Check for updates

Saline sewage treatment and source separation of urine for more sustainable urban water management

G. A. Ekama, J. A. Wilsenach and G. H. Chen

ABSTRACT

While energy consumption and its associated carbon emission should be minimized in wastewater treatment, it has a much lower priority than human and environmental health, which are both closely related to efficient water quality management. So conservation of surface water quality and quantity are more important for sustainable development than green house gas (GHG) emissions per se. In this paper, two urban water management strategies to conserve fresh water quality and quantity are considered: (1) source separation of urine for improved water quality and (2) saline (e.g. sea) water toilet flushing for reduced fresh water consumption in coastal and mining cities. The former holds promise for simpler and shorter sludge age activated sludge wastewater treatment plants (no nitrification and denitrification), nutrient (Mg, K, P) recovery and improved effluent quality (reduced endocrine disruptor and environmental oestrogen concentrations) and the latter for significantly reduced fresh water consumption, sludge production and oxygen demand (through using anaerobic bioprocesses) and hence energy consumption. Combining source separation of urine and saline water toilet flushing can reduce sewer crown corrosion and reduce effluent P concentrations. To realize the advantages of these two approaches will require significant urban water management changes in that both need dual (fresh and saline) water distribution and (yellow and grey/brown) wastewater collection systems. While considerable work is still required to evaluate these new approaches and quantify their advantages and disadvantages, it would appear that the investment for dual water distribution and wastewater collection systems may be worth making to unlock their benefits for more sustainable urban development.

Key words | biological sulphate reduction, urine separation, wastewater treatment

G. A. Ekama (corresponding author)
Water Research Group,
Department of Civil Engineering,
University of Cape Town,
Rondebosch,
7700, Cape,
South Africa
E-mail: *george.ekama@uct.ac.za*

J. A. Wilsenach
Virtual Consulting,
PO Box 1730,
Brooklyn Square,
0075, Pretoria,
South Africa
and
Formerly Natural Resources and Environment,
CSIR, 11 Jan Cilliers Str.,
Stellenbosch,
Cape,
South Africa

G. H. Chen
Department of Civil Engineering,
Hong Kong University of Science and Technology,
Clear Water Bay, Kowloon,
Hong Kong,
China

INTRODUCTION

Early wastewater management originated at the end of the 19th century from a growing attention to reduce the causes of waterborne diseases (cholera, dysentery, etc.), but this has slowly changed over the last century from a system of prevention of diseases in urban society towards a system for protecting the natural environment. So in recent years, new technical developments are assessed in terms of their sustainability. Although sustainability has different meanings in different fields, it is accepted that in the wastewater treatment field (i) reducing fresh water consumption, (ii) improving effluent quality from wastewater treatment plants (WWTPs) with (iii) recovery and reuse of nutrients – nitrogen (N), phosphorus (P), potassium (K), magnesium (Mg) and water – and (iv) decreasing energy requirements are indicators of sustainability. In this paper two opportunities are explored that open the way for greater sustainability of urban wastewater treatment (1) flushing toilets with saline (e.g. sea) water for fresh water savings, reduced sludge production and treatment energy demand and (2) source separation of urine for reduced WWTP size and nutrient recovery.

Svardal & Kroiss (2011) state the issues around energy and water clearly. While the power consumption (mainly based on fossil fuels) in the developed countries is in the range 5–10 kW/person, and the energy available in wastewater even less than two orders of magnitude lower at around 0.02 kW/person, energy recovery from wastewater will not solve the energy problem – the treated wastewater

doi: 10.2166/wst.2011.403

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

has far greater value as a potential fresh water source than the energy it contains or is consumed in its treatment. However, this is not a reason to waste energy at WWTPs, on the contrary, efforts to save/generate energy at WWTPs are important but these should not be at the expense of effluent quality because water quality conservation is more important for sustainable development than the (small) reduction in energy demand. Similarly, regarding carbon emissions by WWTPs, Ekama (2009) shows that this is negligible in comparison with other human activities. Provided methane gas is combusted (either to generate power or flared) and the residual biodegradable organics in the final sludge is the same, then the activated sludge based WWTP layout does not affect its carbon emission much, which is about 60% of influent C or 20 g CO_2-C/person equivalent (PE)/d. This 20 g CO_2-C/(PE.d) is only about (i) 50% of the 40 g CO_2-C/(PE.d) generated at the fossil fuel power station to treat the wastewater, (ii) 12% of the 180 g CO_2-C/(person.d) exhaled by a person with a 6,300 kJ/d diet, (iii) 1% of the 2,000 g CO_2-C/(person.d) emitted by motor car driving at 30 km/(person.d), (iv) 0.2% (1/500th) of the 10,000 g CO_2-C/(person.d) emitted at the fossil fuel power plant for domestic electricity utilization at 27 kWh/(PE.d) or 1.1 kW/person or (v) 0.044% (~1/2,000th) of the 45,000 g CO_2-C/(person.d) emitted at the fossil fuel power plant for total power consumption at 5 kW/person. So while minimization of energy requirement for wastewater treatment is an important goal, clearly this has a far lower priority than the human and environmental health which is closely related to efficient water quality management.

WATER QUALITY MANAGEMENT

N and P sources and sinks

In the production of inorganic fertilizers, large quantities of N and P are extracted from primary sources. Nitrogen is extracted from the atmosphere for the production of ammonia with the Haber-Bosch process ($N_2 + 3H_2 \rightarrow 2NH_3$) and phosphate ore is mined. Phosphate bearing ore is a non-renewable resource and its quantity and quality is declining. Some countries have stopped its import because it was found to be the greatest contributor of cadmium to agricultural land via the application of inorganic fertilizers. While P and N removals have been implemented for many years at wastewater treatment plants (WWTP), dwindling P reserves have started a drive for P recovery from wastewater for reuse.

The crops produced with inorganic fertilizers are consumed by livestock and the crops and livestock are consumed by people. The generally high protein diet of middle and high socio-economic people results in high concentrations of ammonia in urban wastewater, mostly from the urine. Reducing meat consumption would be an environment friendly and pro-sustainability social action not only to reduce the N content of urine but meat production has a very high specific fresh water consumption (l/kg). While WWTPs have been modified over the years to remove N and P (point source control), the largest contributor to eutrophication of receiving surface water systems is the runoff of N and P from agricultural land (diffuse pollution) due to high fertilizer application rates to deliver high crop yields. So over the years, the surface water systems have been a N and P sink of the atmospheric N and mined P, and as a consequence have shown rapid deterioration due to widespread algal and water plant blooms through eutrophication. The discharge of acid mine drainage (AMD) from operating or abandoned mines has also had its negative impact on surface water quality. The high sulphate and metals concentrations have caused increasing salination, which, if the total dissolved solids (TDS) increases above 1,000 mg/l, renders the water unusable for agriculture and difficult to treat to avoid corrosion and aggression to water storage and distribution systems.

Decline in surface water quality

Over the past decade another serious surface water quality problem has emerged. Concentrations of hormones, medicine residues and pharmaceutical compounds, excreted by people are increasing in surface water and the environment. Also industry produces many new organic compounds and chemicals, the effect of which on the environment and people is unknown. Many of these organic compounds are not readily degraded in WWTPs. The release of hormones, medicine residues and pharmaceutical compounds, generically called endocrine disruptors or environmental oestrogens, over the past years have caused severe disruption in the environment affecting the gender of fish and reptiles at WWTP discharges and believed to be detrimental to human health (Cadbury 1997). The removal of these compounds is becoming increasingly important for the environment sustainability and human health, in particular due to increasing indirect reuse of treated water and future direct reclamation of treated wastewater for potable supply, especially in water scarce countries like Namibia, South Africa and Australia.

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

URINE SEPARATION AS A STRATEGY TO ADVANCE SUSTAINABILITY

Source separation of urine can improve surface water quality

Sustainability considerations demand that the N, P and endocrine disruptor burden on surface water be reduced to conserve surface water quality, especially in view of rapidly increasing water demand for food production and urban use as more and more people become urbanized and connected to the water supply and sanitation networks and increase their quality of life and demand for food. To sustain this growth will require greater recycle and re-use of available water which in turn demands better effluent quality of used water. One way this can be achieved is through separation of urine at source. The principles of cleaner production and resource recovery already applied in industrial waste management would point to not diluting the small flow of concentrated urine into the large wastewater flow and rather treat it separately as a concentrate and recover the nutrients from it. Source separation of urine not only offers improved conservation of water quality but also water quantity. Various studies by Larsen & Gujer (1996, 2001), Jönsson & Vinnerås (2007), Wilsenach & van Loosdrecht (2004), Wilsenach *et al.* (2003, 2007, 2009) and Larsen & Lienert (2007) show that source separation of urine with separate treatment and nutrient recovery can bring significant surface water benefits (quality improvement and quantity savings) and unlock greater treatment capacity at existing WWTPs, all aspects aligned with greater sustainability of urban water and sanitation systems.

Impact of urine separation on wastewater treatment

Modern biological nutrient removal activated sludge (BNRAS) WWTPs remove organics, nitrogen (N) and phosphorus (P) to reduce receiving water de-oxygenation and eutrophication. Where separate storm and wastewater collection is practiced, these WWTPs receive predominantly under dry weather conditions, a mixture of brown (faeces and toilet paper), yellow (urine) and grey (kitchen and bathroom) waters. From previous wastewater separation studies, urine contains about 80% of the N, 50% of the P (Otterpohl 2002) and 65% of pharmaceutical residues (Lienert *et al.* 2007) in mixed municipal wastewater, but it represents less than 1% of the total wastewater volume (Otterpohl 2002). Thus, implementing separate urine collection will considerably reduce the nutrient loads on municipal WWTPs, but will this make a significant difference to BNRAS effluent quality and economics? Here key questions are: (1) With nitrification being the BNRAS size governing bioprocess via the sludge age (Ekama 2010), will this still be required with wastewater comprising only brown and grey water? (2) With a significantly reduced P load, will the effluent P concentration also decrease and to what value? (3) With eutrophication prevention requiring ever decreasing effluent N and P concentrations, e.g. 0.1 mg P/l and 3 mg TN/l [Total N (TN) = nitrate + nitrate (NO_x) + free and saline ammonia (FSA) + Organic N (OrgN)], is achieving such limits related to the N and P loads or by other factors that set the limits of technology (LOT) of the BNRAS system (Neethling *et al.* 2010; Pagilla *et al.* 2009), e.g. affinity constants in Monod kinetics? (4) What will be the chemical oxygen demand (COD): total Kjeldahl N (TKN): total P (TP) ratio of urine separated (grey and brown) municipal wastewater?

The minimum N and P requirements for activated sludge production depend on sludge age, the shorter the sludge age the higher the N and P requirements and decrease when settled wastewater is treated. For normal (non-biological excess P removal, BEPR) activated sludge N/VSS (f_n) and P/VSS (f_p) ratios of 0.10 g N/g VSS and 0.025 g P/g VSS respectively, the influent TKN/COD and TP/COD ratios for raw and settled wastewater at 5, 8, 10 and 20 d sludge age are given in Table 1 (calculated from Henze *et al.* 2008). At these ratios, all of the influent N and P will be used for sludge production so that no nitrification denitrification (ND) and BEPR will be required. From Table 1, for 1,000 mg COD/l, the influent TKN concentration of mixed grey and brown wastewater would need to be below 31 to 23 mg N/l for raw wastewater and 25 to 16 mg N/l for settled wastewater to eliminate ND from the BNRAS system and the TP concentration below 9.3 to 6.8 mg P/l for raw wastewater and 7.6 to 4.9 mg P/l for settled wastewater to eliminate BEPR from the BNRAS system. To unlock capacity at existing BNRAS WWTPs,

Table 1 | Influent TKN/COD and TP/COD ratios for complete removal of N and P without requiring ND and BEPR ($f_n = 0.10$ g N/g VSS and $f_p = 0.025$ g P/g VSS)

Sludge age (d)	Raw wastewater TKN/COD	Raw wastewater TP/COD	Settled wastewater TKN/COD	Settled wastewater TP/COD
5	0.031	0.0093	0.025	0.0076
8	0.028	0.0084	0.022	0.0066
10	0.026	0.0079	0.020	0.0061
20	0.023	0.0068	0.016	0.0049

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

it is much more important for ND to be eliminated from the system because then sludge ages can be halved from 15–20 to 8–10 d. BEPR can be achieved at short sludge ages (5–10 d) (Wentzel *et al.* 1989; Mamais & Jenkins 1992).

In their simulation study of the impact of urine separation on WWTPs, Wilsenach & van Loosdrecht (2004) utilized a BNR system called the biological/chemical P and N removal process (BCFS, Barat & van Loosdrecht 2006). The results show a 70% decrease in total effluent nitrogen content with increasing urine separation up to 60% and nearly 100% P removal without urine separation. At greater urine separation, the model showed no improvement in effluent quality due to the magnitude of the Monod affinity constants in the kinetic equations. Do these affinity constants also set the limit of technology (LOT) of real WWTPs? The predicted effluent N and P concentrations only decreased below the affinity constant limit concentrations once nutrient limitation took place.

BENEFITS OF SOURCE SEPARATION OF URINE

Reduction in water consumption

Separate urine collection reduces flush water usage resulting in lower per capita wastewater generation and therefore also in a lower water consumption and water saving. There will be additional costs in installing urine separation toilets and collection systems. However, these will be once off costs which can be recovered by continuous savings in water and increased WWTP capacity. Although with reduction in water, N and P fluxes to the WWTP, some savings in energy and operating costs will be made, these will be small because the dominant costs of WWTPs are fixed costs (Svardal & Kroiss 2011).

Increased WWTP capacity

Reduced hydraulic load

By implementing separate urine collection and treatment the capacity of existing WWTPs may be increased. Around 35l/(p.d) is used to flush urine in conventional water borne sanitation (Jönsson *et al.* 1997; STOWA 2002), while separate toilet urinals and no mix toilets may eventually require almost no flushing water. This implies that with separate urine collection, the flow of wastewater to WWTPs will be less by about 35l/(p.d) (ignoring the volume of urine per person) or about 15% hence leading to an increase in brown water that can be treated. Even though the BNRAS reactor volume is governed by the sludge age and organic (and inorganic) load on the WWTP, the reduced hydraulic load does create additional WWTP capacity by reducing the overflow rate in the secondary settling tanks. Wilsenach & van Loosdrecht (2004) found that a capacity increase of 40% can be achieved.

Reduction in sludge age

Because the requirement to nitrify governs the sludge age of the biological nutrient removal activated sludge (BNRAS) system (Ekama 2010), strategies seeking to reduce sludge age have been implemented over the past two decades. For maximum specific growth rates of nitrifiers at 20 °C (μ_{Am20}) around 0.45/d, to guarantee nitrification, the sludge age of the single sludge system must be around 20 to 25 days at 14 °C, if 40 to 50% of the sludge mass in the system is aerated. Such long sludge ages result in large biological reactors per megalitre (Ml) of wastewater (WW) treated. To reduce the sludge age, and hence the biological reactor volume per Ml WW treated, internal fixed media have been placed in the aerobic reactor (Wanner *et al.* 1988; Sen *et al.* 1994) or external nitrification has been implemented (Sorm *et al.* 1996; Hu *et al.* 2000; Vestner & Gunthert 2001; Muller *et al.* 2006). By achieving nitrification in an external fixed media reactor like old trickling filters at WWTPs, nitrifiers no longer need to be sustained in the activated sludge with the result that the sludge age of the BNRAS system can be reduced from the usual 15 to 20 d to around 5 to 8 days.

The reduction in sludge age increases the WW treatment capacity by some 50% or, alternatively, reduces the biological reactor volume requirement per Ml WW treated by about a 1/3rd, without negatively impacting either biological N or P removal: In fact, a reduction in sludge age increases both biological N and P removal per mass organic load (Table 1) and this would be particularly beneficial for low temperature WWs (10–15 °C).

If the influent TKN/COD ratio can be reduced below those in Table 1 by urine separation, then this would be another and perhaps better way to reduce BNR system sludge age. So here the key question is what degree of urine separation has to be achieved to not require ND at the WWTP? The answer will depend on the diet of the community that generates the wastewater (low protein diets will produce low N content urine) amongst other things. The N, P and K content of source separated urine from low income informal communities is less than half that from middle

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

income people, presumably due to the higher protein diet of the latter (Wilsenach *et al.* 2009).

Nutrient recovery

Even though only a small proportion of mined P ends up in food (~3%, Allenby 2010), separate urine treatment also opens the possibility of nutrient recovery (N, P, Mg, K) for reuse. P can be recovered from urine by precipitating the phosphate as struvite with $Mg(OH)_2$ dosing or seawater treatment (Jiang *et al.* 2011). However, practical solutions for large cities have not been developed yet, but small pilot projects in Sweden, Tibet and Holland have demonstrated its potential (Tilley *et al.* 2008). At this time inorganic fertilizers can be mass produced industrially much more cheaply and efficiently relative to production from urine treatment. So at this stage the driver for adopting urine separation is to save water, improve WWTP effluent quality and keep endocrine disruptors and environmental oestrogens out of the water cycle. In time, nutrient recovery (N, P, Mg, K) from WWTP sludge liquors and separated urine may become economically viable as P resource quality and quantity decline and/or natural gas for the production of ammonia gets expensive. Currently fertilizer (struvite) manufacture from urine is viable in developing agricultural communities (e.g Tibet, Tilley *et al.* 2008) because it is easier to make at small scale than 'import' and buy expensive (relatively) industrially manufactured inorganic fertilizers.

TREATMENT OF URBAN WASTEWATER WITH BIOLOGICAL SULPHATE REDUCTION (BSR)

Reducing sludge production at WWTPs

Primary and/or secondary sludge thickening, stabilization, dewatering, disinfection and disposal presents the most complex problems in WWT (Metcalf & Eddy 2004, p. 1449) and can account for over 50% of WWTP operating costs (Marx *et al.* 2004). So various techniques have been investigated to reduce this cost – chemical, mechanical/physical and thermal treatment (Paul *et al.* 2006; Strünkmann *et al.* 2006; Yamaguchi *et al.* 2006). While considerable reduction in sludge production can be achieved, this is at significantly increased energy consumption – for some techniques in the order of as much as aeration, the highest energy demand at activated sludge WWTPs. While sludge minimization is useful for inner city WWTPs in that it reduces sludge handling and transport costs, the overall (global) loss due to the increase in energy consumption far outweighs the local (WWTP) gain. In contrast, a significant reduction in sludge production together with an overall (global) reduction in energy consumption can be achieved with saline sewage treatment.

Sea water toilet flushing and sulphate reduction

Water reuse has appeared to be an attractive option where low-quality freshwater use is acceptable, such as for irrigation and toilet flushing (Ogoshi *et al.* 2001). However, due to the difficulty of detecting cross-connections between the potable water and the low quality fresh water distribution networks, cross-connections have occurred in the past and compromised public health (van Lieverloo *et al.* 2007). Consequently, only highly treated reclaimed water is allowed to be distributed for toilet flushing to reduce the public health risk associated with cross-connections (Asano *et al.* 2007). In fact, many states like California (US State of California 2001) have prohibited the supply of reclaimed water to residential developments. Therefore, despite toilet flushing representing 20–30% of domestic water consumption (Terpstra 1999; Asano *et al.* 2007), use of reclaimed water for flushing has found limited application so far (e.g. <0.1% in Florida 2008). While the water-flush toilet is a very user-convenient and efficient form of sanitation, the use of high quality potable freshwater for this purpose is unsustainable in water-scarce areas.

In Hong Kong, toilets are flushed with pre-filtered and chlorinated sea water. This saves 1/3rd fresh water, which all has to be transported long distances at four times the cost of sea water distribution (Li *et al.* 2005; Tang *et al.* 2007). The dual water distribution system – one for sea water in plastic pipes and one for potable water in steel pipes – has very seldom been cross-connected because plumbers can measure (via conductivity or taste) the water in the colour coded pipes without endangering their health. Indeed, even if a cross connection does take place, the mixed water would (i) not impose an immediate public health threat and (ii) be detected quickly due to the increase in salinity.

Sulphate causes sewer crown corrosion

The disadvantage of toilet flushing with sea water is that the high sulphate concentration (~200 mg SO_4^{2-}-S/l) in the wastewater exacerbates sewer crown corrosion. In the presence of organics (electron donors) sulphate reducing

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

bacteria (SRB) produce sulphide gas which escapes to the head space above the water in the sewer. Sulphide oxidizing bacteria on the upper walls of the sewer oxidize with oxygen the sulphide to sulphuric acid ($H_2S + 2O_2 \rightarrow H_2SO_4$), which corrodes the crown of the sewer – with time holes form and the sewer caves in resulting in significant repair work. Usually, local authorities enforce municipal bylaws to limit sulphate ingress to sewers to minimize this problem. While crown corrosion and cave-ins do take place in the Hong Kong sewerage system, this problem is preferable and less expensive than distributing 40% more fresh water for toilet flushing (Tang *et al.* 2007).

Sulphate reduction sewage treatment

As so often happens, necessity is the mother of invention and the problem also points to a solution. A wastewater treatment system has been invented that uses the sulphur system as electron carrier in the biological redox reactions that treat the wastewater, i.e. the sulphate reduction, autotrophic denitrification and nitrification integrated (SANI®) process (Figure 1, Lau *et al.* 2006; Wang *et al.* 2009). This novel system uses biological sulphate reduction to breakdown the organics (COD) in the wastewater which (i) eliminates oxygen utilization except for nitrification (which is partially recovered by denitrification) and (ii) decreases sludge production to very low quantities due to the use of anaerobic bioprocesses – essentially only the unbiodegradable particulate organics of the wastewater. Mathematical models of the SANI process are being developed based on laboratory (Lu *et al.* 2009) and pilot scale ($10\,m^3/d$, Lu *et al.* 2010) plants to quantify the reductions in oxygen demand and sludge production, the most economically and environmentally costly parts of wastewater treatment. For this, the models developed for biological sulphate reduction (BSR) from acid mine drainage (AMD) using primary sewage sludge as electron donor (Biosure® process) is helpful because primary sludge is a product of municipal wastewater (Loewenthal *et al.* 2005; Poinapen *et al.* 2009a, b, c, d; Poinapen & Ekama 2010a, b).

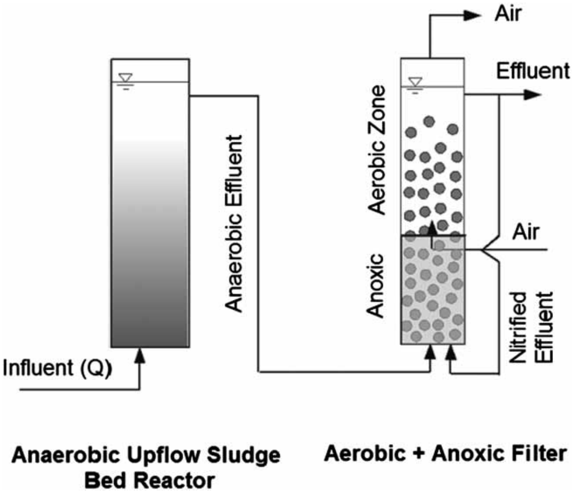

Figure 1 | SANI process flow scheme.

Source separation of urine

Disadvantages of seawater flushing and saline sewage treatment in the SANI process are (i) that biological excess P removal cannot be included in the system and (ii) the exacerbation of sewer crown corrosion. However, these problems can be ameliorated with source separation of urine. The urine collected in urine diversion toilets with sea water flushing is nitrified decentrally, during which also P can be recovered. The nitrified liquor is then discharged to sewer with the grey (kitchen and bathroom) water and brown (toilet faeces) water. The nitrate in the sewer will (i) raise the redox potential of wastewater, decrease biological sulphate reduction and preserve the sulphate for organic removal in the SANI process, (ii) denitrify the nitrate to nitrogen gas and remove dissolved readily biodegradable organics. Because urine contains about 50% of the P, 80% of the N and 67% of the medical residues (Otterpohl 2002; Lienert *et al.* 2007), the SANI process effluent will contain half the P and only 1/3rd of the medical residues compared with not implementing source separation of urine, at the same time reducing sewer corrosion.

Although saline sewage appears to have an adverse impact on some treated effluent reuse practices such as irrigation, the use of saline water toilet flushing need not affect the overall water reclamation program if it is properly integrated into water reuse systems through suitable engineering designs. In fact, if required desalination of treated effluent by low energy reverse osmosis (RO) would be more economic and consume less energy than desalination of seawater because the salinity of saline sewage is only about one-third of that of seawater.

Most of Hong Kong's wastewater is currently treated by chemically enhanced primary treatment (CEPT) followed by ocean discharge (DSD 2005). Secondary biological treatment of this primary effluent is considered necessary in future due to its high dissolved COD and ammonia concentrations. With source separation of urine, nitrifying the separated urine locally (at tower block residential level)

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

and then discharge of nitrified urine to the sewer results in (i) denitrification in the sewer with removal of nitrate and dissolved organics (COD), (ii) reduction in crown corrosion and possibly (iii) a CEPT primary effluent with sufficient low COD and ammonia concentrations to obviate secondary treatment (Jiang *et al.* 2011).

With source separation of urine, the nitrification denitrification part of the SANI process can be reduced in size but cannot be eliminated due to the release of the ammonia during the breakdown of the proteinaceous organics in the anaerobic reactor. Also the sulphide not oxidized to sulphate by denitrification, will need to be oxidized (to S_0 or SO_4^{2-}) to reduce the sulphide concentration of the SANI process effluent. Compact high density suburban areas (like in Hong Kong) will require relatively little transport of source separated urine and so are suited to decentralized treatment of source separated urine. For the sprawling suburban residential areas, perhaps the separated urine can be discharged to sewer at night as a 'plug' during the low flow period and pumped out of sewer at the WWTP before the arrival of the morning high flow with the high grey/brown water organics load, or the nightman, who made the rounds more than a century ago collecting the contents of buckets before waterborne sanitation, will return to collect canisters of yellow fluid with solid waste.

Using acid mine drainage as saline water source

Flushing toilets with saline water is not restricted to seawater. Acid mine drainage (AMD) is a potential water source for this also with the added benefit of using municipal wastewater organics for BSR. In South Africa, AMD from disused and operating mines is a significant polluter of surface water quality and the economics of BSR are governed by the cost of the organic electron donor. The eastern, central and western basins of the Witwatersrand of Gauteng produce an estimated 300 Ml/d. If this AMD is treated with limestone neutralisation and aeration to remove iron and heavy metals, and calcium sulphate precipitation (Morgan *et al.* 2004), the resulting water would still have a sulphate concentration of around 1,000 mg SO_4^{2-}-S/l. After say 1/3rd dilution with the grey and brown municipal wastewater, the mixed wastewater sulphate concentration would be around 350 mg SO_4^{2-}-S/l, higher than the wastewater with seawater toilet flushing in Hong Kong (200 mg SO_4^{2-}-S/l), but necessarily so due

Table 2 | Comparison of conventional, sea water toilet flushing and source separation of urine strategies on the urban water cycle

Criterion	1. Conven'l/membrane	2. Seawater flushing	3. Urine separation	Combination (2) and (3)
Distribution	Single	Dual	Single	Dual
Collection	Single	Single	Dual	Dual
Sewer corrosion	Normal	High	Normal	Normal
Energy demand	High/V. high	Very low	High	Low
Sludge production	High	Very low	High	Very low
Sludge age	Long	Not applic	Low	Not applic
Reactor volume	Large/small	Large	Small	Large
Sludge treatment	High	No	High	No
Energy recovery	Yes	No	Yes	No
Nutrient recovery	Yes	No	Yes	Yes
Effluent quality	V. good	Fair	Good	Good
N&P removal	Yes	No P rem.	Not req'd	No P rem
Water reuse value: ….N&P concs	Low	High P	Low	Some P
… Salinity	Low	High	Low	High
… Suspended solids	Low/V.low	High	Low	High
… Pathogens	High/low	Low	High	Low
…… [a]ED & EO's	High	High	Low	Low
Water saving	No	Yes	No	Yes
Water reclamation	Yes	Yes	Yes	Yes

[a]Endocrine disruptors and environmental oestrogens.

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

to the higher COD concentration in SA wastewater (require 0.5 g SO_4^{2-}-S/g BiodegCOD). The 5 million people of Gauteng produce some 750 Ml/d at 1,000 mg COD/l. If half of this population could be reached with a dual water distribution system and one third of the water consumption is for toilet flushing, then 250 Ml/d would be required. The 300 Ml/d treated AMD would be sufficient to supply this and also provide the opportunity to repair leaks in the existing potable water distribution system for significant additional water savings. The problem would be to not oxidize the sulphide back to sulphate after the upflow anaerobic sludge bed (UASB) BSR reactor but remove it as S so the WWTP is also an S recovery plant. A high S removal is important to keep the salinity of the receiving surface water low for downstream (of Gauteng) agriculture. P removal would also be required which would have to be done by pre-precipitation with iron sulphate so the additional sulphate can also be removed to keep salinity low. Residual metals in the distributed AMD will be precipitated with sulphide in the BSR UASB reactor. Sulphide oxidation to S can be done chemically with $FeCl_3$ and the resulting $FeCl_2$ water regenerated to ferric with iron oxidizing bacteria (Loewenthal *et al*. 2008). Cross contamination in the dual distribution network would be a major problem because the treated AMD would have a much higher public health threat than seawater. Also source separation of urine and decentralized nitrate addition to sewer would be required to reduce sewer crown corrosion and remove N. The result is quite a complex collection and biological–chemical treatment system but provides a combined AMD and municipal wastewater treatment with a 'cheap' carbon source for BSR. Treated AMD distribution could reduce WWTP energy consumption, reduce fresh water consumption, use AMD as a water source and municipal wastewater organics for BSR, all in one urban water management system.

CONCLUSION

In this paper the impact of seawater toilet flushing and source separation of urine were considered to reduce fresh water consumption, reduce WWTP energy demand and improve effluent quality. A comparison with the current conventional approach of these two strategies on the urban water cycle – distribution, collection, treatment, effluent quality and reuse – is given in Table 2. While both these strategies will require significant urban infrastructure investment for dual distribution and collections systems, both hold major promise for more sustainable urban development.

ACKNOWLEDGEMENTS

At the University of Cape Town, this research was supported by the Water Research Commission, the National Research Foundation and the University of Cape Town and is published with their permission.

REFERENCES

Allenby, B. 2010 Earth systems engineering and water. *Presented at 7th IWA LET 2010 Conference*, Phoenix, AZ, USA.

Asano, T., Burton, F. L., Leverenz, H. L., Tsuchihashi, R. & Tchobanoglous, G. 2007 *Water Reuse: Issues, Technologies, and Applications*. Metcalf & Eddy/AECOM, McGraw-Hill, New York. ISBN-13:978-0-07-145927-3.

Barat, R. & van Loosdrecht, M. C. M. 2006 Potential phosphorus recovery in a WWTP with the BCFS® process: interactions with the biological process. *Water Res.* **40** (19), 3507–3516.

Cadbury, D. 1997 *The feminization of Nature – Our Future at Risk*. Penguin, London.

DSD 2005 Annual Report 2004–2005. Drainage Services Department of the Hong Hong Government, Hong Kong, SAR, China.

Ekama, G. A. 2009 Using bioprocess stoichiometry to build a steady state plant wide wastewater treatment plant model. *Water Res.* **43** (8), 2101–2120.

Ekama, G. A. 2010 The role and control of sludge age in biological nutrient removal activated sludge systems. *Water Sci. Technol.* **61** (7), 1645–1652.

Florida Department of Environmental Protection 2008 *Reuse Inventory Report* (2010; www.dep.state.fl.us/water/reuse/inventory.htm).

Henze, M., van Loosdrecht, M. C. M., Ekama, G. A. & Brdjanovic, D. 2008 *Biological Wastewater Treatment: Principles, Modelling and Design*. IWA Publishing, London, ISBN 13: 9781843391883.

Hu, Z.-R., Wentzel, M. C. & Ekama, G. A. 2000 External nitrification in biological nutrient removal activated sludge systems. *Water SA* **26** (2), 225–238.

Jiang, F., Chen, Y., Mackey, H. R., Chen, G. H. & van Loosdrecht, M. C. M. 2011 Urine nitrification and sewer discharge to realize in-sewer denitrification to simplify sewage treatment in Hong Kong. *Water Sci. Technol.* **64**, 618–626.

Jönsson, H., Stenström, T. A., Svensson, J. & Sundin, A. 1997 Source separated urine – nutrient and heavy metal content, water savings and faecal contamination. *Water Sci. Technol.* **35** (9), 142–152.

Jönsson, H. & Vinnerås, B. 2007 Experience and suggestions for collection systems for source separated urine and faeces. *Water Sci. Technol.* **56** (5), 71–76.

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

Larsen, T. A. & Gujer, W. 1996 Separate management of anthropogenic nutrient solutions (human urine). *Water Sci. Technol.* **34** (3–4), 84–97.

Larsen, T. A. & Gujer, W. 2001 Waste design and source control lead to flexibility in wastewater management. *Water Sci. Technol.* **43** (5), 309–318.

Larsen, T. A. & Lienert, J. 2007 *Novaquatis Final Report. NoMix – A New Approach to Urban Water Management*. EAWAG, 8600, Duebendorf, Switzerland.

Lau, G. N., Sharma, K. R., Chen, G. H. & van Loosdrecht, M. C. M. 2006 Integration of sulfate reduction, autotrophic denitrification and nitrification to achieve low-cost excess sludge minimization for Hong Kong sewage. *Water Sci. Technol.* **53** (3), 227–235.

Li, X. Z., Luk, S. F. & Tang, S. L. 2005 Sustainability of toilet flushing water supply in Hong Kong. *Water Environ. J.* **19** (2), 85–90.

Lienert, J., Burki, T. & Escher, B. I. 2007 Reducing micro-pollutants with source control: substance flow analysis of 212 pharmaceuticals in faeces and urine. *Water Sci. Technol.* **56** (5), 87–96.

Loewenthal, R. E., Ristow, N. E., Sötemann, S. W., Wentzel, M. C. & Ekama, G. A. 2005 *Hydrolysis of Primary Sewage Sludge Under Methanogenic, Acidogenic and Sulphate-Reducing Conditions*. Final WRC Report on 1216/1/05, Water Research Commission, PO Box X03, Gezina, 0033, Pretoria.

Loewenthal, R. E., Morgan, B., Lahav, O. & Hearne, G. 2008 *Research on an Investigation into Sulphur Chemistry with Specific Application to Biological Sulphate Removal Processes*. WRC Report No 1244/1/04, Water Research Commission, Private Bag X3, Gezina, 0031, Pretoria, South Africa. ISBN:1-77005-043-4.

Lu, H., Wang, J., Li, S., Chen, G. H., van Loosdrecht, M. C. M. & Ekama, G. A. 2009 Steady-state model-based evaluation of sulfate reduction, autotrophic denitrification and nitrification integrated (SANI) process. *Water Res.* **43** (14), 3613–3621.

Lu, H., Wu, D., Tang, D. T. W., Chen, G. H., van Loosdrecht, M. C. M. & Ekama, G. A. 2010 Pilot scale evaluation of SANI process for sludge minimization and green house gas reduction in saline sewage treatment. In *Procs. IWA Leading Edge Technologies conference LET2010*, Phoenix, Arizona. 2–4 June.

Mackay, H. R., Zhu, X. J., van Loosdrecht, M. C. M. & Chen, G. H. 2011 An Economic Phosphorus Recovery from Source-Separated Urine Using Seawater from Seawater Toilet Flushing System. In *8th IWA Leading Edge Conference on Water and Wastewater Technology (LET2011), June 6–10, 2011, Amsterdam, The Netherlands*.

Mamais, D. & Jenkins, D. 1992 Effect of MCRT and temperature on enhanced biological phosphorus removal. *Water Sci. Technol.* **26** (5–6), 955–965.

Marx, C. J., Alexander, W. V., Johannes, W. G. & Steinbach-Kane, S. 2004 *A Technical and Financial Review of Sewage Sludge Treatment Technologies*. WRC Report No 1240/1/4. Water Research Commission, Private Bag X3, Gezina, 0031, Pretoria, South Africa. ISBN:1-77005-156-2.

Metcalf & Eddy 2004 *Wastewater Engineering: Treatment and Reuse*. McGraw-Hill, New York. ISBN:007-124140-X.

Morgan, B., Lahav, O., Hearne, G. & Loewenthal, R. E. 2004 *A Novel One Step Ambient Temperature Ferrite Process for the Removal of Metals from Acid Mine Drainage*. WRC Report No 1244/1/04, Water Research Commission, Private Bag X3, Gezina, 0031, Pretoria, South Africa. ISBN:1-77005-043-4.

Muller, A. W., Wentzel, M. C. & Ekama, G. A. 2006 Estimation of nitrification capacity of rock media trickling filters in external nitrification BNR. *Water SA* **32** (5), 611–618.

Neethling, J. B., Clark, D., Pramanik, A., Stensel, H. D., Sandino, J. & Tsuchihashi, R. 2010 WERF nutrient challenge investigates limits of nutrient removal technologies. *Water Sci. Technol.* **61**, 945–953.

Ogoshi, M., Suzuki, Y. & Asano, T. 2001 Water reuse in Japan. *Water Sci. Technol.* **43** (10), 17–23.

Otterpohl, R. 2002 Options for alternative types of sewerage and treatment systems directed to improve the overall performance. *Water Sci. Technol.* **45** (3), 149–158.

Pagilla, K., Sattayatewa, C., Urgen-Demirtas, M. & Baek, S. 2009 Effect of influent N speciation on organic nitrogen occurrence in activated sludge process effluents. In *Procs 2nd IWA specialized Conference Nutrient Management in Wastewater Treatment Processes*. Krakow 6–9 Sept., pp. 221–228.

Paul, E., Camacho, P., Lefebvre, D. & Ginestet, P. 2006 Organic matter release in low temperature thermal treatment of biological sludge for reduction in excess sludge production. *Water Sci. Technol.* **54** (5), 59–68.

Poinapen, J. & Ekama, G. A. 2010a Biological sulphate reduction using primary sewage sludge in a upflow anaerobic sludge bed reactor – Part 5: development of a steady state model. *Water SA* **36** (3), 193–202.

Poinapen, J. & Ekama, G. A. 2010b Biological sulphate reduction using primary sewage sludge in a upflow anaerobic sludge bed reactor – Part 6: development of a dynamic simulation model. *Water SA* **36** (3), 203–214.

Poinapen, J., Wentzel, M. C. & Ekama, G. A. 2009a Biological sulphate reduction using primary sewage sludge in a upflow anaerobic sludge bed reactor – Part 1: feasibility study. *Water SA* **35** (5), 525–534.

Poinapen, J., Ekama, G. A. & Wentzel, M. C. 2009b Biological sulphate reduction using primary sewage sludge in a upflow anaerobic sludge bed reactor – Part 2: modification of simple wet chemistry analytical procedures to achieve COD and S mass balances. *Water SA* **35** (5), 535–542.

Poinapen, J., Ekama, G. A. & Wentzel, M. C. 2009c Biological sulphate reduction using primary sewage sludge in a upflow anaerobic sludge bed reactor – Part 3: performance of the system at 35 and 20 °C. *Water SA* **35** (5), 543–552.

Poinapen, J., Ekama, G. A. & Wentzel, M. C. 2009d Biological sulphate reduction using primary sewage sludge in a upflow anaerobic sludge bed reactor – Part 4: measuring the settling rate of bed sludge solids fractions in the reactor. *Water SA* **35** (5), 553–560.

Sen, D., Mitta, P. & Randall, C. W. 1994 Performance of fixed film media integrated in activated sludge reactors to enhanced nitrogen removal. *Water Sci. Technol.* **30** (11), 13–24.

Sorm, R., Bortone, G., Saltarelli, R., Jenicek, P., Wanner, J. & Tilche, A. 1996 Phosphate uptake under anoxic conditions

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest

and fixed film nitrification in nutrient removal activated sludge system. *Water Res.* **30** (7), 1573–1584.

STOWA 2002 *Separate Urine Collection and Treatment: Options for Sustainable Wastewater Systems and Mineral Recovery*. Report No 2001.39, Stichting Toegepast Onderzoek Waterbeheeer, Postbus 8090, 3503 RB, Utrecht, Netherlands.

Strünkmann, G. W., Müller, J. A., Albert, F. & Schwedes, J. 2006 Reduction of excess sludge production using mechanical disintegration devices. *Water Sci. Technol.* **54** (5), 69–76.

Svardal, K. & Kroiss, H. 2011 Energy requirement for waste water treatment. *Water Sci. Technol.* **64**, EM10347.

Tang, S. L., Yue, P. T. & Ku, C. C. 2007 *Engineering and Costs of Dual Water Supply Systems*. IWA Publishing, London.

Terpstra, P. M. J. 1999 Sustainable water usage systems: models for the sustainable utilization of domestic water in urban areas. *Water Sci. Technol.* **39** (5), 65–72.

Tilley, E., Atwater, J. & Mavinic, D. 2008 Recovery of struvite from stored human urine. *Envir. Tech.* **29** (7), 797–806.

US State of California, California Health Laws Related to Recycled Water: The purple book (2001; www.cdph.ca.gov/certlic/drinkingwater/Documents/Recharge/Purplebook-update6-01.PDF, Div. 4, Chap. 3.)

van Lieverloo, J. H. M., Mirjam Blokker, E. J. & Medema, G. 2007 Quantitative microbial risk assessment of distributed drinking water using faecal indicator incidence and concentrations. *J. Water Health* **5** (Suppl 1), S131–S149.

Vestner, R. J. & Gunthert, F. W. 2001 Upgrading of trickling filters for biological nutrient removal with an activated sludge stage. *GWF – Water·Wastewater* **142** (15), 39–46.

Wang, J., Lu, H., Chen, G. H., Lau, G. N., Tsang, W. L. & van Loosdrecht, M. C. M. 2009 A novel sulfate reduction, autotrophic denitrification, nitrification integrated (SANI) process for saline wastewater treatment. *Water Res.* **43** (9), 2363–2372.

Wanner, J., Kucman, K. & Grau, P. 1988 Activated sludge process with biofilm cultivation. *Water Res.* **22** (2), 207–215.

Wentzel, M. C., Ekama, G. A., Loewenthal, R. E., Dold, P. L. & Marais, G. v. R. 1989 Enhanced polyphosphate organism cultures in activated-sludge systems 0.2. Experimental behavior. *Water SA* **5** (2), 71–88.

Wilsenach, J. A., Germanis, J., Anderson, C., McMillan, M. & Mbaya, A. 2009 *Effects of Urine Separation and Treatment on Wastewater Effluent Quality*. Progress Report to the Water Research Commission, Project K5/1824, July 2009.

Wilsenach, J. A., Scurbiers, C. A. H. & van Loosdrecht, M. C. M. 2007 Phosphate and Potassium recovery from source separated urine through struvite precipitation. *Water Res.* **41**, 458–466.

Wilsenach, J. A., Maurer, M., Larsen, T. A. & van Loosdrecht, M. C. M. 2003 From waste treatment to integrated resource management. *Water Sci. Technol.* **48** (1), 1–9.

Wilsenach, J. A. & van Loosdrecht, M. C. M. 2004 Effects of separate urine collection on advanced nutrient removal processes. *Envir. Sci. Technol.* **38** (4), 1208–1215.

Yamaguchi, T., Yao, Y. & Kihara, Y. 2006 Biological sludge solubilization for reduction of excess sludge production in wastewater treatment process. *Water Sci. Technol.* **54** (5), 51–58.

First received 22 June 2010; accepted in revised form 19 October 2010

Downloaded from http://iwaponline.com/wst/article-pdf/64/6/1307/444586/1307.pdf
by guest