi aerator is shown in Fig. 3, which comprises a nozzle that injects wastewater into a barrel. The entrained air, because of high shear, as it meets the influent, is dispersed into very small bubbles. The small bubble and floc sizes facilitate oxygen transfer, as up to 40% of the air has been claimed to be solubilized as a result (Young and Barlow, 1998).

An 8-month pilot trial using domestic wastewater showed that the extensive aeration was harmless to protozoa, which is an important element for reducing excess sludge production. No sludge was wasted during the trial and the MLSS was maintained between 6500–11,600 mg/L at an SRT of 50 days. The process was predicted to be extremely cost effective, at less than \$0.03 U.S./m^3, for both domestic and industrial wastewaters (Young and Barlow, 1998), with no mention of the capital cost of the equipment. A similar process called high efficiency compact reactor (HCR) was used at the Norske Skogindustrier AS's integrated mill in Honefoss, Norway. Operating at MLSS levels of 8000–10,000 mg/L, low sludge production rates of 0.15–0.2 kg TSS/kg COD were obtained (Grom, 1997).

2.1.1.4. Low sludge production (LSP) process. In this approach the ASP is modified to a two-stage process to establish a microbial food chain that would result in reduced sludge production. The first stage, because of short solids residence time (3–5 h) and no sludge recycle, selectively promotes the dispersed growth of bacterial cells. Its primary function is to remove soluble BOD. The second stage is designed as a predator stage (long SRT) by selectively growing filter feeders (protozoa and rotifers) that prey on the single cell bacteria produced in the first stage and thus converting excess sludge into energy, water and carbon dioxide. The process principle is theoretically sound and is based on the fact that during biomass conversion from a lower trophic level (bacteria) to a higher one (protozoa and metazoa) energy is lost because maintenance and other physiological processes require energy. Decreased energy thus remains available for anabolic processes (biomass production) when higher life forms are present (Ratsak et al., 1994). The loss of energy and biomass production has an inverse relationship. As the loss of energy approaches maximum the biomass production approaches minimum (Ratsak et al., 1994). A schematic of this low sludge process is shown in Fig. 4.

Lab and field studies have verified that sludge production from a two trophic level process is typically less than one third of that from the conventional ASP (Asselin et al., 2004; Stuart et al., 2000). In ciliate proliferation studies, Ratsak et al. (1994) found that the introduction of a second stage led to a 12–43% reduction in sludge yield compared with a single stage system. Lee and Welander (1996a, b) applied this concept to a synthetic wastewater (acetate and MeOH) and different pulp and paper industry wastewaters. The first stage was a laboratory completely stirred reactor for decomposition of dissolved organics by bacteria. The second stage was a biofilm reactor for the growth of predators. Sludge yield reductions ranging from 32% to 93% were obtained depending upon the type of biofilm carrier-material and the type of wastewater used. Microscopic examination of the biofilm confirmed the presence of a variety of species of both protozoa and metazoa, ranging from filter feeders (ciliates, rotifers) to more advanced predators (Lee and Welander, 1996a, b). Lee and Welander (1996b) investigated various designs of the predator stage, i.e., suspended growth and biofilm reactors, for pulp and paper sludge reduction. The sludge production decreased from 0.2 to 0.4 g SS/g COD removed for the conventional ASP to between 0.01 and 0.23 g SS/g COD removed for the two-stage process.

The two trophic level approach can be applied to suspended growth (such as activated sludge) and attached growth (such as trickling filter) processes. Trickling filters carry a more

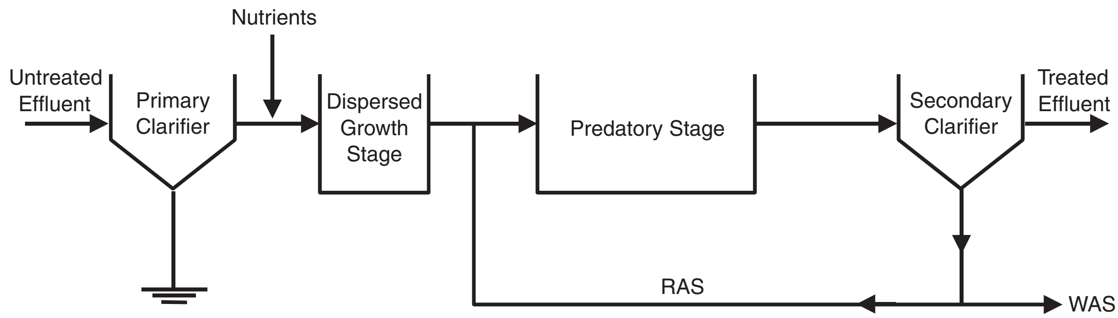

Fig. 4 – Schematic of the two-stage low sludge process.

diverse ecosystem than the ASP. In particular, the metazoa population is larger in trickling filters (Rensink and Rulkens, 1997). This might be a reason that, in general, the sludge production in trickling filters is much less than that in the activated sludge systems. Also, one can infer that the presence of a carrier in activated sludge systems could improve growth conditions for higher metazoa. Rensink and Rulkens (1997) compared sludge production rates of two activated sludge pilot plants treating settled domestic wastewater. Both units were equipped with plastic carrier material to a volume of 30% of the aeration tank. One pilot plant was used as a control while the other was inoculated with oligochaete worms such as *Tubificidae*. Experimental results showed that the excess sludge production decreased from 0.4 to 0.15 MLSS/g COD stabilized when *Tubificidae* were present. The sludge volume index (SVI) decreased from 90 to 45 mL/g when the worms were present. The lower sludge production rates were accompanied by an increase in nitrate and phosphate concentrations, which are often regulated (Rensink and Rulkens, 1997).

Mill implementation of this process, at the Norske Skog Folla CTMP Mill in Norway, was the site for the first installation of this process (Welander et al., 2000). The mill treatment plant receives 5000–6000 m^3 of effluent per day that flows into a 1300 m^3 selector followed by a 15,700 m^3 aeration basin and a secondary clarifier. In the original design, the RAS was recycled to the selector. The conventional design was converted to a low sludge process by redirecting the RAS to the main aeration basin thereby leaving the selector without return sludge. This 3-month study showed no change in the COD removal efficiency while the sludge yield dramatically decreased from above 0.15–0.05 kg TSS/kg COD. A follow-up study after fine tuning the operational parameters further reduced sludge production to 0.03 kg TSS/kg COD, which is comparable to sludge production from anaerobic processes. Since the LSP more completely oxidizes organics, the oxygen demand in this study increased 30% over that of the conventional ASP (Sointio, 2000).

A modification of the LSP process has been suggested relatively recently, involving the addition of suspended inert media in the first aeration stage. The technology is known as the biofilm activated sludge (BAS) process (Welander et al., 2002). In this process, the addition of supplemental nutrients is kept very low in order to encourage the proliferation of polysaccharide producing bacteria which attach to the media in the first stage. The subsequent predatory stage consumes the sloughed off bacteria and polysaccharides which are carried from the first stage. This second stage operates in a similar manner as the predatory stage of the LSP system. The added benefits of the BAS process, as compared to the LSP process, are described to be minimal use of supplemental nutrients, lower nutrient discharge and improved sludge properties (Malmqvist et al., 2004a, b). A major drawback of implementing the patented BAS process is the capital cost of the media.

2.1.1.5. *Anoxic/oxic zone treatment*. About a 25% reduction in sludge production and thus less sludge management expenditure has been reported as a result of a denitrification stage added to the aerobic treatment (McClintock et al., 1988). In the denitrification stage nitrate served as terminal electron acceptor as opposed to dissolved oxygen. Lower sludge production in the presence of an anoxic/denitrifying stage(s) within an aerobic system is consistent with electron transport chain considerations (McClintock et al., 1988). The true yield (yield in the absence of maintenance energy requirements) of heterotrophic microorganisms using nitrate as a terminal electron acceptor is expected to be less than 78% of that realized under aerobic conditions (Payne, 1981).

2.1.1.6. *Additives*. The degradation of the biological matter found in the ASP can be increased with the addition of additives or biostimulants. Many of these additives are patented products with their compositions confidential. These products are generally believed to be metabolism promoting enzymes and/or stimulatory nutrients (Miller et al., 1996). Folic acid (vitamin B) has been reported to act in a similar manner to reduce sludge yields (Akerboom et al., 1994). At a U.S. boardmill, the addition of folic acid was reported to have reduced landfill costs by as much as $10 U.S. tonne of sludge (Akerboom et al., 1994).

Bioaugmentation involves the continual addition of engineered biomass into a treatment system. The conditions within most treatment systems do not facilitate the natural growth of these organisms, thus the treatment system must be continually seeded (Foster and Kramer, 1997). Some of these high degrading rate strains of biomass also have low anabolic rates that result in lower sludge yields.

Uncoupling the oxidative phosphorylation (ADP–ATP) cycle with the additive para-nitrophenol has been shown to reduce sludge production by 50% (Low et al., 2000). However, effluent quality was compromised as a result of its adverse affect on settleability. Ye et al. (2003) identified 3, 3′, 4′, 5-tetrachlorosalicylanilide (TCS) as the best of the five uncouplers tested due to its low effective dose and its being relatively nontoxic. Reductions in sludge yields from 0.5 to 0.3 g sludge/g BOD removed were achieved.

2.1.2. *Sludge reduction through RAS treatment*

The exposure of RAS to conditioning agents has been found to reduce the net production of biological sludge in conventional activated sludge systems. The premise is to expose the recycled fraction of the settled secondary sludge to conditions that render it to a partially non-viable sludge fraction (Fig. 5). This reduces the need to remove excess sludge from the system for maintaining constant MLSS in the aeration basin. A brief description of various RAS conditioning programs that have been reported to date is noted below.

2.1.2.1. *Ultraviolet (UV) irradiation*. The exposure of RAS to UV light in a contact chamber to reduce the generation of secondary sludge has been studied. Smith (1971) subjected a portion of the RAS to UV light in a microbial biolysis unit. The sludge was aerated prior to irradiation. Excess biosolids from an activated sludge unit equipped with a biolysis unit treating municipal sewage was reduced by 55%. In other studies (Elliott et al., 1996, 1999), UV irradiation of RAS in an activated sludge system treating pulp and paper effluents was shown to reduce sludge yields by 15–18%. The exposure time of the sludge was approximately 30 s. The RAS passed through quartz tubing in an UV chamber. This method of conditioning was found to render a fraction of the RAS non-viable and releasing the bacterial cell content during subsequent aeration stages. Additional benefits of conditioning included a release of nutrients (N and P) for recycle and a reduction in abundance of filamentous bacteria. Favourable capital and operating cost estimates were predicted.

2.1.2.2. *Anaerobic/anoxic conditioning*. The flow of the RAS through a holding chamber has been shown to reduce excess sludge production (Elliott et al., 1996, 1999; Saby et al., 2003). Westgarth (1966) developed a patented process that used an anaerobic chamber (residence time between 1 and 7 h) to condition the RAS prior to its return to the aeration basin. The sludge was kept in suspension by mechanical agitation. Because of anaerobic exposure, the sludge wastage rate was reduced between 25% and 50% when treating municipal sewage. Benefits were also claimed in regards to improved settling properties of the sludge. Similar results using anoxic conditions with pulp and paper wastewaters have been documented (Elliott et al., 1996, 1999). In this study, the retention time in the anoxic chamber was much shorter (1–2 h) inhibiting the anaerobic activity. Sludge reductions of 30–35% were reported. Sludge age was found to influence reduction potential. Short aged sludges exhibited greatest reductions in production rates as a result of anoxic conditioning. UV pretreatment of the RAS prior to entering an anoxic conditioning cell was shown to enhance reduction potential (Elliott et al., 1999).

2.1.2.3. *Heat and vacuum*. Smith (1973) patented a process which essentially was the same as his UV irradiation process, described earlier (Smith, 1971). The difference between the two is that heat and or vacuum were supplied in the microbial biolysis unit instead of UV irradiation. Treatment temperatures between 50 and 100 °C were applied for periods ranging from 0.5 and 5 min.

2.1.2.4. *Acidic bleach effluent*. Lee et al. (1976) demonstrated that the WAS from an activated sludge system could completely be hydrolyzed, using acidic conditions in conjunction with high temperatures, and returned to the treatment system eliminating the need for sludge removal. The exposure of RAS to bleach plant acidic effluent was demonstrated to reduce the quantity of excess sludge needed for disposal from an ASP (Elliott and Dorica, 1999). Combining first-stage acid effluent from a Kraft bleach plant with the RAS effectively reduced the quantity of secondary sludge generated in an ASP for removal by as much as 48% (Elliott and Dorica, 1999; Dorica et al., 2000). In this approach, instead of the acidic effluent combining with other mill streams prior to its entering the front end of the treatment system, it enters via the sludge return line providing conditioning of the RAS. The contact time is governed by the residence time the effluent/RAS mixture has in the sludge recycling line although a contact chamber could be installed to increase

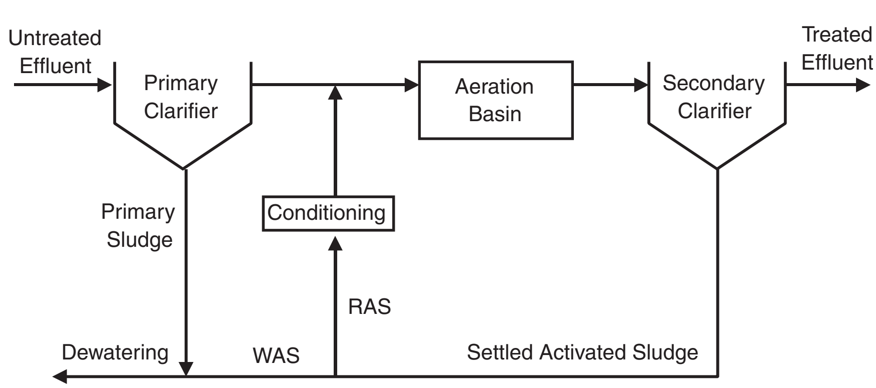

Fig. 5 – Schematic of recycled sludge conditioning.

the exposure time. In addition to sludge yield reduction by as much as 48%, chemical requirements for neutralizing the acidic bleached Kraft mill effluent were lowered by 41%. The destruction of a portion of the sludge liberated nutrients for reuse resulting in 66% less supplemental nitrogen that needed to be added to the system. Economically, this method is considered to be cost-effective for reducing sludge yield because it utilizes an existing wastewater stream, which minimizes operating cost.

2.1.2.5. *Ozonation*. Yasui and Shibata (1994) claimed to be the first to show that ozonation enhances biological degradation of activated sludge. In their experiments, an MLSS concentration of 4200 mg/L was maintained in the aeration basin at 1 kg/m^3 day loading without drawing excess sludge for six weeks using an ozone dose of 0.05 g O_3/g SS. TOC concentrations in the effluent were slightly higher (when O_3 was used) than those from the conventional ASP operated in this study. Yasui et al. (1996) demonstrated that a full-scale ASP could be operated without excess sludge production. In a long-term experiment, that lasted 10 months, no excess sludge was needed to be withdrawn and no significant accumulation of inorganic solids occurred in the aeration tank at a BOD loading of 550 kg/day. The aeration tank was 1900 m^3 in volume.

As reported by Murakami (1998), ozonation is one of the three techniques, which are being considered by the Japan Sewage Works Agency for sludge reduction. The principle involves ozonation of return sludge before it enters the aeration basin. Ozonation solubilizes bacterial cells in the activated sludge and the solubilized cell material is then decomposed to CO_2 and H_2O through aerobic biological oxidation. In order to reduce the sludge production by a given amount, three times that amount has to be treated with ozone. Depending on the extent of treatment, the amount of surplus sludge can be reduced to any ratio. Even zero excess sludge production can be achieved. Murakami (1998) showed that 1.7 kg of ozone is consumed in order to reduce 1 kg of sludge. A disadvantage of this approach is that the cell lysis results in more BOD or COD loading to the reaction basin. Approximately 1.2 kg of additional oxygen is needed to process 1 kg SSs. A process called BIOLEADER, which utilizes these principles, has been developed and applied to pulp and paper wastewater treatment systems (Chu et al., 1999). Another such process, called the Biolysis® O process (a trademark of an ozone-based process licensed by Degremont), has been shown to reduce sludge yield by as much as 80% (Rewcastle et al., 2004). The process involves the exposure of ozone to a portion of MLSS withdrawn from the aeration basin. After sufficient contact time, the MLSS is returned to the aeration basin.

2.1.2.6. *Mechanical shearing*. Springer et al. (1996) described a proprietary adaptation of a wet milling technology (KADY mill) for mechanically shredding RAS. Sludge yield values were reduced from 0.75 to 0.33 kg TSS/kg of BOD removed. Increased aeration was needed as the recycled sludge contained solubilized BOD. The cost of increased aeration was suggested to be compensated by savings in sludge management costs. A subsequent full-scale trial at a dual train activated sludge treatment plant in Maine (flow rate of 57,000 m^3/day) found a sludge reduction of 56% for the train equipped with mechanical shearing as compared to the conventionally operated train (Springer and Higgins, 1999). Again, the return on investment was calculated to be 2 yr.

2.2. *Sludge reduction through post treatment*

Methods are available to reduce the quantity of generated sludge requiring ultimate disposal. Once produced, the sludge can be treated using a number of treatment technologies, such as heat application, chemical oxidation and digestion to reduce the amount requiring disposal.

2.2.1. *Heat treatment*

2.2.1.1. *Incineration*. Among heat treatment alternatives for sludge reduction, incineration is the most popular method. It completely evaporates water in sludge and effectively oxidizes organics at high temperatures to CO_2 and H_2O. In addition to being energy intensive and thus costly (Modell et al., 1992), incineration generally suffers from operating problems in incinerators and bark boilers. The problems include sludge handling, bark and sludge mixture consistency variations and the downgraded boiler capacity because of high water content (Nichols, 1992). Particulate and gaseous emissions requiring air pollution control equipment remain additional issues with sludge incineration. Incineration of sludge in the recovery boiler can, and has been practised in the pulp and paper industry (Harila and Kaila, 1995). The drawback is a reduction in black liquor treatment capacity of the recovery boiler.

Hydro Québec has developed a plasma-assisted incineration technique, which uses a low-power plasma torch to catalyze the oxidation of organics in sludge. Organic content of sludge and its calorific value are important in process economics as heat released from oxidation is used to evaporate water and heat by-products. Sludges containing 20% organic matter have been reported to produce enough heat to keep the reactor at 600 °C. The electrical energy consumption has been estimated to be less than 100 kWh/tonne of treated wet sludge (Chari, 2000).

2.2.1.2. *Carbonization*. Sludge carbonization is another of the three sludge reduction approaches being considered by the Japan Sewage Works Agency (Murakami, 1998). In this process dried or dehydrated sludge is thermally decomposed at 750 °C under oxygen depleted conditions in an externally heated rotary kiln furnace. The resulting product is a mixture of carbon and inorganic components. The inorganic fraction gets chemically tied up with char, and is, therefore, not expected to leach out readily. Around 60–70% of the sludge carbon forms the residue after carbonization while the remaining 30–40% is volatilized to a low-BTU gas (200–300 BTU/ft^3), which can be used as the main heat source of the furnace. Char is 50% carbon (Scott and Smith, 1995) and has a high specific surface area (about 9 m^2/g as granules and about 28 m^2/g as powder), which is roughly one order of magnitude smaller than that of the commercial activated carbon (Murakami, 1998). The product can be used as a soil amendment, horticulture agent, snow melting agent,

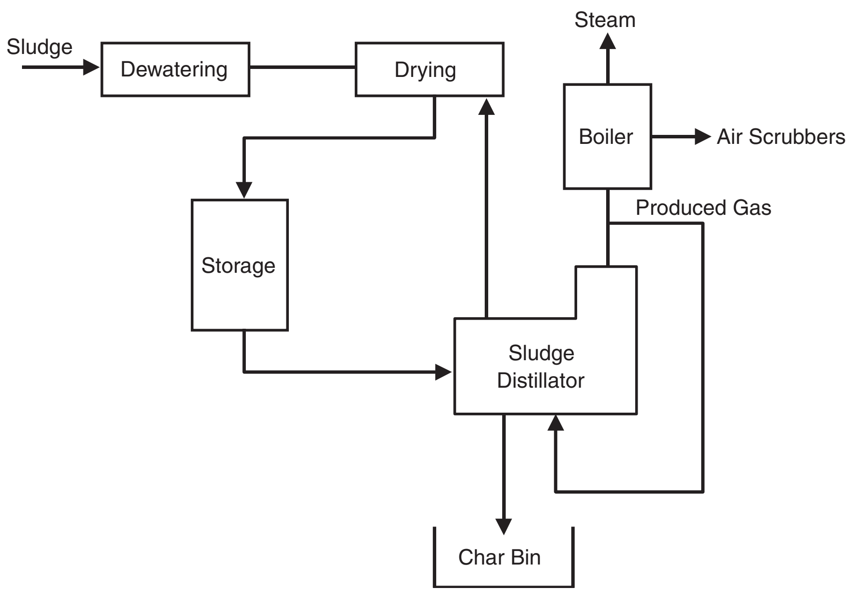

Fig. 6 – Schematic of the sludge carbonization process (modified from Nichols, 1992).

deodorant, decolouring agent and so forth. A sludge dryer is an integral component of the carbonization unit. It is essential that the feedstock is dried to more than 90% solids before carbonization (Nichols, 1992). A process schematic is shown in Fig. 6.

This process is capable of achieving a volume reduction of 90% (Fio Rito, 1993; Nichols, 1992) and a mass reduction of 75% (Nichols, 1992). Although sludge carbonization produces less air emissions of heavy metals and other pollutants compared to incineration, the process has to be equipped with an air scrubber. The system has been claimed to have lower capital and operating costs than those of incinerators (Nichols, 1992). The economic viability of this process depends on the heat value of sludge and the initial dewatering and marketing the residue.

2.2.1.3. *Vitrification*. Sludge vitrification is a two-stage process carried out at high temperatures in the presence of oxygen. In the first stage, the dewatered biosolids are dried and are sent to the melting/solidification second stage. High temperature operation destroys the organic portion of the biosolids leaving a glass aggregate material as a final product, which can be used to manufacture ceramic floor tiles, roofing shingles, sand blasting grit and some other abrasives. A full-scale plant employing this technology has been reported to be in operation since 1998 in Neenah, Wisconsin, processing 1200 wet tonne/day of sludge from 11 different paper companies (Canning, 1999).

This approach can be viewed as a post incineration solidification technology. Ash, which is incinerated at around 800 °C is further heated to melt between 1400 and 1500 °C in a specially designed melting furnace. The molten sludge is either air-cooled or water-cooled depending upon the end use of the final product. From 6 yr of experimental data from a 40 tonne/day dewatered secondary sludge plant in Japan showed that a sludge cake at 35% solids could be reduced to 1/7th by mass and to 1/14th by volume through the use of this technology (Okuno et al., 1997). Leaching tests on slag showed no detectable metals in the leachate, which allows the use of slag in the construction industry.

Despite the very attractive sludge reduction potential of this approach and reusability of the end product, the process economics are not that attractive. Okuno et al. (1997) estimated that incineration and melting processes individually cost \$118 and \$125/ton of dewatered cake at 35% solids, respectively. A combined cost of \$243 for incineration plus melting is much higher than the current disposal costs in the North American sludge disposal practices. The slag is saleable at \$4/ton only (Okuno et al., 1997). The chances of rapid commercialization of sludge melting technology in the North American pulp and paper industry therefore seem to be limited. However, solidification is attractive if immobilization of heavy metals is required. A full-scale vitrification plant is in operation in Japan since 1990, which converts 160 tonne/day of sludge cake to 13 tonne/day of slag (Spinosa et al., 1994).

2.2.1.4. *Gasification*. In this process sludge is reacted with steam within a temperature range of 650–720 °C, which converts sludge into a low to medium heat value gas. The inorganics are complexed with residual carbon. Durai-Swamy et al. (1990) used a pulse-enhanced, indirectly heated fluidized bed gasifier system to treat wood chips, recycle paper mill sludge and Kraft mill sludge. Carbon conversions to dry gas were found to be 93%, 87% and 80% for wood chips, recycled mill fibre waste and Kraft mill sludge, respectively. The respective char productions from carbon were 4%, 8.5% and 19%. A small fraction of sludge organics was converted to oil. Wood chips, recycled mill fibre waste and Kraft mill sludge were processed wet as received at moisture contents of 25%, 50% and 62%, and ash content of 0.2%, 3% and 19%, respectively.

Dioxin emission from sludge incineration has been a concern. The technology developed by Durai-Swamy et al. (1990) performs gasification in a bed of calcium-based material, which is reactive towards chlorine containing dioxin precursors. The technology has thus been claimed to suppress the dioxin formation potential (Durai-Swamy et al., 1990).

Krause and Levert (2004) are marketing the deep bed bark/sludge combustion process which incorporates gasification to maximize power generation from the combustion of wet fuels from pulp and paper mills.

2.2.1.5. *Low-temperature pyrolysis*. The main thrust for the development of low temperature sludge conversion to oil and solid fuel came from Environment Canada (Spinosa et al., 1994). This technology is similar to sludge carbonization with the exception that lower reaction temperatures (300–350 °C) are needed to promote volatilization. The volatiles are then condensed to oil in a subsequent stage as shown in Fig. 7.

It has been postulated that low-temperature pyrolysis vapour phase reactions convert organics to straight chain hydrocarbons, which resemble those in crude oil. Chemical analysis (Spinosa et al., 1994) has shown that the aliphatic hydrocarbons produced in this process differ from aromatic and cyclic compounds generated in other pyrolysis processes, regardless of the nature of the substrate. Oil yields ranging from as low as 13% for anaerobically digested domestic sludge to as high as 46% for mixed (primary plus secondary) domestic sludge have been reported (Campbell, 1989). Char yield in this study was found to vary from 40% to 73% at the optimal operating temperature. The additional benefit of recovering nutrients from sludges with pyrolysis has been documented by Bridle and Pritchard (2004).

Environment Canada completed a thorough evaluation of sludge incineration costs and compared them with sludge liquefaction costs. Taking capital costs from actual construction contracts of four sludge handling facilities (comprising sludge digestion, conditioning, thickening, dewatering, incineration and ash disposal) and updating them by applying the engineering news record construction cost index, the sludge incineration costs were calculated. The total (capital and operating) cost of sludge incineration for the four facilities was found to vary from \$350 to \$1042 per dry tonne while the cost of low-temperature pyrolysis was found to be \$207 per dry tonne. When the economic value of the oil was factored in, the cost of treatment declined to \$148 $tonne^{-1}$ (Campbell, 1989). The calculations assumed an oil yield of 30% on a total solids basis. Although the economic viability of the approach depends upon oil yield, it can potentially reduce sludge volume to 40% (char) of the original volume.

2.2.2. *Chemical oxidation*

2.2.2.1. *Supercritical water oxidation (SCWO)*. Water above 374 °C and 22 MPa exists as a supercritical fluid, which is an intermediate state between liquid and gas. In its supercritical state water becomes a superb solvent for organic molecules and the oxidation reactions never become diffusion limited (Murakami, 1998). In addition to being an excellent solvent, supercritical water also reacts with organic molecules transforming them to small molecules without forming char

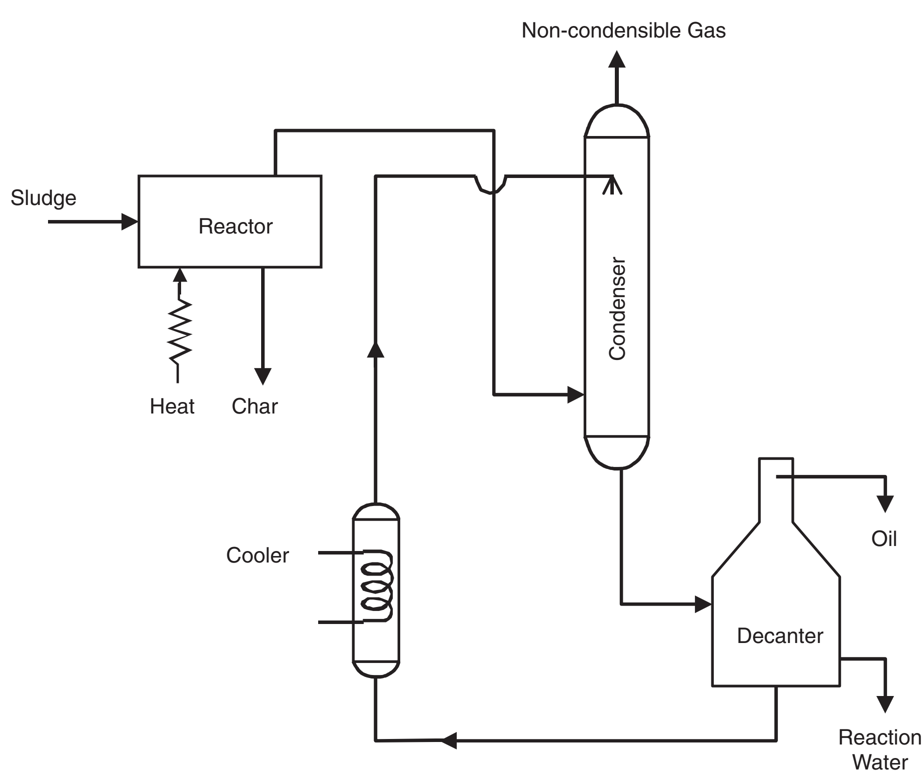

Fig. 7 – Schematic of a sludge to oil converter (modified from Campbell, 1989).

(Modell, 1985). In the presence of oxidants, such as air and oxygen, the small molecules can be readily oxidized.

SCWO of sludge is a process, in which, water briefly reacts with sludge under supercritical conditions forming water, CO_2, and nitrogen gas from organic compounds, and acids, salts and oxides from inorganic materials (Modell et al., 1992). The overall chemical transformations in SCWO are similar to those in incineration but the process of oxidation is different. A schematic of a SCWO system is shown in Fig. 8. In this process the sludge is pressurized to 25.5 MPa (250 atm) and mixed with pressurized oxygen at room temperature in a 90:6 (sludge:oxygen) ratio. The sludge/oxygen mixture is then heated to a temperature ranging from 300 to 400 °C, depending upon the nature of the waste material. The rate of oxidation increases as the pre-heater temperature increases. Somewhere between 360 and 380 °C the liquid phase in the pre-heater disappears. The reaction mixture enters the main reactor where the remaining portion of the organics is oxidized in short hydraulic residence time of 5–10 min at the maximum process temperature of around 600 °C. Once the oxidation approaches completion the reaction mixture is cooled to room temperature in a heat exchanger and the water vapour is converted back to liquid. The gas phase, mainly comprising CO_2 and excess O_2, is then separated from the solid/liquid slurry. In a subsequent step, CO_2 is separated by liquefaction from excess O_2, which is recycled back to the process. CO_2 can be purified, if needed, and sold for economic value. It is interesting to note that despite a highly oxidizing environment in the pre-heater and the main reactor, the feed nitrogen is not oxidized to NO_x. Below 500 °C ammonia is the predominant nitrogen species (Helling and Jefferson, 1988) and above 600 °C molecular nitrogen and nitrous oxide (N_2O) are abundant species (Modell et al., 1992).

This process can very effectively oxidize organics at moderate temperatures (400–650 °C) and high pressure (25.5 MPa). Lab-scale studies have shown TOC and AOX reductions of 99–99.9% and dioxin reductions of 95–99.9% (Modell et al., 1992). Murakami (1998) obtained optimum SCWO results at 650 °C. At this temperature complete removal of organic carbon was achieved in 10 s and the exhaust gas remained free of NO_x and SO_x.

One advantage of SCWO is that it does not require a high degree of dewatering prior to oxidation. Sludge can be processed at 10% solids by weight or even less. Modell et al. (1992) have claimed that SCWO is cheaper than dewatering plus incineration for treating pulp mill sludges because of regenerative heat exchange to preheat feed and cool effluents. Also, SCWO was predicted to compete effectively with dewatering plus landfilling costs where tipping fees exceed U.S. \$45 m^{-3} (Modell et al., 1992). Cooper et al. (1997) treated primary clarifier sludge (at 30% solids) mixed with bleach plant effluent (30% Do:70% EoP) at 7% solids and decant pond sludge by SCWO. The sludges were wet ground before oxidation in order to reduce particle size below 0.15 mm. Results showed that the two sludges were oxidized at 81.4% and 97.7% organic carbon destruction efficiencies and 99.47%

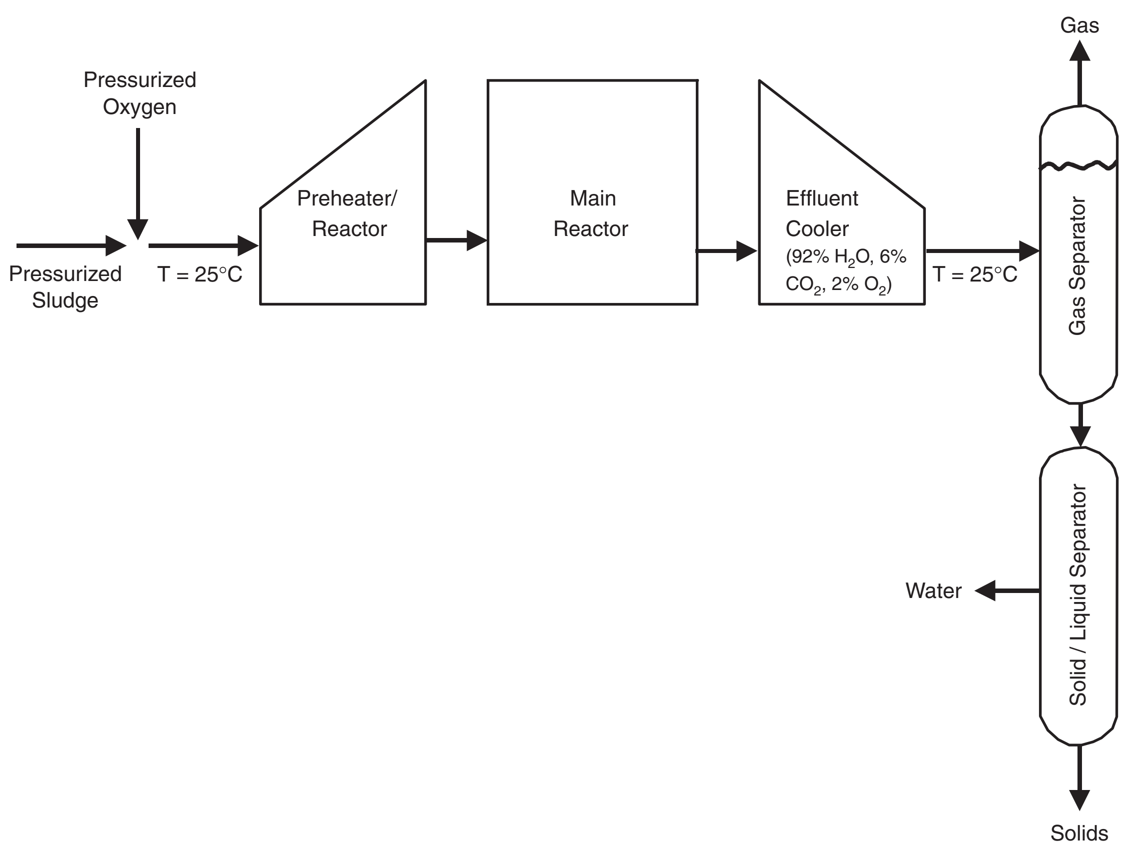

Fig. 8 – Schematic of a SCWO system (modified from Modell et al., 1992).

and 99.93% AOX removal efficiencies, respectively. The liquid effluents from the two sludges were colourless and neutral in pH but contained 460 and 420 mg/L dissolved organic carbon (DOC), respectively.

Dahlin (2002) and Gidner and Stenmark (2002) documented how SCWO can be effective at treating deinking sludge from a pulp and paper facility. The process not only oxidizes organics but it also allows the recovery of a paper filler. The trade name of the process is Aqua Critox® (Minett and Fenwick, 2001).

2.2.2.2. Wet air oxidation. Wet air oxidation (subcritical water oxidation) is a flameless oxidation method for the oxidation of mainly organic substances with air or other oxidizing agents at pressures of 2–20 MPa and temperatures of 150–370 °C (Zimmerman and Diddams, 1960). The process is similar in principle to SCWO with the exception that the reaction mixture is kept below the critical point of water. While SCWO can achieve complete oxidation of the organic fraction of the sludge solids, wet air oxidation can achieve effective hydrolysis (>95% as COD) of the sludge organic compounds but incomplete oxidation (<95% as COD) (Shanableh, 2000). Process efficiency depends upon operating parameters such as temperature, pressure, air supply and feed solids concentration. The extent and the rate of oxidation can be increased by elevating the reaction temperature within the range 120–370 °C. Operating pressures between 1 and 27 MPa can be used depending upon the degree of oxidation desired. As with incineration, an external oxygen supply is needed. Since thermal efficiency and process economics are strongly dependant on air input it is thus critically important that the optimum air requirement is determined. As an example, activated sludge with a heat value of 15212 kJ/kg (6540 BTU/lb) typically requires 5.14 kg air/kg of sludge for wet air oxidation (US EPA, 1979). Although wet oxidation does not require pre-dewatering (as low as 1% solids sludge can be fed to the process), the process operating costs can be significantly reduced by increasing sludge consistency. In one study wet air oxidation costs decreased from \$38 to \$23 tonne^{-1} by increasing sludge solids from 3% to 6%. High sludge solids keep the oxidation process self-sustaining.

The wet air oxidation process has been commercialized as the Zimpro Process. Among major installations of the Zimpro process were the Chicago Sanitary District's installation at its West Southwest Wastewater Treatment Plant (US EPA, 1979) and the Springfield Water and Sewer Commission's regional activated sludge wastewater treatment facility located on Bondi Island (Borgatti et al., 2000). The Chicago installation was operated for 10 yr, after which, the Zimpro process was withdrawn in favour of a landfill disposal system. The Springfield Water and Sewer Commission's facility achieved a 50% destruction of volatile sludge solids using the Zimpro process since its installation in 1970. In 1995 renovations of the treatment plant, the odorous and maintenance and energy intensive Zimpro process was replaced with an extended aeration operation of the upgraded ASP, which, since then, has lessened secondary sludge production by 30% (Borgatti et al., 2000).

Wet air oxidation offers the advantages of air pollution-free operation, far lower ash production than that by incineration, no need of sludge dewatering and thus savings in sludge conditioning costs (Collyer et al., 1997). A major drawback of the wet air oxidation is the fact that it produces high-strength liquors, which need to be recycled to the treatment plant (Shanableh, 2000) requiring increased aeration capacity. BOD of the liquor can be as high as 40–50% of that of the raw sludge, which translates to a 30–50% increase in BOD loading to the treatment system.

Both surface and subsurface reactor designs have been proposed to achieve super and subcritical water oxidation. Surface reactors are high-pressure vessels capable of sustaining high pressures applied by mechanical devices such as pumps. A subsurface reactor is constructed under ground; deep enough that the hydrostatic head achieves the required pressure. If the entire pressure required for the SCWO is to be developed by the hydrostatic head, a reactor depth of approximately 3650 m would be needed (US EPA, 1992). Based on similar principles, a wet oxidation technology (deep-hole technology) has been developed in Netherlands for sludge oxidation, which consists of a set of vertical pipes, drilled vertically about 1500 m from the surface of the Earth. Pure oxygen is supplied to start-up the process and afterwards, heat is generated from the exothermic sludge oxidation. The liquid is biologically treated while the ash is dewatered for disposal. The technology is inappropriate for use in some geological conditions (De Bekker and Van den Berg, 1988) and still has a long way to go to demonstrate its economic viability (Spinosa et al., 1994).

2.2.2.3. Alkali digestion. Using the total oxidation approach of Gaudy and co-workers (Gaudy et al., 1970, 1971; Yang and Gaudy, 1970), Lee et al. (1976) showed that pulp mill WAS for disposal can be reduced by 70–80% by alkali digestion of the excess sludge and returning the solubilized fraction back to the aeration basin. Investigations on the effect of temperature, alkali concentration and contact period showed that even under optimal sludge hydrolysis conditions, complete solubilization of sludge did not occur and 20–30% sludge remained in suspended form. The rate of solubilization was highest during the first half hour and no significant additional digestion was observed after 2 h of contact period. The extent of solubilization increased with increasing concentration of NaOH up to 0.1 N. Concentrations beyond 0.1 N did not enhance solubilization. Temperature showed a linear correlation with the extent of solubilization. Maximal sludge hydrolysis was achieved at 70 °C. Each milligram of solubilized sludge produced 0.4–0.5 mg BOD_5, 0.06–0.08 mg TKN and 0.01 mg P. The excess nutrients released in the solubilization process would result in savings in the costs due to nutrients. The BOD released in sludge solubilization is an inherent disadvantage of this approach as it requires additional aeration and it increases organic loading to the treatment plant. This can be a major concern where the treatment systems are already near or above capacity.

2.2.3. *Sludge digestion*

Aerobic or anaerobic sludge digestion is commonly practised in the municipal sector (Metcalf and Eddy Inc., 1991). These processes convert raw sludge into a less offensive form with regard to odour, rate of putrefaction and microorganism content in addition to mass reductions of 50–70% (Lee et al., 1976). Anaerobic and aerobic digestion partially converts putrescible matter into liquid, dissolved solid and gaseous by-products with a significant destruction of pathogens. Anaerobic digestion has been described as a potential method of reducing the quantity of pulp-mill WAS (Puhakka et al., 1992a, b), but neither of the digestion methods are commonly used in the pulp and paper industry. The large amounts of generated sludge and the long retention times needed for its digestion are main reasons to limit its implementation. Recent advances in reducing the retention times are making these technologies more attractive to the pulp and paper industry. Excessive foaming is another problem in both aerobic and anaerobic digestion of sludge. Control methods are available but not always effective since the causes of foaming are not fully understood (Spinosa et al., 1994).

2.2.3.1. Aerobic digestion. Aerobic digestion has been found to reduce the volume of WAS by as much as 50%. This method of sludge reduction involves aerating and mixing the sludge in an open holding tank or lagoon for a period of time. The sludge is digested to carbon dioxide and water. The digestion time required is dependant on a number of factors such as temperature and pH (Krishnamoorthy and Loehr, 1989) and generally ranges between 10 and 25 days. There are a multitude of types of aerobic digestion processes. The inclusion of an anoxic stage to encourage endogenous nitrate respiration (Hao et al., 1991) produced similar sludge reduction rates but at a lower operating cost.

Aerobic digestion of waste sludge can occur under mesophilic or thermophilic temperatures. Thermophilic digestion results in a greater reduction in solids but the indigestible solids are more difficult to dewater. Other operational problems of thermophilic digestion include nuisance odour and foaming. It can be a practical method for sludge stabilization and disinfection. This process has found several applications, especially in Europe. Its use as a preliminary stage before anaerobic digestion was also demonstrated in Switzerland (Spinosa et al., 1994).

A two-stage digester with the first stage operating with mesophilic temperatures and the second with thermophilic maintained a high reduction in solids with improved dewatering properties of the remaining sludge (Murthy et al., 2000). This process is known as the autothermal thermophilic aerobic digestion (ATAD) process. The ATAD process has been in use in Europe for the past 2 decades. This process can be an economical candidate for small to medium flows (between 0.3 and 15 MGD). In 1990, the U.S. EPA issued a document (US EPA, 1990) describing the process concept and design criteria. At present, there are three full-scale applications in the U.S. The ATAD process utilized two or more digesters in series operated under slightly aerobic conditions and at short hydraulic retention times between 6 and 8 days. For the thermophilic process operation, high temperatures between 55 and 65 °C are achieved and maintained through the heat released by the destruction of degradable organics in the biosolids. Efficient mixing, aeration and a minimum feed solids content (typically 4–5%) are essential to achieve reliable thermophilic temperatures (Murthy et al., 2000). Being thermophilic in nature, the ATAD process is attractive in achieving Class A vector and pathogen reduction requirements of the U.S. EPA. A hydrolysis stage prior to the ATAD process has been found to reduce further the undigested sludge fraction (Rozich, 1990).

2.2.3.2. Anaerobic digestion. Anaerobic digestion conventionally uses a two-stage sealed reactor system. The first stage digests the solids, and the second stage separates the undigested solids from the liquid. Digestion transforms the sludge to carbon dioxide, methane and water. Rintala and Puhakka (1994) reported in a review paper that "both primary and secondary sludges from pulp and paper industry wastewater treatment plants are amenable to anaerobic digestion". Puhakka et al. (1992b) reported that anaerobic pilot-scale digestion of Kraft WAS resulted in deteriorated dewaterability. The dewaterability of WAS and digested sludge (DS) improved when mixed with primary sludge. The filtrate of DS was found to be a potential source of nutrients for the treatment of phosphorus- and nitrogen-deficient Kraft mill wastewaters.

Puhakka et al. (1992a) operated a pilot-scale anaerobic digester for 21 months to determine optimum operational conditions for solids reduction and biogas production. Sludge volume reductions of 40% with sludge containing about 38% lignin produced 0.5 m^3 biogas/kg sludge removed. Optimal process performance was obtained at the sludge loading of 2.2 kg/m^3 day. Alkalinity was needed (13 g NaOH/kg sludge) to maintain the optimum pH for maximum sludge yield reduction.

A Kraft mill in Ontario (Espanola) developed and used an in situ anaerobic fermentation method to digest the primary sludge as opposed to conventional landfilling. The anaerobic process was found to be significantly more economical than the traditional settling basin dredging and land disposal method (Fein et al., 1989). Primary sludge inoculated with municipal digester sludge prior to digestion showed no advantage over the unseeded control. However, fermentation with acclimatized seed microorganisms exhibited higher fermentation rates and had the shortest lag times. An in situ pilot test showed that 87% of the initial sludge (volume basis) got solubilized in 81 days. The initial sludge was fibrous and was at high consistency while the sludge at the end of the fermentation had a loose watery consistency. Throughout the field study no significant increases in the ambient TRS were observed. The in situ method was demonstrated on full scale; the associated costs of this full-scale sludge disposal method were significantly less than those for dredging and landfilling historically employed at the mill.

Saiki et al. (1999) investigated anaerobic sludge hydrolysis using an upflow anaerobic sludge blanket (UASB). The system combines sludge solubilization with existing treatment (Fig. 9). The anaerobic exposure solubilized 40% of the organic fraction of a brewery wastewater treatment plant sludge in four days of incubation at 60 °C. NaOH at a final concentration of 0.01 N was added to the sludge before incubation. The gas bubbles formed in this process when attached to sludge flocs reduced the floc specific gravity to 0.6–0.7, which caused thickening of the residual solids by flotation.

Enhancement of digestion can be achieved by the addition of enzymes. Lagerkvist and Chen (1993) found that anaerobic digestion was improved by 45% with the addition of Econase.

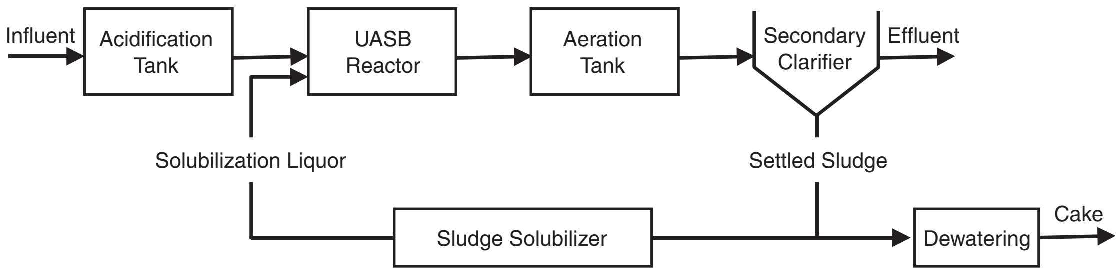

Fig. 9 – Schematic of a wastewater treatment system with solubilization producing less sludge (modified from Saiki et al., 1999).

The anaerobic treatment of combined primary and secondary sludge has been shown to produce valuable products. Domke et al. (2004) demonstrated that pulp and paper mill biosolids can be fermented to form alcohol fuels. The use of anaerobic digestion has also been reported to pre-treat wastewater, thereby off loading organics entering an aerobic treatment system (Risse and Datschewski 2004; Stahl et al., 2004). The American–Israeli paper mill in Hadera, Israel, has recently performed a pilot trial using an anaerobic pre-treatment reactor to digest untreated mill effluent (Stahl et al., 2004). The anaerobic digestion of the wastewater resulted in a much lower amount of organics entering the activated sludge system, thus substantially reducing the quantity of WAS that was required to be wasted from the aerobic treatment system.

2.2.3.3. Solubilization pre-treatment enhancing digestion. - Sludge solubilization and cryptic growth is viewed as a series of three steps—death, hydrolysis and growth. Decay coefficient measurement studies of Lishman and Murphy (1994) have shown that hydrolysis of the organisms is the reaction rate-controlling step in the overall sludge digestion process. In sludge digestion, solubilization occurs due to enzymatic hydrolysis (Mukherjee and Levine, 1992). It has been shown that mechanical, thermal, chemical or biological means or a combination of them can provide efficient solubilization of the biosolids (Tiehm et al., 1997; Tanaka et al., 1997; Mukherjee and Levine, 1992; Saiki et al., 1999). Through solubilization, the insoluble sludge biomass is disrupted to soluble molecules of carbohydrates, amino acids, nucleotides, fatty acids and small amounts of some other materials, which when recycled to the aeration basin are metabolized to CO_2, water and new biomass. Repetition of this procedure ultimately maximizes sludge conversion to CO_2 and water, thereby minimizing the amount of excess sludge produced (Lee et al., 1976).

Various physical and chemical sludge solubilization techniques with varied degree of impact on sludge digestibility have been reported in literature. Among solubilization strategies investigated are thermal pre-treatment (Stuckey and McCarty, 1984; Li and Noike, 1992), thermo-chemical pre-treatment (Tanaka et al., 1997), addition of enzymes (Howell et al., 1978), ozonation (Yasui and Shibata, 1994), solubilization by acidification (Gaudy et al., 1971; Woodard and Wukasch, 1994), alkaline hydrolysis (Mukherjee and Levine, 1992), acid/alkaline treatment (Reincke, 1989), mechanical sludge disintegration (Nah et al., 2000; Baier and Schmidheiny, 1997), and ultrasonic disintegration and biological solubilization of sludge (Saiki et al., 1999; Forster et al., 2000; Schneider, 2003).

Bench scale experiments of Choi et al. (1997) and pilot scale experiments of Nah et al. (2000) showed that WAS can be solubilized effectively by jetting and colliding it to a collision-plate at 30 bar pressure (Fig. 10). Experimental results showed that sludge solubilization by this method positively impacted digester performance as indicated by gas production, volatile fatty acids, pH and volatile mass reduction efficiency. The pre-treatment resulted in 13–50% volatile solids reduction in anaerobic digestion at 2–26 days retention time as compared to 2–35% removal without pre-treatment but under same digestion conditions (Choi et al., 1997). The mechanical pre-treatment of WAS allowed a decrease in anaerobic digester SRT from 13 to 6 days without adversely impacting effluent quality and volatile solids destruction efficiency, which remained at 30% (Nah et al., 2000).

Wet milling studies of Baier and Schmidheiny (1997) showed that a ball mill could achieve high degrees of cell disruption. At a power consumption of 1.0–1.25 kW/m^3 of sludge treated per day, the fraction of soluble COD changed from 1% to 5% in original sludges to up to 47% after wet milling. Of eight sludges investigated, the ones with a high percentage of microbial cells (compared to total solids) showed highest potential for COD solubilization, which perhaps limits the use of wet milling (or perhaps sludge solubilization techniques in general) for disintegrating pulp and paper primary sludge. Wet-milled sludge showed good anaerobic digestibility of the solubilized intracellular material. COD-degradation of the sludge was enhanced by a factor of 1.2–1.5. Biogas production was enhanced in the same order of magnitude and volatile solids destruction increased from 40% to 60%.

Turai (1980) found ultrasound to be effective at reducing the generation of biological sludge, with the added benefit of increasing the metabolism of the surviving sludge fraction. Ultrasound mechanically disintegrates cells and flocs by inducing dramatic stress conditions in sludge such as high mechanical shear stress, radical reactions through the generation of H and OH radicals, and thermal breakdown (pyrolysis) of organic substances. Schneider et al. (1998) also used ultrasonic disintegration of sludge to improve its biodegradability. Excess sludge from a domestic wastewater treatment plant in Germany was disintegrated in a

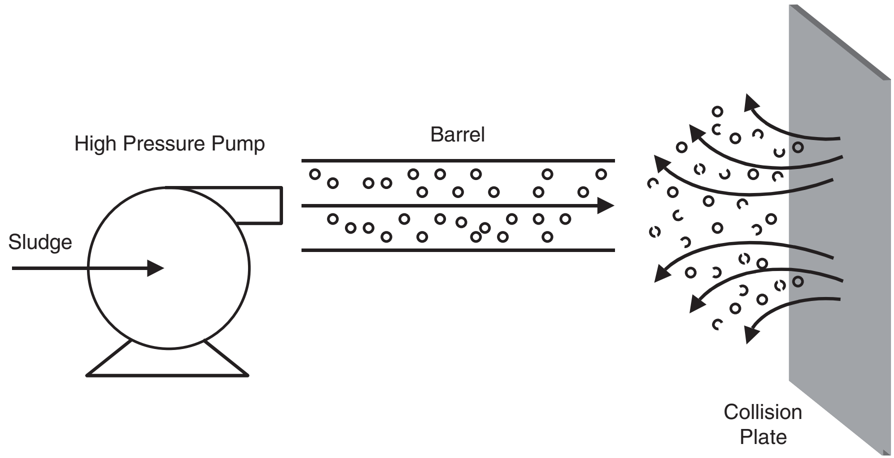

Fig. 10 – Schematic of high pressure impact for sludge pretreatment (modified from Nah et al., 2000).

Table 2 – Effect of ultrasound treatment and digestion time on fermentation of sludge (after Tiehm et al., 1997)

Raw sludge	Residence time (day)	% Reduction of VS
Untreated	22	45.8
Disintegrated	22	50.3
Disintegrated	16	49.3
Disintegrated	12	47.3
Disintegrated	8	44.3

sonoreactor and digested in a fermenter with different residence times. Depending on the duration of ultrasonic treatment and the residence time, the excess sludge was reduced by 40%. Also, ultrasonic treatment reduced the sludge digestion time by a factor of four. Tiehm et al. (1997) made similar observations when they fermented untreated and ultrasonically disintegrated sludge at 31 kHz. A summary of semi-continuous fermentation experimental results with disintegrated and untreated sludge is presented in Table 2. The first two fermenters, operated at similar residence times of 22 days, showed volatile solids reductions of 45.8% and 50.3% for the untreated and ultrasonically treated sludges. Sludge digestion was quite stable even at the shortest residence time of 8 days with biogas production 2.2 times that of the control fermenter. Neis (2000) also reported a decrease in anaerobic sludge digestion time from 22 to 8 days. Sludge sonication for only 96 s increased the supernatant COD of the sludge samples to more than 6000 mg/L. Despite high investment costs and the high amount of energy required, the authors (Schneider et al., 1998; Tiehm et al., 1997) claimed that the use of this technology was cost-effective.

Rocher et al. (1999) investigated four cell-disruption techniques for sludge solubilization, ultrasound, temperature, NaOH and NaOH plus temperature. Of these, thermo-chemical treatment using NaOH was most efficient for inducing cell lysis, which is consistent with the findings of Tanaka et al. (1997) and Mukherjee and Levine (1992). Alkali hydrolysis was ranked superior to acid addition because of alkali compatibility with the biological treatment (Rocher et al., 1999; Mukherjee and Levine, 1992). It was hypothesized that the alkaline treatment of the lignocellulosic materials induces swelling in the particulate organics, rendering them more susceptible to enzymatic and chemical attack thereby significantly enhancing biodegradability (Pavlostathis and Gossett, 1985). Rocher et al. (1999) found that the optimal conditions to induce cell lysis were a pH of 10 at a temperature of 60 °C and an incubation period of 20 min. Carbon release was suggested to be a two-step process. In the first step an immediate release occurs followed by a post treatment release. The post treatment comprised 15 days of sterile incubation under conditions equivalent to the first step. The total DOC released by the WAS varied from 200 to 270 mg DOC/g TSS. The lysate contained only small amounts of refractory COD (10–30 mg DOC/g TSS), which is in agreement with the values reported by Chudoba (1985). A large fraction (80–90%) of the solubilized DOC (immediate or post treatment) was capable of being assimilated. By using fresh WAS, the degradation yields of the immediately released DOC and the DOC released after treatment were found to be independent of the cell disruption technique employed. In similar studies, Tanaka et al. (1997) evaluated the effect of pre-treatment on anaerobic digestion of WAS. Mixed-sludge (domestic, commercial and industrial) showed 40–50% VSS solubilization when the WAS was heated to 130 °C for 5 min with an NaOH dose of 0.3 g/g VSS. Anaerobic digestion of the solubilized sludge resulted in a 200% increase in methane production. Ray et al. (1990) showed that mild alkali solubilization at ambient temperature can significantly improve volatile solids removal and gas production. In addition, anaerobic digestion time was reduced to 7.5 days.

Li and Noike (1992) studied the effect of thermal pretreatment on anaerobic treatment. Heat treatment effectively transformed the WAS to soluble carbohydrates, lipids and proteins or converted them to lower molecular weight volatile fatty acids. The optimal solubilization results were achieved at incubation for 60 min at a temperature of 170 °C. With thermal treatment, the sludge anaerobic degradability was greatly increased and the anaerobic digestion time could be reduced to 5 days. Out of lipids, proteins and carbohydrates in the WAS, carbohydrates were relatively more difficult to be degraded, because they were mainly composed of cell walls of a strong structure. However, soluble carbohydrates produced by thermal treatment were almost completely decomposed during anaerobic digestion.

3. Discussion and evaluation of relevant technologies

It is evident from the above presentation that a choice among the various available reduction technologies is likely to be case specific and would require techno-economic evaluation of the candidate technologies. In order to narrow the focus down to technologies that hold greatest potential for implementation at pulp and paper mills the following criteria have been considered:

- The technology is transferable to the pulp and paper industry and can readily be adapted or retrofit to the existing equipment.
- The selected technology requires as little capital as possible and offers acceptable pay back time.
- Operation and maintenance costs are minimal.
- The treatment efficiency of the existing wastewater treatment plant either improves or remains unaltered upon implementation of the selected reduction technology.
- The reduction process is not detrimental to subsequent dewatering operations.
- The reduction technology preferably does not require air pollution control equipment.
- The chosen process is as simple and robust as possible and high operating skills are not required.
- The technology is safe and durable.

A critical evaluation of all candidate reduction technologies listed in Fig. 1 (based on above criteria) indicates that, in general, sludge reduction through treatment process changes appears more appealing than post treatment alternatives. The former approach offers clear advantage over the later in that the treatment process changes allow for decreased sludge to be produced in the first place thus decreasing sludge management costs.

As listed in Fig. 1, a multitude of operational control strategies can be implemented for lower sludge production. Because of a low food/microorganism ratio, the *extended aeration* mode of operation of the ASP produces 30–50% less sludge compared to the conventional activated sludge operation. Although the low sludge production feature of the extended aeration process is attractive, a 300% increase in aeration basin holding capacity together with approximately a 100% increase in aeration capacity could in many cases limit the conversion of the conventional ASP to an extended mode of aeration. Nonetheless, those conventional activated sludge systems, which operate at significantly lower capacities than the design values as a result of over design or decreased production rate or decreased loading rate due to process changes or system closure, can be good candidates for conversion to the extended aeration mode of operation.

In *MBRs*, the secondary clarifier is eliminated altogether and the sludge/supernatant separation is achieved by using membrane processes directly from the aeration basin. Although an MBR offers advantages such as physically retaining solids, eliminating secondary clarification and its associated problems, providing an absolute control over the solids retention time, and the retention and degradation of higher molecular weight compounds (Tardif and Hall, 1996), the process has not yet been economically justified. Issues such as high capital and operating cost of membranes, membrane clogging, etc. need to be addressed before the process can be commercialized for the pulp and paper industry.

Many activated sludge plant operators maintain dissolved oxygen concentrations on the order of 2 mg/L. However, higher bulk phase oxygen concentrations through *improved aeration* have been shown to be effective in reducing excess sludge production. This concept can be implemented without incurring any significant capital expenditure at mills that already have excess oxygenation capacity available. The increased aeration costs or a part of that can perhaps be offset by savings due to decreased sludge dewatering (improved P/S ratio) and disposal costs.

Although the *low sludge process*, which uses a two-trophic-level approach, sounds very attractive, one should be aware of its pros and cons before implementation at a mill site. Stuart et al. (2000) identified potential negative aspects of the LSP process such as excessive first stage polysaccharide generation, increased aeration costs, and an increase in ammonia nitrogen residual in the effluent, which in sufficient concentration can be acutely toxic to rainbow trout. Excessive nitrates overflowing to the secondary clarifier can give rise to sludge rising problems, while the predator stage phosphorus residuals can exceed phosphorus discharge regulations. A modified LSP system known as the BAS process has been suggested to be able to minimize nutrient discharge. A reasonable knowledge of process microbiology and adaptable system design seems vital to the success of the LSP and BAS approaches. As well, the complexity of conversion of existing biological treatment systems to the LSP and BAS configurations could be problematic. Finally, a major concern of implementing the patented BAS process is the capital cost of the media.

Most aeration basins are designed as completely stirred tank reactors. In order to render *anoxic/oxic zone treatment*, the aeration basin needs to be compartmentalized, which could involve significant capital investment. Also, in nitrogen deficient pulp and paper wastewaters more nitrogen will have to be added to serve as a terminal electron acceptor thereby increasing operational costs. This approach seems more suitable for plug flow reactors designed to treat nitrogen rich wastewaters. Also, a serious odour nuisance might result if the anoxic zone gets anaerobic.

Additives introduced to the aeration basin have been found effective at reducing sludge yield. The major constraint is their high cost and continual/continuous addition, particularly for the large treatment facilities employed in the pulp and paper industry. Environmental implications of some additives, such as metabolic decouplers, in general, remain additional constraints towards their use.

RAS treatment with various agents before its recycle to the aeration basin of a conventional ASP is attractive from a capital investment point of view. Optimal selection of the RAS conditioning agent can keep operational costs low. Also, the solubilization of a portion of sludge releases nutrients for reuse, resulting in a significant decrease in supplemental nutrients that are otherwise needed to be added to the system. Significantly lower excess sludge production coupled with low capital and operating costs and convenient adaptation to the ASP make this approach very attractive. Possible trepidations can be associated with the increased BOD loading to the aeration basin.

As *heat treatment* alternatives achieve complete water evaporation, the performance of dewatering equipment can be critically important in process economics of all heat treatment approaches. In many circumstances costs due to supplemental fuel for low heat value sludges and the air pollution control equipment for gaseous and particulate emission control can be prohibitively high. Incineration is the most popular and perhaps most economical form of heat treatment. Sludge incineration and ash disposal costs of \$350–\$1042/dry tonne estimated by Environment Canada for four wastewater treatment facilities in 1989 (Campbell, 1989) underscore the expensive nature of the heat treatment option. In addition, pulp and paper mills' sludge incineration is often associated with operational problems such as sludge handling, bark and sludge mixture consistency variations and the downgraded boiler capacity because of high water content, making this approach less attractive. Particulate and gaseous emissions requiring air pollution control equipment remain additional issues with sludge incineration.

Chemical oxidation alternatives for sludge reduction appear to be the most expensive ones. Potential problems associated with SCWO in commercial operation may include corrosion of equipment (in particular when chlorides are present and the pH is low) and deposition of pyrolytic char and salts in the reactor (US EPA, 1992). A drop in reactor pH can be expected as a result of low molecular weight organic acids and dissolution of CO_2 in the aqueous phase. The development of this process is on going, particularly regarding the fabrication material for the reactor vessels, which have to be safe and durable. The process is capital, maintenance and energy intensive. Regenerative heat exchangers can be used for energy efficiency but they suffer from their own set of problems. As with the SCWO, wet air oxidation generally provides particulate matter-free operation since it involves flameless oxidation. Although sludge dewatering is not required, process economics can be improved with higher sludge solids. A major drawback of wet air oxidation is the fact that it produces high-strength liquors, which need to be recycled to the treatment plant (Shanableh, 2000) requiring increased aeration capacity. Furthermore, if aggressive process conditions (high temperature and pressure) are required for satisfactory oxidation of a process stream, the use of advanced and expensive construction materials will make the process highly capital intensive. Because of the odorous and energy intensive nature of the wet air oxidation process, at least two of its major installations in the U.S. were abandoned in favour of other technologies.

Long retention times have often limited the use of *sludge digestion*. Recent advances in sludge pre-conditioning allow significant reductions in digestion times thus rendering sludge digestion less capital intensive and therefore more attractive for the pulp and paper industry. Excessive foaming is often a problem with both aerobic and anaerobic digestion of sludge. Control methods are available but they are not always effective since the causes of foaming are not always fully understood. The benefits of aerobic digestion include the supernatant having a lower BOD and lower installation costs with fewer operational difficulties than other methods of digestion. The major disadvantages of aerobic digestion are its high-energy requirements for aerating and mixing of the sludge and its reduced efficiency during cold weather. Also, aerobic digestion is often found to be detrimental to the dewatering properties of sludge (Novak et al., 1977). Among a multitude of aerobic processes investigated to date, the two-stage ATAD process offers certain advantages compared to conventional aerobic digestion. The ATAD process requires solids retention times of 6–8 days, which is attractive. Owing to the slow growth kinetics of methanogens, anaerobic digestion is inherently a slow process. Conventional anaerobic digesters are designed for a solids retention time of at least 20–30 days, resulting in high construction costs. However, it has been shown that mechanical, thermal, chemical or biological means, or a combination of them, can efficiently solubilize biosolids allowing anaerobic digestion to complete in as little as 5–6 days without negatively impacting gas production or volatile fatty acids' destruction. With these advancements in the preconditioning of sludge, anaerobic digestion seems to be another feasible alternative. However, anaerobic sludge digestion suffers from high capital costs, operational challenges and is detrimental to dewatering process. Operator experience is vital for the smooth digester operation.

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