stowa

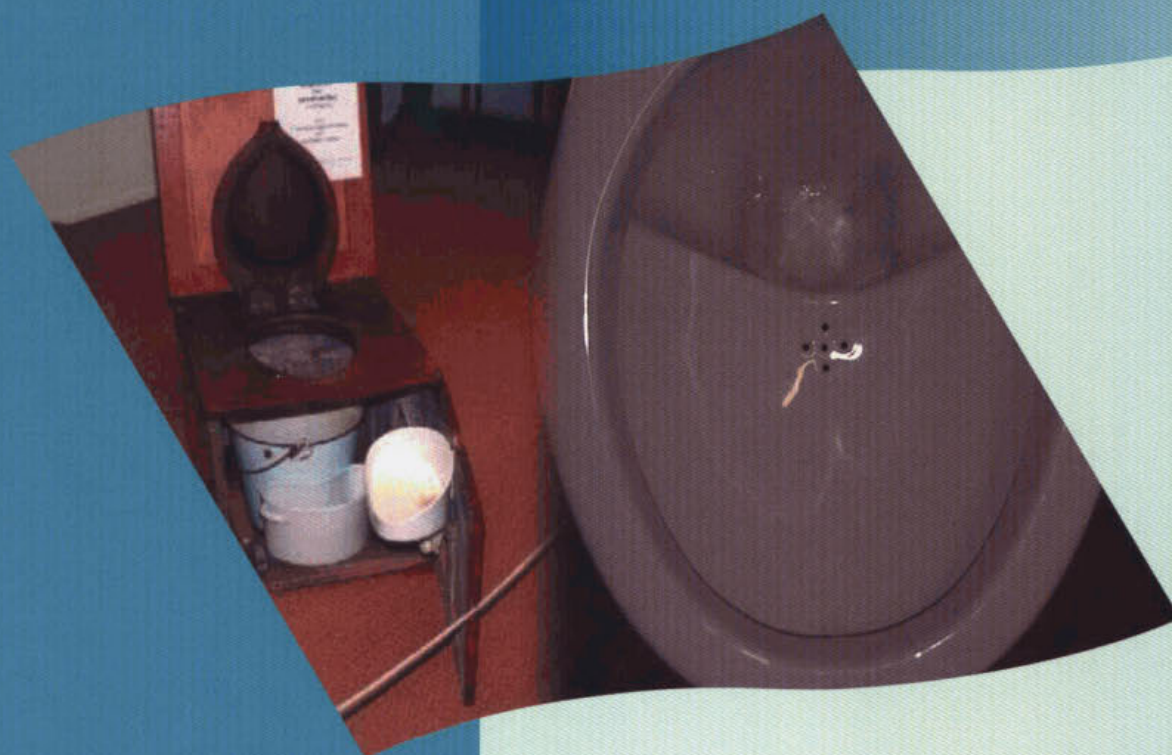Stichting Toegepast Onderzoek Waterbeheer

Separate urine collection and treatment

Options for sustainable wastewater systems and mineral recovery

2001 39

stowa

Separate urine collection and treatment

Options for sustainable wastewater systems and mineral recovery

2001 39

Ten Geleide

Uit literatuurgegevens blijkt dat urine de belangrijkste bron van mineralen in het afvalwater is. Urine bevat ongeveer 12 gN p.p.d. en 1 gP p.p.d. oftewel 80 % van de totale stikstofvracht en 45 % van de fosfaatvracht in afvalwater. Het volume van de geproduceerde urine is slechts ongeveer 0.3 % van de afvalwaterstroom. Het mineralen gehalte van urine is dus veel hoger dan in afvalwater, en een goede bron om nutriënten terug te winnen.
De afvalstoffen in urine zijn in feite waardevolle mineralen die door de grote verdunning met het 'transport'-water niet eenvoudig herbruikbaar zijn. Deze mineralen hebben een economische waarde, maar zijn ook deels afkomstig uit eindige natuurlijke bronnen en de productie ervan kost veel energie. Separate urine collectie en terugwinning van deze mineralen kan bijdragen aan een meer duurzame samenleving.

Dit rapport geeft een overzicht van de mogelijkheden en de (buitenlandse) ervaringen met betrekking tot het separaat inzamelen en terugwinnen van nutriënten uit urine. In het algemeen lijkt toepassing van separate urine inzameling voordelen te hebben met name omdat de huidige waterzuivering goedkoper en duurzamer kan worden opgezet. Terugwinning van fosfaat en eventueel ammonium is een extra bijkomstigheid in deze.

Utrecht, november 2002
De directeur van de STOWA
Ir. J.M.J. Leenen

Abstract

Drinking water is mainly used as a transport medium to remove waste products away from urban areas. This has a historical explanation since successful sanitation reduced the occurrence of diseases and improved human health. Some of these waste products are valuable minerals that are lost in existing treatment technology. Besides having an economic value, these minerals are finite natural resources and their production processes require energy. Reclaiming minerals may contribute towards making our systems and society more sustainable. This report evaluates the possibilities to reclaim minerals from wastewater.

A literature study revealed that urine is the main source of minerals in municipal wastewater. Urine contributes around 12 gN/p.d. (80%) of the nitrogen load in wastewater. Nitrogen in urine occurs as urea, which rapidly hydrolyses in wastewater to form ammonium. Urine also contributes around 1 gP/p.d. (45%) of the total phosphorus load in wastewater. The volume of urine produced is only between 1 and 1.5 l/p.d while the total municipal wastewater production is around 300 l/p.d (including faeces, urine, domestic effluent, some industrial effluent an some rainwater). The mineral concentration in urine is therefore much more concentrated than the concentrations in wastewater.

The study focussed on separate urine collection as a starting point for mineral recovery.

An inventory was made of various initiatives to collect urine separately. Some "ecological villages" in Sweden have been collecting urine separately for the past decade. Various studies are listed where these systems have been studied to solve technical problems and to compare the urine separation and treatment technology with the existing end-of-pipe technology. The urine from "eco-villages" is mostly used as fertiliser (high N and P content). Novaquatis (EAWAG, Switzerland) adopted a more pro-active approach and considers the complete cycle of urine separation technology. Their research covers diverse issues, such as human excretion, sanitation, urine collection, -transport and -treatment, wastewater treatment, surface water quality, agriculture and socio-political aspects and conceptions of urine technology. Some projects to collect urine separately in Germany and Austria are also listed.

Dilution of urine should be prevented with separate collection, as the hydrolysis rate of urea to ammonia is proportional to the amount of dilution. Ammonia evaporation could lead to a loss of recoverable minerals, but could also lead to problems with odours and corrosion. The report describes various commercial urine separation or "no-mix" toilets, produced and installed during the past decade. Waterless urinals were first introduced to save water, but it could be applied to collect urine undiluted. The technology seems to be developing in the right direction. Research and experience have shown that separate urine collection, handling and storage are relatively safe to communities or operators. Methods exist to determine microbial risks in urine separation systems and to monitor occurrence of faecal contamination.

Besides mineral recovery, possible effects of urine on aquatic eco-systems and human health were summarised, based on a literature study. During most rainstorms, nutrients and micro-pollutants, such as hormones and pharmaceutical end products, are discharged directly into surface water with combined sewer overflows. Micro-pollutants in municipal wastewater mainly originate in human urine. Furthermore, wastewater treatment works are not designed to remove micro-pollutants under normal operating conditions. Studies on aquatic ecosystems indicated the occurrence of endocrine disrupters (micro-pollutants) which might be responsible for some chronic illnesses and disorders in eco-systems and amongst people. Although models fail to quantify the emissions accurately, urine separation would most likely reduce the discharge of nutrients and micro-pollutants to surface waters.

The effects of separate urine collection on biological nutrient removal processes were evaluated with computer simulation (Aquasim). According to the model, modified UCT-type wastewater

treatment processes, such as the BCFS® process, would require between 20% and 30% less oxygen if 50% of all urine were collected separately. The total nitrogen effluent concentration also decreases with increasing urine separation efficiency. The effects of urine separation combined with primary sedimentation and pre-precipitation were also investigated. Total nitrogen effluent concentrations of 2.2 gN/m^3 could be possible in these processes with settled or pre-precipitated wastewater where more than 50% of all urine was collected separately. It was also shown that primary sedimentation doesn't reduce the denitrification capacity of advanced biological nutrient removal processes drastically, especially not when combined with urine separation. With urine separation, low effluent concentrations of nitrate and ammonia could be maintained while a large amount of the COD load could be removed, either with pre-settling or with pre-precipitation. The relative improvement in effluent quality could be even more significant in less advanced treatment processes. If all nutrients were consumed in cell metabolism, very small treatment works with low sludge ages (SRT = 1) could be possible. Simulation results suggest that with between 50% and 60% separate urine collection, such a process could still achieve $N_{tot\_eff} < 10$ gN/m^3. Energy consumption in wastewater treatment would be reduced where urine is collected separately; from 25% reduction at 50% urine separation to 50% reduction at 100% urine separation.

Based on a literature study, techniques to reclaim minerals from wastewater or urine were summarised. Calcium phosphate can be recovered from wastewater with complex and expensive processes. Recovered calcium phosphate is virtually indistinguishable from mined apatite, which is used in the production of commercial phosphorus or phosphoric acid. Struvite (magnesium ammonium/potassium phosphate) is a form of kidney stone. Struvite precipitation is a simple process in comparison to calcium phosphate precipitation. Controlled struvite precipitation from side streams could be economically beneficial when compared to more conventional chemical phosphate removal. Struvite can be recovered from urine by addition of magnesium. The price of apatite is only around € 40/ton, but minerals don't have to be recovered as a secondary raw material for industry. The value of struvite could be as high as 300 €/ton and could be used directly as fertiliser or in combination with other fertilisers. The production cost of struvite is equal to its value, but in the case of apatite the production cost is higher and the potential value lower than struvite.

Possibilities to recycle minerals in wastewater through agriculture or industry were evaluated. Raw materials, such as phosphate and potash, are finite resources. In order to meet the aims of sustainability, these minerals must eventually be recycled even if recycling costs are higher than current production costs. However, it is not obvious whether these minerals should be recovered from wastewater. The amount of phosphorus in Dutch wastewater is equal to the amount used in fertiliser (14 ktP/y), but much lower than the amount in animal manure (80 ktP/y). Still, if separate urine collection and treatment could lead to considerable benefits for conventional wastewater treatment, it would present a highly concentrated source of minerals that could be recovered.

Recovery of "valuable" minerals from aqueous waste streams should be preceded by careful economic and thermodynamic analysis. Minerals in waste streams are valuable because they originate from finite resources and require energy in the production processes. Mineral resources and energy have not reached critically low levels and exploitation of natural resources is therefore still economically attractive.

Ammonia can be recovered from solution by a number of techniques, such as ammonia stripping. Higher ammonia concentrations, liquid pH and temperature lead to more efficient stripping. Ion exchange reactions or ammonia adsorption could also be used to recover ammonia. However, nitrogen is not a finite resource. If nitrogen removal (including the energy required for industrial ammonia production) requires less energy than ammonia recovery, it could be more sustainable. Ammonia removal with the Sharon/Anammox process requires around 4 kJ/g N and costs less than 1.00 €/kg N removed. The production of ammonia requires

around 30 kJ/g N and costs less than 0.20 €/kg N. (Total for the cycle would be 30 - 40 kJ/gN and 1.00 €/kg N removed and then produced). On the other hand, ammonia recovery via stripping requires only around 24 kJ/g N, but costs around 8 €/kg N recovered.

In general, higher mineral concentrations reduce the energy demand and chemical dosing required for recovery or removal. Urine is a concentrated mineral solution, even when compared to some industrial waste streams and it should be possible to recover or remove minerals from urine in more cost-effective ways than with current technology.

The most important questions that remain are transport of separated urine and the most favourable treatment/recovery option based on exergy. In Switzerland, Novaquatis considers the option to drain urine via existing sewers late at night (when supposedly there is no wastewater). The benefits of urine separation probably wouldn't "pay" for truck transport. Local or de-central treatment of urine seems to be a solution. The treated liquid could drain through existing sewers and trucks could remove recovered minerals.

STOWA in brief

The Institute of Applied Water Research (in short, STOWA) is a research platform for Dutch water controllers. STOWA participants are ground and surface water managers in rural and urban areas, managers of domestic wastewater purification installations and dam inspectors. In 2002 that includes all the country's water boards, polder and dike districts and water treatment plants, the provinces and the State

These water controllers avail themselves of STOWA's facilities for the realisation of all kinds of applied technological, scientific, administrative-legal and social-scientific research activities that may be of communal importance. Research programmes are developed on the basis of requirement reports generated by the institute's participants. Research suggestions proposed by third parties such as centres of learning and consultancy bureaux, are more than welcome.

After having received such suggestions STOWA then consults its participants in order to verify the need for such proposed research.

STOWA does not conduct any research itself, instead it commissions specialised bodies to do the required research. All the studies are supervised by supervisory boards composed of staff from the various participating organisations and, where necessary, experts are brought in.

All the money required for research, development, information and other services is raised by the various participating parties. At the moment, this amounts to an annual budget of some five million euro.

For telephone contact STOWA's number is: (31 (0)30-2321199.
The postal address is: STOWA, P.O. Box 8090, 3503 RB, Utrecht.

E-mail: stowa@stowa.nl.
Website: www.stowa.nl.

Samenvatting

Drinkwater wordt hoofdzakelijk gebruikt om afvalproducten uit de stedelijke omgeving af te voeren. Dit is historisch zo gegroeid, en deze handelwijze is zeer succesvol in het voorkomen van ziektes en verbeteren van de gezondheidstoestand van de bevolking. Sommige van de afvalstoffen zijn in feite waardevolle mineralen die door de grote verdunning met het 'transport'-water niet eenvoudig herbruikbaar zijn. Deze mineralen hebben een economische waarde, maar zijn ook deels afkomstig uit eindige natuurlijke bronnen en de productie ervan kost veel energie. Terugwinning van deze mineralen kan bijdragen aan een meer duurzame samenleving. Dit rapport evalueert de mogelijkheden om deze mineralen terug te winnen uit afvalstromen.

Een literatuurstudie liet zien dat urine de belangrijkste bron van mineralen in het afvalwater is. Urine bevat ongeveer 12 gN p.p.d. en 1 gP p.p.d. oftewel 80 % van de totale stikstofvracht en 45 % van de fosfaatvracht in afvalwater. Het volume van de geproduceerde urine is slechts ongeveer 0.3 % van de afvalwaterstroom. Het mineralen gehalte van urine is dus veel hoger dan in afvalwater, en een goede bron om nutriënten terug te winnen. Dit onderzoek had als doel te inventariseren hoe separate urine collectie bij zou kunnen dragen aan het verduurzamen van de samenleving.

Allereerst is een overzicht gemaakt van verschillende initiatieven om tot urine separatie te komen. Sommige "ecological villages" in Zweden hebben al meer dan 10 jaar ervaring met separate urine collectie. Verschillende deelstudies hebben allerlei optredende problemen en ervaringen beschreven die ook gebruikt kunnen worden in een Nederlandse situatie. De urine van deze eco-dorpen wordt direct hergebruikt in de landbouw. Het project Novaquatis (EAWAG, Zwitserland) richt zich meer op de stedelijke omgeving en had een meer pro-actief karakter waarbij het integrale systeem onderzocht wordt: productie van urine, sanitatie, urine - collectie, -transport en- behandelingssystemen, afvalwaterbehandeling, oppervlaktewater- kwaliteit, landbouw toepassing en sociaal economische aspecten. Er zijn in Oostenrijk en Duitsland een paar kleinere projecten die separate urine behandeling onderzoeken.

Verdunning van urine tijdens inzameling moet worden vermeden aangezien dit de terugwinning moeilijker maakt. Ook een eventuele separate behandeling in een N-verwijderingsprocess is gebaat bij een geconcentreerde stroom. Momenteel zijn reeds diverse commerciële urine scheidings- of "no-mix" toiletten op de markt. Ook watervrije urinoirs zijn vanuit het oogpunt van waterbesparing reeds op diverse plaatsen geïnstalleerd. In principe is het sanitair benodigd voor separate opvang van urine dus al aanwezig. Onderzoek heeft ook aangetoond dat separate behandeling en collectie van urine geen extra gezondheidsrisico met zich meebrengt voor de gemeenschap en voor personeel dat met de urine omgaat. Methodieken om de microbiële veiligheid en contaminatie met fecaliën te detecteren zijn reeds voorhanden.

Afgezien van het terugwinnen van nutriënten kan separate inzameling van urine effecten hebben op aquatische ecosystemen en de menselijke gezondheid. Deze zijn op basis van de literatuur geëvalueerd. Door riooloverstorten komen nutriënten en micro-verontreinigingen zoals hormonen en pharmaceutische producten direct in het oppervlaktewater terecht. Veel van deze micro-verontreinigingen in huishoudelijk afvalwater komen oorspronkelijk uit de urine. Een andere bron van deze verontreiniging in het oppervlakte water is het effluent van de rioolwater- zuivering. Deze processen zijn normaal niet specifiek uitgelegd op het verwijderen van microverontreinigingen. Diverse studies in aquatische ecosystemen wijzen op een negatieve invloed van deze microverontreinigingen op het ecosysteem. Goede modellen voor het beschrijven van dergelijke vervuiling ontbreken nog, separate collectie van urine kan echter een goed aangrijppunt zijn om dergelijke verontreiniging te voorkomen.

Het effect van separate collectie van urine is in modelsimulaties geëvalueerd. Hiertoe is een goedwerkende biologische nutriëntenverwijderende afvalwaterzuivering op basis van het BCFS® geëvalueerd. Een gekalibreerd procesmodel voor de rwzi Harderberg was het uitgangspunt voor deze evaluatie. Deze evaluatie toonde dat 20-30 % minder beluchtingsenergie nodig als 50 % van de urine separaat wordt behandeld. Ook de totale stikstofvracht in het effluent daalt significant bij een toenemende urineseparatie (en een onveranderde procesvoering op de rwzi). Omdat er minder denitrificatie hoeft plaats te vinden kan het afvalwater ook na voorbezinking worden behandeld. Met voorbezonken afvalwater kan een N_{tot} van 2.2 gN/m^3 in het effluent worden bereikt indien 50 % van de urine separaat wordt ingezameld. Voorbezinking of pre-precipitatie maakt het mogelijk om een groot deel van het CZV om te zetten in methaangas, hetgeen bijdraagt aan de verduurzaming van de afvalwaterzuivering.

Separate urine collectie maakt echter ook geheel nieuwe zuiveringsconcepten mogelijk. De overblijvende nutriënten zouden volledig door slibgroei kunnen worden verwijderd in compacte hoog belaste zuiveringen. Bij 50-60 % separate urinecollectie kan in een rwzi bij een slibleeftijd van 1 dag een N-effluent van < 10 gN/m^3 worden bereikt. Een dergelijke zuivering heeft ook een veel lagere energiebehoefte dan een laag belast proces.

De bekende technieken om mineralen uit urine terug te winnen zijn op basis van een literatuurstudie vergeleken. Calciumfosfaat kan worden geproduceerd middels complexe en relatief dure processen. Voordeel is echter dat het calciumfosfaat vrijwel overeenkomt met apatiet dat nu als grondstof voor de fosforzuurindustrie dient. Struviet (magnesium ammonium/kalium fosfaat) is een belangrijke component van nierstenen. Struviet precipitatie is een simpel proces in vergelijking met de calciumfosfaat precipitatie. Gecontroleerde struviet precipitatie kan economisch van voordeel zijn ten opzichte van de conventionele fosfaatprecipitatie op rwzi's. Struviet kan ook eenvoudig worden gewonnen uit urine door dosering van magnesiumoxide. De prijs van apatiet is slechts 40 €/ton. De marktwaarde van struviet ligt rond 300 €/ton. Struviet kan direct worden toegepast voor bemesting. De productie kosten van apatiet liggen veel hoger dan de commerciele waarde, voor struviet zijn deze ongeveer gelijk. Dit maakt dat struviet de voor de handliggende vorm is voor terugwinning van fosfaat.

De mogelijkheid om mineralen uit afvalwater te hergebruiken in landbouw en industrie zijn verder in kaart gebracht. Ruwe materialen zoals fosfaaterts en potas zijn eindige grondstoffen. Met het licht op een duurzame ontwikkeling moeten deze materialen uiteindelijk worden teruggewonnen. De vraag is echter of afvalwater de eerst aangewezen bron van mineralen is. De totale hoeveelheid fosfaat in Nederlands afvalwater is vergelijkbaar met de hoeveelheid die in kunstmest wordt toegepast (14 ktP/jaar), maar veel lager dan de fosfaatvracht in dierlijke mest (80 ktP/jaar). Fosfaat kan eenvoudiger uit dierlijke mest worden teruggewonnen. Separate urinecollectie and behandeling moet dus meer voordelen hebben dan alleen het terugwinnen van mineralen.

Voor het terugwinnen van ammonium is een groot aantal technieken beschreven. Ammonium strippen lijkt echter in de praktijk de beste methode. Hogere ammonium gehaltes, pH en temperatuur zorgen voor verhoging van de strip efficiency. Terugwinning uit urine heeft dus duidelijk voordelen boven terugwinning uit afvalwater. Stikstof is echter geen eindige grondstof. Als stikstofverwijdering en ammoniumproductie uit stikstofgas samen minder energie kosten dan terugwinning ligt directe terugwinning niet voor de hand. Ammonia verwijdering met het Sharon/Anammox proces kost ongeveer 4 kJ/g N en 1.00 €/kg N verwijderd. De productie van ammonium kost 30 kJ/g N en minder dan 0.20 kg N. Daarnaast kost ammoniak herwinning via strippen 24 kJ/g N en ongeveer 8 €/kg N herwint. Het is niet duidelijk dat deze hogere prijs de besparing in energie rechtvaardigt.

In het algemeen lijkt toepassing van separate urine inzameling voordelen te hebben met name omdat de waterzuivering goedkoper en duurzamer kan worden opgezet. Terugwinning van fosfaat en eventueel ammonium is een extra bijkomstigheid in deze. Een voorzichtige eerste afweging laat zien dat de voordelen van urinescheiding aan de bron eenvoudig teniet kunnen worden gedaan als de urine per truck uit de stad moet worden afgevoerd. Ontwikkeling van een decentrale technologie om urine te behandelen of nutriënten terug te winnen is dan ook de meest gewenste richting voor de ontwikkelingen in de toekomst.

De STOWA in het kort

De Stichting Toegepast Onderzoek Waterbeheer, kortweg STOWA, is het onderzoeksplatform van Nederlandse waterbeheerders. Deelnemers zijn alle beheerders van grondwater en oppervlaktewater in landelijk en stedelijk gebied, beheerders van installaties voor de zuivering van huishoudelijk afvalwater en beheerders van waterkeringen. In 2002 waren dat alle waterschappen, hoogheemraadschappen en zuiveringsschappen, de provincies en het Rijk (i.c. het Rijksinstituut voor Zoetwaterbeheer en de Dienst Weg- en Waterbouw).

De waterbeheerders gebruiken de STOWA voor het realiseren van toegepast technisch, natuurwetenschappelijk, bestuurlijk juridisch en sociaal-wetenschappelijk onderzoek dat voor hen van gemeenschappelijk belang is. Onderzoeksprogramma's komen tot stand op basis van behoefte-inventarisaties bij de deelnemers. Onderzoekssuggesties van derden, zoals kennisinstituten en adviesbureaus, zijn van harte welkom. Deze suggesties toetst de STOWA aan de behoeften van de deelnemers.

De STOWA verricht zelf geen onderzoek, maar laat dit uitvoeren door gespecialiseerde instanties. De onderzoeken worden begeleid door begeleidingscommissies. Deze zijn samengesteld uit medewerkers van de deelnemers, zonodig aangevuld met andere deskundigen.

Het geld voor onderzoek, ontwikkeling, informatie en diensten brengen de deelnemers samen bijeen. Momenteel bedraagt het jaarlijkse budget zo'n vijf miljoen euro.

U kunt de STOWA bereiken op telefoonnummer: +31 (0)30-2321199.
Ons adres luidt: STOWA, Postbus 8090, 3503 RB Utrecht.
Email: stowa@stowa.nl.

Website: www.stowa.nl.

TABLE OF CONTENTS

Ten geleide
Abstract
Samenvatting

        Appendix 1 Mineral concentrations in human urine
        Appendix 2 Report on visiting the Novaquatis research group at EAWAG
        Appendix 3 Changing wastewater concentrations with increasing urine
        separation efficiency

1. INTRODUCTION

1.1 Water and waste

Society manages urban water to comply with three main requirements:

- Drinking water is supplied for household purposes
- Rainwater is drained to prevent flooding
- Wastewater is removed and treated to maintain urban hygiene.

We use most of our drinking water as a transport medium to remove waste products. Although such products (human excreta and wash water) are considered "waste", it actually contains many valuable minerals. In the pre-industrial European society, human and animal excreta formed part of a natural cycle.

Most agricultural enterprise employed mixed farming practices, incorporating crops and stocks. Excreta were returned to farmland, because many of the minerals contained therein are plant nutrients. Medieval land rent contracts specified that animal droppings had to decompose wherever it fell. The introduction of industrial fertiliser increased crop production per hectare and made intensive crop farming possible. Traditional (mixed) farming can not compete against intensive agriculture that separates stock and crop farming. The separation of animal manure from farmland changed its connotation from valuable fertiliser to excess waste product in today's bio-industry.

In cities and towns, night soil carriers removed most of the human excreta to farmland with buckets and wagons until the end of the 19th century. However, population explosions in cities lead to a serious sewage problem. Night soil carriers could not keep up with removing the increasing loads. Pit latrines were overloaded and leaked into groundwater, which was an important water source at the time. After outbreaks of diseases, such as cholera in the 1850's, John Snow proved that contaminated groundwater was responsible for the spreading of diseases in London (Snow, 1855 and Snow, 1856). The majority of people did not consider his work seriously while theories such as miasma[1] prevailed. Sewers were then constructed to "clean up cities" and to "prevent diseases from spreading through foul air". This measure was successful, because sewers protected drinking water sources (ground water) from pathogenic bacteria. However, the reason behind the success only became known around 1885, after the discovery of bacteria that cause diseases.

Sewer networks cleaned streets and protected groundwater, improving personal hygiene and health, but rivers became polluted. In 1857, also known as the year of the "great stink", the Thames in London stank so badly that the House of Parliament had to close for summer. In the Netherlands too, most methods of sanitation available then became inadequate and many alternative solutions were suggested at that time (Langeveld, 1999). Due to the magnitude of the problem, municipalities in the Netherlands became involved for the first time around 1900. This made the establishment of centrally controlled infrastructure possible, but also required a choice of technological system. As most of the other alternatives had serious logistical problems, water borne sewers were the best option around 1900.

End-of-pipe solutions were introduced in varying degrees of sophistication to solve these new problems. At first, only solids (containing mostly carbon) were removed by settling. Trickling filters (attached growth process) was introduced to aerate sewage and reduce oxygen demand. Later, this was improved by the aeration of bio-mass. Diluted nutrients (nitrogen and phosphorus) were still not removed and led to the growth of algae and other aqueous plants in

---

[1] Miasma: the formerly belief that poisonous or pungent vapours infect air and cause diseases, such as cholera.

surface waters. Excess amounts of nutrients in surface waters lead to eutrophication[2] and change aqueous habitats. Modern centralised wastewater treatment plants are effective in producing good quality effluents with low nutrient loads. Still, they have given rise to new problems, such as excess sludge handling. Problems of heavy metals and micro-pollutants (hormones, pharmaceuticals) remain mostly unsolved today.

Since changing the closed nutrient cycles of pre-industrial societies into one directional waste transport, pollution treatment solutions have given rise to new and always more complex problems. The debate on urban water management seem to indicate that we are, instead of solving problems, trapped in a cycle of problem replacement (Larsen and Gujer, 1997 and Otterpohl *et al*, 1997). It seems that once a certain technological system has been introduced successfully, it also determines and limits further development or improvements within that system. The evolution of large technological systems is discussed by Hughes (Hughes *et al*, 1992). His description of large technological systems also holds for the evolution of our urban water system.

The 300 litres of wastewater[3] that has to be treated per capita and day in the Netherlands today is a result of infrastructure that was first introduced a century ago. The slow evolution of this technological system now makes radical changes to existing infrastructure complex and expensive. From this point of view, technological development could remain trapped within the limits of existing systems. However, innovation should not be lamed by these restrictions.

1.2 Mineral transport in wastewater

The water used to transport "waste" away from urban areas actually removes valuable elements. These elements also represent an amount of energy consumed in their various production processes.

The three most important elements in wastewater are carbon, nitrogen and phosphorus:

- Carbon is present in nature as plant material and in the atmosphere as CO_2 gas. Concentrated carbon resources are found in finite fossilised sediments such as coal, oil or gas. The uses of reduced carbon compounds are numerous. It is burned on large scale to produce heat for electricity production, but it is also used for the production of synthetic materials, like plastics. Carbon in wastewater originates mainly from faeces, wash water and foods.
- Nitrogen is abundantly present in the earth's atmosphere. More than 75% of the atmosphere consists of N_2 gas. Nitrogen is a vital plant nutrient and is supplied to soil through natural process (e.g. nitrogen fixation or lightning), or through synthesised ammonium fertilisers. Around 80% of the nitrogen in wastewater is in the form of ammonium and forms when the urea in urine hydrolyses.
- Phosphorus is present in the earth's crust as various different phosphates, or in diluted form in the ocean. Concentrated finite resources are found in $Ca_3(PO_4)_2$ ores, and although exact quantities of phosphorus ore are unknown, the quantity of good quality "clean phosphorus" seems to be limited.[4] Phosphorus is another vital plant nutrient and is therefore mainly used in the production of industrial fertiliser. In addition, it has many industrial applications. Phosphorus in wastewater originates from urine (45%) as well as from faeces, foods, soaps and detergents (in some cases). In most European countries, phosphate in detergents has been prohibited for the past decade.

---

[2] Eutophication means, "life giving" and refers to increasing domination by aquatic weeds in water where the limiting nutrient, phosphorus, is present in unnaturally high concentrations. Die-off and settling of plant growth results in sediment oxygen demand (Tchobanoglous, 1991).

[3] This volume is an average that includes municipal sewerage, some industrial effluent and rainwater discharged from combined sewers (see Chapter 2).

[4] Most phosphorus sources (90%) have an average cadmium concentration of 165 mg Cd/kg P. The limit for cadmium in Swiss fertiliser is 50 mg Cd/kg P (Larsen et al, 1990). The limiting concentration in the European Union varies between countries (40 mg Cd/kg P in the Netherlands and 200 mg Cd/kg P in Germany).

Other elements found in wastewater include potassium, magnesium, sodium, chlorine and some heavy metals in low concentrations.

The concern for non-renewable resources is increasing with the realisation that the earth has a limited life support capacity. A consequence of the industrial revolution, is our use of finite natural resources. In pre-industrial times, fertilising minerals were recycled between animals and plants and the effects of solar energy (water and wind power) were used to do work. However, global economies today maintain a one-directional throughput of minerals and fossil energy. This practice presents waste treatment problems as well as the threat of over-exploiting raw materials.

Minerals are diluted between 100 and 200 times by transporting (removing) them through water and they consequently loose most of their value. The low concentrations at which the minerals are present, makes it energetically costly to reclaim them. High energy consumption adversely effects the viability of any reclaiming scheme. We should therefore consider and develop more sustainable practices for managing waste(water).

1.3 Sustainability and development

The definition of sustainability or sustainable development most often quoted, is still that of the Brundtland commission:

"Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs." (Brundtland, 1987)

The problematic assumptions and implications of this definition and other similar definitions were pointed out by Roger Levitt (Levitt, 1998). Apart from philosophical critique, this definition does not allow for quantification of sustainability. The following more tangible definition was introduced by a former German minister of Environmental affairs:

"In a sustainable society, resources are not exploited faster than they can regenerate themselves and emissions are not generated faster than they can be assimilated in a natural way." (Merkel, 1998)

Figure 1 Mineral cycles in society and natural environment
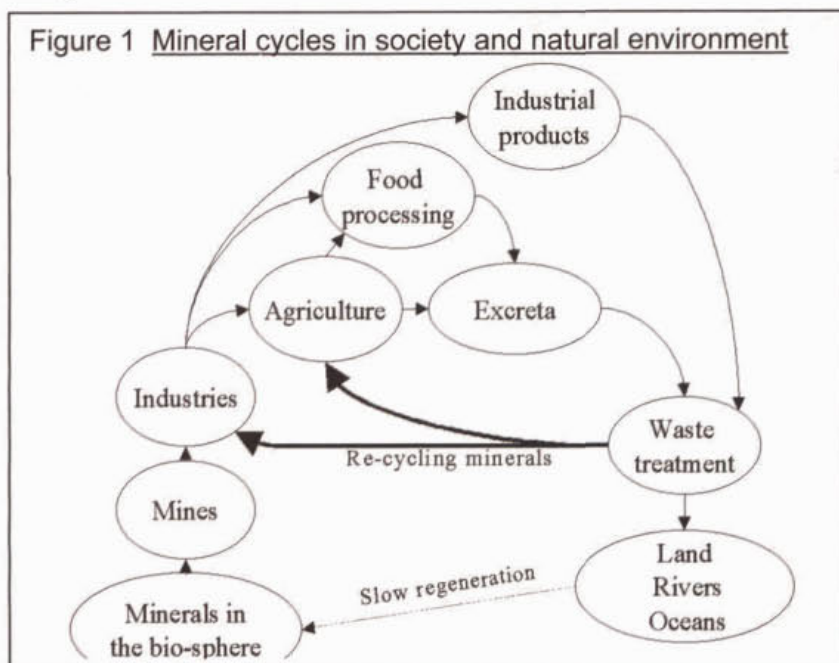
Figure 1 illustrates man made processes that exploit minerals faster than their natural regeneration rate (dotted line). In a sustainable society, new pathways (bold lines) must supplement natural processes. Recycled minerals can replace a portion of some of the raw materials from the bio-sphere. Recycling can also decrease harmful emissions to the natural environment. Reclaiming minerals from waste in general requires some natural resources.

A thermodynamic view of the world, concludes that our consumption of *energy* (more specifically, solar energy) is the ultimate indicator of sustainability.

Apart from this, we must realise that all materials at standard conditions contain an amount of energy, or the potential to perform work, called *exergy*. The sustainability of alternative technological systems can be evaluated on basis of their *exergy* consumption.

In this report, the focus is set on possibilities for reclaiming minerals from *municipal wastewater*. A few alternative technological systems may be possible to reclaim minerals from wastewater. The effect of urine separation on wastewater treatment is specifically evaluated.

1.4   Guidelines for more sustainable urban water systems in the Netherlands.

A few common sense guidelines can be used to evaluate new ideas for more sustainable technologies. Most of the guidelines below appeared in an article on sustainable urban water management (Larsen and Gujer, 1997). These should all apply to the various techniques or technological systems to reclaim minerals from wastewater.

- First, define the service that is required, instead of focussing on improvement of existing technology. Sustainability is not the same as optimisation of existing processes. Although the minimisation of energy and raw materials is desirable, the question here is "What is required of the technological system?" Systems or techniques are often taken as given. The history of sewers and wastewater treatment reflects this misconception. Sewer networks and centralised wastewater treatment is one possible solution to what is fundamentally a question of personal hygiene.
- Achieve totality of *"necessary services"* with minimum exergy use (look for local minima). Water is one of nature's forces that the Dutch have been fighting for centuries. Today the Netherlands still has too much water. Recycling of *water* is therefore not a first priority.
- It is likely that various techniques exist to reclaim minerals from wastewater. A choice between techniques should be based on the consumption of energy *and* materials by each technique to produce a certain result.
- Solve problems locally where possible. Do not export the problems of pollution, either into time or into space.
- Adopt a pro-active approach to water management instead of reactive.
- Make use of existing infrastructure before (or during) implementation of new technology. Do not replace systems working well before they reach their life span. This requires innovative thinking to incorporate historical facts with new developments.
- Dilution is no solution for pollution.

2. ORIGINS OF THE MINERAL LOAD IN MUNICIPAL WASTEWATER

2.1 The human body as source of minerals

2.1.1 Minerals contained in human urine

The kidneys regulate the concentration of constituents in the bodily fluids by three main processes:
- filtration of plasma;
- re-absorption of water and solutes from filtrate;
- secretion of selected solutes.

Urine is the final product of these processes (Strand, 1978 and Berne and Levy, 1998). Urine contains both inorganic and organic constituents. The volume of urine and the concentration of constituents vary according to a number of parameters. References encountered in this study illustrate these variations. Table 1 shows average values of the concentrations of constituents in urine that was compiled from a few values found in literature. Refer Appendix 1 for all values, ranges and references to literature.

Table 1   Summary of urine composition and mineral concentration

Urine parameter (per person)	Average values		
	g/p.d	g/m^3	mol/m^3
Volume	1250	1.020E+06	
PH		6	
Urea – $CO(NH_2)_2$	25.00	20000	333
NH_4 – N	0.75	600	43
N- total	12.00	9600	686
PO_4 – P	1.00	800	26
K	3.00	2400	62
Ca	0.20	160	4
Mg	0.20	160	7
Na	4.00	3200	139
Cl	7.00	5600	160
S	1.00	800	25
total COD	12.00	9600	300

Diet strongly influences the concentration of constituents in urine. One could for example expect variations in the average mineral concentrations of Dutch and American urine. One could also assume that the mineral concentrations in Dutch urine would now be closer to older American figures (Oosterom and Wulkan, 2000). Some dietary studies show higher mineral concentrations in Dutch urine (Govers, 1993, and Van Dokkum, 1984). However, no conclusive reference to specifically Dutch urine concentrations could be found.

The volume of urine production depends on sex and fluid intake. The normal minimum limit is 0.6 λ/p.d and the normal maximum limit 2.5 λ/p.d An average value of 1.25 λ/p.d is used in this report.

The pH of urine is slightly acidic, pH = 6 (Berne and Levy, 1998), due to end products of protein metabolism. On a vegetarian diet, urine can become alkaline. Extreme minimun and maximum values of urinary pH are 4.8 and 8.2 (Houssay, 1955).

Nitrogen in urine is mainly present as urea, an end product of protein metabolism. The concentration of urea, $CO(NH_2)_2$, in urine ranges between 20 g/p.d and 35 g/p.d. Urea is a soluble weakly basic compound. Some microbes are able to break down urea into ammonia and carbon dioxide. The formation of ammonia gives rise to the pH. Urea hydrolysis in urine takes 5 - 8 days to complete in closed systems (Hellström, 1998). Urea hydrolysis is described in more detail in Chapter 4 of this report.

Ammonia (NH_3) is very soluble in water, where it forms ammonium ions (NH_4^+). Ammonia is highly toxic and terrestrial animals have powerful enzymatic mechanisms for converting newly formed ammonia into more complex non-toxic compounds (e.g. urea), which are excreted. Total daily ammonia excretion is less than 1 g/p.d.

A small amount of uric acid is also present in urine (0.7 g/p.d).

The total load of nitrogen excreted through urine is assumed 12 gN/p.d.

Phosphorus in urine is directly related to phosphorus intake, since, on a mixed diet, there is likely to be a relative surplus of phosphorus. Phosphates occur in foods as a natural component of biological molecules and in the form of various food additives and preservatives.

The total phosphate (P_2O_5) excreted daily in urine is dependent on the phosphorus-containing substances ingested and on the elimination of calcium and magnesium in the faeces. When urine is alkaline, calcium and magnesium phosphates can easily crystallise.

A wide range of phosphorus concentrations in urine was found, related most probably to the typical diet of the people tested. The amount of phosphorus contained in cola drinks or beer is typically 150 - 250 mg P/λ. Milk has a typical phosphorus concentration of 900 mg P/λ. The average amount of phosphorus excreted through urine is assumed to be 1 gP/p.d. This may be considered as a conservative value that could be higher in the Netherlands (due to particulars of diet, which was not yet verified).

Potassium (K) in urine is present in relatively large concentrations. Values range between 2 g/p.d and 4 g/p.d. In this report, a value of 3 g/p.d is used.

Calcium (Ca) and Magnesium (Mg) in urine is present in small concentrations only. Values range between 0.1 g/p.d and 0.4 g/p.d. A value of 0.2 g/p.d is used for both minerals.

Sodium (Na) in urine is present in relatively large concentrations. Values range between 3 g/p.d and 5 g/p.d. In this report, a value of 4 g/p.d is used.

Chlorine (Cl) in urine is present in relatively large concentrations. Values range between 5 g/p.d and 9 g/p.d. In this report, a value of 7 g/p.d is used.

Sulphur (S) is also present in urine. Values range between 700 mg/d and 4 g/d. In this report, a value of 1 g/p.d is used.

Chemical oxygen demand (COD) results mostly from decomposed organic matter. Experiments with activated sludge showed, that about 85-90% of the COD in diluted and stored urine are biodegradable, the rest is inert. (Udert, 2001). The COD content in urine is around 12 g/p.d.

When urine is flushed into sewers, it mixes with water from various other household sources, known as "domestic wastewater", and rain (storm) water to form "municipal wastewater". If urine were collected and treated separately, it might be referred to as "yellow water". The different wastewater fractions and their mineral concentrations are described later in this chapter.

2.1.2 Minerals contained in human faeces

Unlike urine, that is a filtrate of nutrients, faeces contain mostly water and dead bacteria from the intestines. Digestion and absorption of foods is virtually complete by the time the foods reach the large intestine (Stacey and Santolucito, 1966). The faecal mineral content is therefor hardly related to dietary changes. Faecal matter is rather a product of the activity of the intestines than a residue of foodstuffs. For instance: there is a reduction of the mass of faeces produced during periods of complete fasting (or starvation), but the chemical composition is the same as during normal feeding (Houssay, 1955).

Human faecal matter consists of 67% water and 33% solids. The actual produced quantity depends on age, body mass and food consumption. An average value of the dry solids mass would be between 10-50 g/p.d (Keel and Neil 1965, Altman and Dittmer 1968, Davson and Segal, 1975). The solids consist of endogenous bacteria, lipids (lecithin, fatty acids and some neutral acids), some minerals and products of disintegration of bile pigments (Houssay, 1955) as well as undigested cellulose which passes unchanged with the faeces (Keel and Neil, 1965).

In general, the mineral load in faeces is much less than that of urine, with calcium and phosphorus being the only exceptions. Table 2 is a summary of the concentrations of constituents in faeces given by Altman and Dittmer (1968).

Table 2   Summary of faecal composition and mineral content

Faecal parameter (per person)	Average values g/d	mol/kg faeces
Water	95	
Solids	40	
N- total	1.40	0.7
P	0.69	0.2
K	0.47	0.1
Ca	0.53	0.1
Mg	0.18	0.1
Na	0.12	0.0
Cl	0.02	0.0
S	0.14	0.0
COD	41	9.5

Nitrogen in the intestines is absorbed and mainly transformed into amino acids. The urea cycle releases nitrogen from amino acid and the kidneys excrete urea with urine. Almost no nitrogen is present in faeces. Some nitrogen may be present in undigested food rests.

Phosphorus in faeces originates from various sources or bodily functions, such as phosphorus secretions, endogenous phosphorus and phosphorus contained in the bacteria. Between 35-45% of phosphorus is excreted via faeces, amounting to a total of 0.7 g P/p.d.

Potassium, Calcium, Magnesium and Sodium are present in small quantities, related to the content of these minerals in the undigested food rests.

Chemical Oxygen Demand: Faeces consist of mostly proteins, fats and carbohydrates. In a study by STOWA (Van der Wijst and Groot-Marcus, 1998), the COD of faeces per person per day was calculated as 41 g/p.d.

Faeces are commonly flushed with (drinking) water and mixed with domestic wastewater. If faeces and flush water were collected and treated separately, it might be referred to as "brown water" or "black water"[5].

2.2   Domestic grey water production

"Grey water" includes all wastewater from households with low COD and mineral concentrations. This water comes from showers, bathtubs, wash basins, kitchen water and automatic washing machines, which in effect means everything but toilet water. Grey water has a significant COD load, but very little minerals. Grey water production is between 125 and 135 λ/p.d on average. Figure 2 illustrates the volume and the COD load produced in different household sources, based on values by van der Wijst and Groot-Marcus (1998).

Figure 2 Domestic grey water production

---
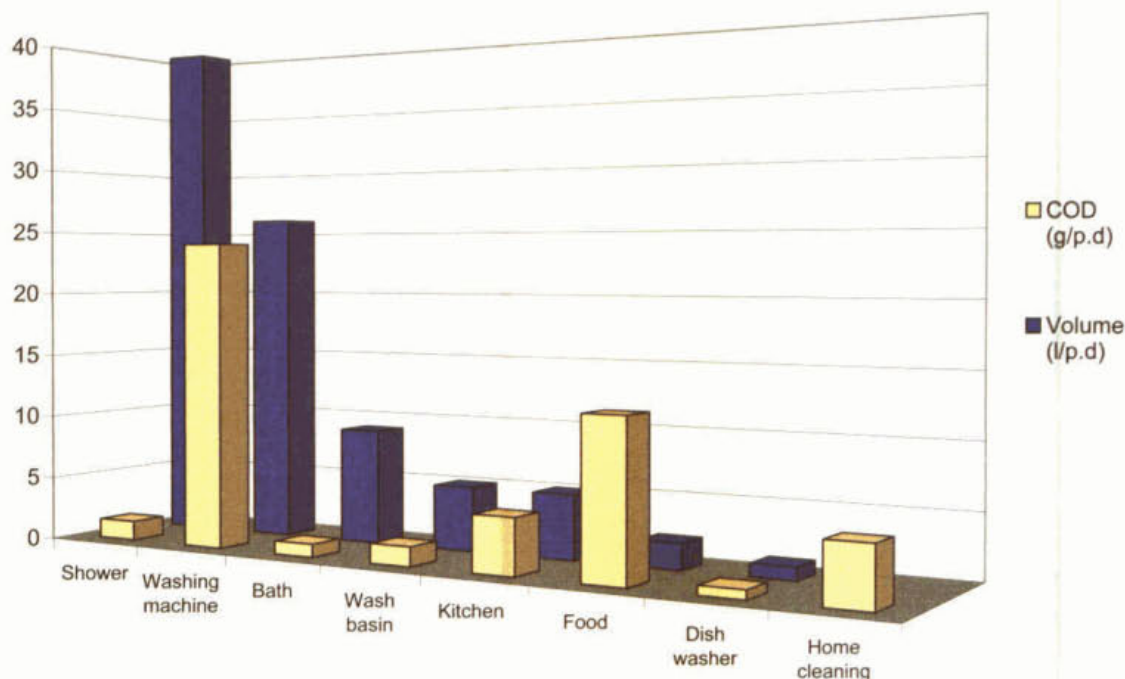
[5] "Black water" is normally used to mean water containing only faeces and urine. Where urine is collected separately, "brown water" may be used to refer to water containing only faeces.

Impurities in grey water originate in washing products (such as soap, shampoo, detergents and universal cleaners). Other substances include minute amounts of clothing residue, dirt, fats and food rests. Most of the COD in grey water originates from automatic washing machines.

Since 1990, the use of phosphates in detergents has been phased out in the Netherlands and some other countires in the European Union. Therefore, grey water should contain very little phophate, resulting from food rests only (e.g. milk, beer). The phosphate concentration in grey water has not yet been meausured (and most recent wastewater data is that of 1997, CBS 2000 [i.01]).

Very little nitrogen is present in grey water. Other minerals such as sodium and calcium are also present in grey water. The exact amounts vary considerably between regions, based on the concentration in drinking water. The incremental mineral load (Na, Ca, Mg, K) in grey water, above the drinking water concentration, is not significant. Tchobanoglous (1991) suggested the total additional mineral load of these metals above the original water quality when used as flush water or wash water.

2.3   Municipal wastewater production and composition

2.3.1 Rain water runoff

The total volume of water discharged to wastewater treatment works in the Netherlands is between 1500 and 2000 million m^3/a (CBS, 2000 [i.01]). This volume divided by the population gives an average value of 300 λ/p.d. Of this total daily volume, rainwater runoff is assumed to contribute in the order of 100 λ/p.d on average (Boer, 1995). Figure 3 presents rainfall and wastewater influent data and shows a direct relation between rainfall and wastewater discharge.

Figure 3   Wastewater discharge in relation to rainfall
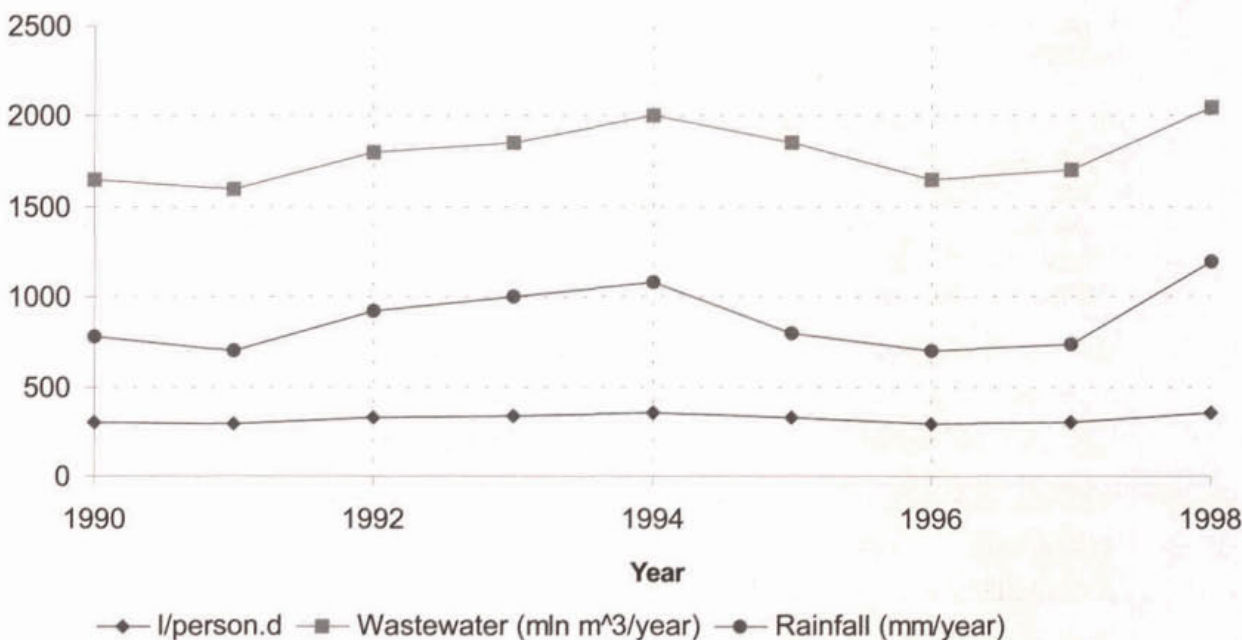
The increase in wastewater due to rainwater is mostly due to combined sewers. Apart from combined sewers, this fraction of water also includes an amount of infiltration.

The mineral and COD loads contributed by rainwater runoff are unknown and vary over time and from one place to the next. Substances such as oil and grease from vehicles, animal excreta, solid waste and litter all influence the COD and mineral load.

2.3.2 Industrial effluent

Small and medium sized industries discharge most of their wastewater to municipal treatment works. This typically includes water from offices, schools, restaurants, hospitals, supermarkets, laundromats, car wash bays, food industries, laboratories and other industries. The effluent composition from these various practices is unknown. The sum of industrial effluent and rainwater runoff is a "black box" that contains all the minerals and chemical oxygen demand that is not accounted for in domestic wastewater. The volume of industrial wastewater is inferred to be 70 λ/p.d[6].

2.3.3 Total municipal wastewater composition

Municipal wastewater is a concoction of domestic wastewater, rainwater runoff and some industrial effluent. The total volume of wastewater, divided by the total population gives an average volume of 300 λ/p.d. The concept of "population equivalents" (p.e.) is widely accepted. The volume of water per population equivalent is 150 λ/d during dry weather (STOWA 98-29). The wet weather flow is then three times this flow rate. However, this report has to differentiate between the different sources of the total mineral load contributed per person and day. Because population equivalents relate to assumed domestic loads only, it fails to put these loads in perspective.

Based on actual measurements, the CBS calculated the approximate total daily load of nutrients and chemical oxygen demand in the Netherlands. Table 3 presents the total daily load of nutrients and chemical oxygen demand (tonnes per day) in the Netherlands in 1997. The average municipal wastewater composition is the quotient of total mass load and water discharge.

The total phosphorus load is 2.4 gP/p.d, but only 1.7 gP/p.d is excreted by the body. The 0.7 gP/p.d *not excreted* by the body could not be traced to all its sources. As detergents are believed to have contained no phosphate, there could not be this much phosphate in grey water. It could be that industrial effluent contained relatively high amounts of phosphate, but this could not be verified. It could also mean that the amount of phosphorus excreted by Dutch is more than average. This could be due to the rather high consumption of diary products.

Table 3   Average load and concentration of the most important constituents in wastewater [10]

Wastewater parameters		ton/d	g/p.d	g/m^3
Chemical oxygen demand	(COD)	2508	161.1	537
Total nitrogen	(N_{tot})	233	15.0	50
Total phosphorus	(P_{tot})	37.2	2.4	8
Dutch population in 1997		15567100		

---

[6] 300 λ/p.d (total) – 130 λ/p.d (domestic) – 100 λ/p.d (rain) = 70 λ/p.d (industrial)

STOWA reported in 1998 on the influent quality of wastewater. In a study by Van Nieuwen-
huijzen (2000), wastewater from eight municipal wastewater treatment works was sampled and
analysed. Hao (2001) made use of wastewater characteristics in his modelling of the biological
nutrient removal process at Hardenberg. These values are all presented in table 4. The average
of these values is in the same order of magnitude as those in table 3.

Table 4   Comparison of wastewater influent data from different studies

Reference		(1)	(2)	(3)
Wastewater parameters		g/m^3	g/m^3	g/m^3
Chemical oxygen demand	(COD)	600	461	625
Total nitrogen	(N_{tot})	55	43.7	60
Total phosphorus	(P_{tot})	9	7.7	9.5

(1) STOWA rapport 98-29 (based on figures from CBS)
(2) Van Nieuwenhuijzen, (2000)
(3) Hao, (2000)

The average influent mineral load and chemical oxygen demand in wastewater presented in
Table 3, is used further in this research.

2.3.4 Heavy metals in municipal wastewater

Municipal wastewater contains heavy metals in concentrations that are generally above the
limits regarding re-use and human health. Table 5 presents average values for influent and
effluent concentrations. The activated sludge in wastewater treatment plants takes up large
quantities of heavy metals, thereby prohibiting the use of sludge as fertiliser. The figures
presented below are partly measured (both influent and effluent values) and partly based on
an assumed uptake of metals in activated sludge. (CBS, 2000[i.01])

Table 5   Heavy metal content of wastewater and sludge from different origins

Metal	Influent load (Holland)	Effluent load (Holland)	Metal load in biomass (Holland)	Metal conc. in dried matter (Average)	Metal conc. in dried matter (Lelystad)	Metal conc. in dried matter (Limit for re-use)
	ton/d	ton/d	ton/d	g/kg	g/kg	g/kg
As	6.9	3.2	3.7	11	<2	15
Cd	1	0.3	0.7	2	1.7	1.25
Cr	21	6	15	43	20	75
Cu	166	20	146	417	420	75
Hg	0.5	0.1	0.4	1	2	0.75
Ni	30	14	16	46	14	30
Pb	59	8	51	146	31	100
Zn	445	123	322	920	340	300

Most of the sewers in the Netherlands drain both domestic wastewater and rainfall runoff.
Newly built areas, such as Lelystad, have separate drainage networks for domestic wastewater

and rainwater runoff. From table 5 it is clear that concentrations of Chromium, Nickel, Lead and Zinc in waste activated sludge is much lower in Lelystad than in areas where combined wastewater is treated. Rainwater running over large areas of these metals dissolves them.

Zinc is a corrosion protection material on steel surfaces such as roofs, fences and lampposts. Lead in rainwater runoff originates from leaded fuel, lead in roofs and possibly leaded water supply pipes (Boer, 1995). It is also clear that Copper in wastewater is not related to rainfall runoff. Most of the drink water pipes inside houses are made of copper, of which a small amount dissolves in water.

The amount of metals in daily human excreta is insignificant in relation to the daily metal load absorbed in waste activated sludge. Metal loads in excreta range between zero and eight per mill of the amount in sludge.

2.4 Comparison of municipal wastewater, domestic wastewater and concentrated waste sources

The concentrations in tables 4 and 5 represent the wastewater characteristics when reaching the wastewater treatment works. This water is, as was shown in the previous paragraphs, composed of water from many different sources. The load and concentrations from these different sources are summarised in table 6. Urine is an important source of minerals (especially phosphorus, nitrogen and potassium) It also contributes some of the chemical oxygen demand. Washing machines add the largest chemical oxygen demand to grey water. The total mineral load in washing machine effluent is unknown, but is expected to be low, when compared with minerals from human excreta and the total load in wastewater.

Table 6   Mineral loads from various domestic waste sources compared to composition of municipal wastewater and seawater

Mineral	Urine		Faeces		Washing mach		Wastewater		Sea water [1.02]
(per person)	g/p.d	g/m^3	g/p.d	g/m^3	g/p.d	g/m^3	g/p.d	g/m^3	g/m^3
Volume-flush	0.5		15						
Volume-tot	1.75		15		26		300		
N	12	6857	1.4	93	0.1	4	15.0	50	1
P	1	571	0.7	46			2.4	8	0.5
K	3	1714	0.5	31			4.2	14	400
Ca	0.2	114	0.5	35			21.0	70	400
Mg	0.2	114	0.2	12			3.6	12	2900
Na	4	2286	0.1	8			25.5	85	10500
Cl	7	4000	0.0	1			15.0	50	17400
S	1	571	0.1	9			3.0	10	900
COD tot	12	6857	41.0	2733	24.0	923	161.1	537	

Urine has the highest mineral concentrations among the considered liquids. When compared to sea water, phosphorus in urine has a much higher concentration, which is required for recovery. Cations such as magnesium and sodium can be recovered, as salt, from seawater.

Different management of wastewater and separate collection of waste at the various sources will result in a completely different wastewater composition. Table 7 illustrates the effects of two alternative scenarios on wastewater composition. Source separation of urine or urine-and-faeces will in general reduce the mineral load in wastewater dramatically.

The load of phosphorus in wastewater could be reduced by between 40 - 70% with separate urine and urine-and-faeces removal respectively. The loads of potassium and nitrogen could be reduced by between 70 - 85% and 80 - 90% with separate urine and urine-and-faeces removal respectively. The second scenario (separate collection of urine-and-faeces) in effect represents the sum of all grey water, industrial water and rainwater runoff.

Table 7   Mineral loads from different scenarios of source separation, compared with municipal wastewater and drinking water

Mineral	Conventional wastewater composition		Wastewater: separate urine collection		Wastewater: separate urine& faeces collection		Drinking water quality norm
(per person)	g/p.d	g/m^3	g/p.d	g/m^3	g/p.d	g/m^3	g/m^3
Volume-flush	55		25		1		
Volume-tot	300		270		246		
N	15	50	3.0	11.1	1.6	7	0.26
P	2.4	8	1.4	5.2	0.7	3	2
K	4.2	14	1.2	4.4	0.7	3	12
Ca	21	70	20.8	77.0	20.3	82	150
Mg	3.6	12	3.4	12.6	3.2	13	50
Na	25.5	85	21.5	79.6	21.4	87	120
Cl	15	50	8.0	29.6	8.0	32	150
S	3	10	2.0	7.4	1.9	8	50
COD tot	161.1	537	149.1	552.2	108.1	439	0

5. Normen uit het Waterleidingbesluit, VROM, 2000/13

Human excreta contain almost no heavy metals (in relation to the load in wastewater) which make recovery of "clean" minerals from urine and faeces more attractive than from mixed municipal wastewater.

Source separation of urine and/or faeces will also reduce the salinity of wastewater effluent in general. The amount of potassium, for instance, in wastewater is above the limit for drinking water. The concentration of potassium in urine is even higher than that of seawater. The concentrations of chlorine and sulphate in urine are less than that seawater, but still quite high. If urine were collected separately at the source, the concentration of potassium in the remaining wastewater would drop to well below the drinking water norm. The concentration of other minerals would also decrease, but the concentrations of many minerals in conventional wastewater (such as Na, Ca, Mg and Cl) were already lower than the drinking water standard. Separate collection of urine and faeces could also reduce the concentration of chlorine in domestic wastewater by almost 40%. Source separation of urine and faeces could be attractive where recycling of *wastewater* is considered.

The three most important minerals in municipal wastewater are nitrogen, phosphorus and potassium. Considering different origins of minerals in wastewater, it becomes evident that these minerals originate mostly from human excreta, and in particular from urine. When both urine and faeces are collected separately, more than 80% of the minerals in wastewater could be reclaimed. Mineral concentrations in urine are 100 times higher than that of municipal wastewater. One feels intuitively that reclaiming minerals from such a concentrated waste product should be more viable than current practice whereby minerals are first diluted (conventional sewers) before they are removed in treatment processes.

This report will focus on the possibilities to reclaim minerals form urine and the effects that urine separation would have on advanced wastewater treatment processes.

3. PROJECTS AND INITIATIVES TO COLLECT AND TREAT URINE AND FAECES SEPARATELY

3.1 Introduction

Various pilot projects and research initiatives focus on alternative sanitation. The high concentration of nutrients in urine has led to special interest for separate collection of urine and faeces (refer Chapter 2 for average concentrations of nitrogen and phosphorus in urine, faeces and wastewater). The motivation for urine separation and the application of urine technology differ between countries. This chapter gives an overview of different pilot- and research projects and other initiatives in a few European countries and a few non-European countries.

3.2 Switzerland

3.2.1 EAWAG (Eidgenössische Anstalt für Wasserversorgung, Abwasserreinigung und Gewässerschutz)

The Swiss Federal Institute for Environmental Science and Technology (EAWAG) is located in Zürich. EAWAG's task as the national research centre for water pollution control is to ensure that:
- Concepts and technologies pertaining to the use of natural waters are continuously improved.
- Ecological, economical and social water interests are integrated.

3.2.2 Fundamental research on alternative sanitation technology

Within EAWAG, Novaquatis consists of a group of about ten researchers concerned with fundamental aspects of source separated sanitation. At the time of this report, no other group had such a dedicated and thorough investigation into the possibilities of source separation technology. The research within Novaquatis integrates different fields of research to gain knowledge of how to develop and implement new technology and avoid pitfalls.

The Novaquatis group was visited in September 2000, and a report of this visit is included in this document as Appendix 2. The main aspects of their research are highlighted below:
Laboratory scale experimental work is carried out to develop technology to treat urine and recover phosphorus (and other minerals).
Urine technology will be evaluated comprehensively by modelling of two scenarios (urban - Winterthur and rural - Basel) where the ultimate question is: Does urine separation technology constitute an overall improvement on current technology?
Novaquatis assumes that all separately collected urine could be used as fertiliser. Urine as fertiliser could be effective, but there are many conditions to meet. These conditions include questions regarding hormones and pharmaceutical end products returning to land (endocrine disrupters), transport, crop requirements and social acceptance. These are all areas of interest and further research within Novaquatis.
Wastewater regulations and wastewater treatment in Switzerland and the Netherlands differ (e.g., de-nitrification is not mandatory in Switzerland). However, these differences will become less in the future. Switzerland will have to reduce its total nitrogen emission to water by half within the next five years (Tagesanzeiger, 2001). Furthermore, due to the relatively steep slopes, Swiss wastewater treatment works receive a peak flow in mornings with a peak load of ammonia[7].

Novaquatis can be visited on web their page [i.03].

---

[7] During night time, the body builds protein (containing nitrogen), of which a great deal is absorbed by the kidneys and released with urine in the morning, giving rise to "ammonia peaks". In the Netherlands, wastewater takes much longer to reach the treatment works (due to absence of slopes). Peaks are dissipated, resulting in a more constant ammonia concentration.

3.2.3   Using existing infrastructure - discharging urine at night

One of the novel approaches of Novaquatis considers the use of existing sewers very early in the morning. During the early hours, there is virtually no wastewater discharge. If one could store urine in a tank (below the building) until this time, urine can flow via existing sewers. One could then collect urine by diverting it just before it reaches the treatment works, where it could be treated separately. One of the remaining problems is to determine the effect that ammonia evaporation would have, and how it can be prevented.

3.2.4 Pilot projects in Zürich and Basel

A new building project in Zürich includes a couple of residential units with No Mix toilets. *Kraftwerk 1* was designed as a modern village to house 350 people of whom 150 would work on the premises. Residents will move in during this summer. The toilets will keep urine separate, which will be stored in a tank. Farmers can then collect the urine to use as fertiliser. It could also be released at off-peak times to reduce shock loads of ammonia. *Kraftwerk1* can be visited on the internet [i.04].

A pilot project was started in Basel to examine the performance of the wastewater treatment works when ammonia-nitrogen (urine) is discharged to the treatment works intermittently instead of in single peaks.

3.2.5 Evaluation of public opinion and consumers' interest

Novaquatis evaluated consumer attitudes through discussion sessions with laymen and an on-line educational computer programme. The "NoMix Tool" is informative about all aspects of urine technology, alternative sanitation and closing nutrient cycles. Although only available in German, it is easy to understand and fun to explore! [i.05]. Novaquatis compiled a report after sessions with working groups to determine the consumer's opinion. The report should be available in July 2001.

3.3   Germany

Ralph Otterpohl lectures at the University of Hamburg Harrburg. He also works as consulting engineer for Otterwasser [i.06]. A paper by Otterpohl on the design of source control sanitation is available on the internet: Otterpohl, R., Design of highly efficient Source Control Sanitation and practical Experience, EURO-Summer School DESAR, Wageningen, June 18-23, 2000, The Netherlands [i.07]. This paper summarises his views on alternative sanitation and his experience on urine separation.

The paper introduces the concept of "Source control sanitation". Four lines of development are discussed as alternatives to conventional end-of-pipe technology. The traditional sanitation concepts lead to problems that have to be solved, instead of avoiding these problems altogether. According to Otterpohl, the conventional modern system of sewers with treatment plants at the end of the line are not only expensive, but fails to solve all pollution problems and is therefor not sustainable. New ideas for zero emissions include a shift away from "wastewater management" to "resource management". There is no *a priori* reason why (drinking) water should be used to remove (and dilute) domestic waste. If these "wastes" were considered resources, they could be removed and reclaimed by other means.

The four lines of development discussed in the paper are:

1. No mix toilets and gravity flow
   This concept is sustainable for single houses and rural settlements. Urine is collected separately (in a sorting toilet) and stored for agricultural use. Faeces are flushed to one of two chambers of a composting tank. An example of this approach was demonstrated in the sanitary design during restoration of a historic mill at Lambertsmühle [i.08].

2. Vacuum toilets and vacuum transport to a bio-gas plant
   This system has been implemented in a new settlement (Flintenbreite) in Lubëck, on the Baltic Sea in Germany. This settlement is not connected to the town's central wastewater network. Instead a bio-gas plant digests faeces anaerobically and uses the products for fertiliser (solids) and heating (bio-gas). Grey water is treated in a constructed wetland while rainwater is retained and infiltrated. The various concepts implemented at Flintenbreite can be found on the internet [i.09].

3. Low cost low maintenance on-site systems
   This idea is very similar to the first line of development. The most important difference is that excreta should not necessarily have to be removed with water. On-site treatment requires solar heating for composting and storage of urine.

4. Upgrade existing wastewater infrastructure
   If urine and faeces were collected separately, existing infrastructure could be enhanced and its life could be prolonged. Nutrient removal in enhanced biological nutrient removal plants would become obsolete when urine (and faeces) is collected separately. Otterpohl also refers to Larson's idea (Novaquatis) to use sewers to transport urine at night.

Jutta Niederste-Hollenberg does research on new sanitation concepts and modelling the mineral load at the origin.

3.4   Scandinavian initiatives

3.4.1   Ecological villages

A number of ecological villages were designed and constructed in Scandinavia in the previous decade. The term "ecological village" is widely used in public and in the media to describe a variety of different concepts. In this report, "ecological" villages or houses are of interest, because they nearly always incorporate some form of alternative sanitation, which often includes collecting urine separately. Ecological villages have a few other interesting features regarding re-use and recovery of different materials and energy, but it is not included in this report.

Bjorsbyn ecological village

Bjornsbyn was built in 1994, 5km from Lulea's city centre (Sweden). It consists 17 houses (57 people) with urine separation toilets. Urine is collected and stored in tanks for between six and eight months. Farmers then collect the urine to spray it on their grazing fields. Faeces flow into a septic tank, where sludge decomposes before being applied to grazing fields. During the study period, less than half of the expected nutrients were recovered. Problems such as leakage, infiltration, toilet design and overall system design decreased the success of this particular system. (Hanaeus, Hellstrom, Johansson, 1997)

Underststenshojden

Understenshojden that is built inside Stockholm has 44 apartments with 160 inhabitants. All urine is separated at source from where it flows to two large collection tanks. Separated urine samples were measured in a study (Jönsson, 1997) and the actual mineral loads were close to the expected loads (5g N/p.d and 0.4g P/p.d were collected and recovered on average). 1.34λ of urine solution was collected per inhabitant and day (of which 0.34λ was flush water).

Local organic farmers collect the stored urine and spray it on their fields. However, in the likely event of ammonia evaporation during application to land, a great deal of nitrogen would be lost. Furthermore, ammonia is toxic and contributes to acidification, which is not environmentally sensible.

Separation toilets saved about 50% of toilet flush water. Urea rapidly dissociated to ammonia and carbon dioxide and the pH increased to 9 (measured in storage tank). Contamination of separated urine by faecal material was very small. (Jönsson, 1997) and was considered insignificant. It was also found that around 50% of all toilet visits occur at home.

Munkesogaard

Munkesogaard was built in Roskilde (Denmark) and consists 20 houses with urine separation toilets. Urine flows to a central tank where local farmers collect it to spray on their lands.

3.4.2 Research initiatives

In contrast to the Novaquatis project, where there is a co-ordinated research agenda, a number of individuals (mostly Swedes) is concerned with aspects of urine separation technology.

Source separated urine - nutrient and heavy metal content, water saving and faecal contamination. Jönsson et al (1997)

Findings from this study were discussed under 3.4.1 Ecological villages; Understenshojden.

Modelling the sewage system - evaluating urine separation as a complementary function to the conventional sewage system: Jönsson et al, 1998

The study argues that separation of urine and its direct re-use as fertiliser is more energy efficient than the total of removing nutrients in treatment plants and production of industrial fertiliser. Results indicate that urine separation leads to decreased eutrophic effect[8]. It could be argued that the more advanced the wastewater treatment system is, the less advantageous the effects of urine separation might seem.

On a local scale, there should be an increase in energy (fuel) consumption by trucks removing urine. However, on a global scale, the total energy demand should be decreased. The energy saving associated with the reduction in fertiliser use should more than compensate for transport of urine.
Nutrient management in sewerage systems: investigations of components and exergy analysis, Hellström (1998) Lulea University of Technology.

This PhD thesis presents results from different experiments to improve nutrient management in sewerage systems. A urine separation system in a village in northern Sweden was investigated. About half of the nutrients contained in the urine was collected. An exergy analysis showed that urine separation technology might be a better alternative to conventional wastewater systems.

---

[8] The wastewater treatment plant is expected to remove 63% of the total nitrogen and 95% of phosphorus.

Exergy analysis is discussed in more detail in Chapter 4 of this report.

Analysis of wastewater systems, with respect to environmental impact and the use of resources, Karrman (1997) Chalmers University of Technology.

This thesis argues that urine separation systems are favourable because of comparatively low use of natural resources and low degree of eutrophication. An environmental impact assessment was conducted for two towns to compare hypothetically what the impact of planned installations would be. The study showed that the exergy during operation would be lower for a separation system than for the conventional alternative. This would have a smaller environmental impact than conventional sanitation.

"Environmental systems analysis" comprised using the ORWARE (ORganic WAste REsearch) software tool in combination with a life cycle analysis to evaluate four scenario's of wastewater and solid waste management:
- conventional sanitation and wastewater treatment;
- irrigation of energy forests;
- liquid composting;
- urine separation.

There is still a need to develop more energy efficient collection and treatment systems. Four proposed strategies for less environmental impact and less resource usage of wastewater management were developed during the study:
- handle nutrient rich flows separately from other waste flows;
- recycle nutrients and use energy efficiently;
- avoid contamination of wastewater flows;
- put unavoidable pollution to landfill.

Field experiment at Stockholm on the fertilising value of source separated human urine. Carlander, Hoglund, Vinnerås (1999)

This project has been running since 1996. Human urine is said to be a complete fertiliser with the composition of the macro-nutrients N:P:K being approximately 11:1:2.5. Different field experiments allow researchers to compare the fertilising effect of human urine to the effects of industrial fertiliser. (Sponsored by Stockholm Water Company and two housing companies).

In 1997, the yield of the crop fertilised with urine was 75% of the crop yield obtained by industrial fertiliser. In 1998, the results were comparable.

3.4.3 Pilot and laboratory scale projects

Universeum: Goteborg

A totally new concept was developed for handling the nutrients in human urine. Transforming the nutrients of the urine into solid materials reduces transport cost dramatically. On the addition of small amounts of MgO to human urine, most of the phosphorus (70-80%) and significant amounts of the potassium and nitrogen were precipitated. Crystalline struvite was the major component formed, together with similar compounds. Nitrogen recovery could be improved to 85% by adsorption on natural wollastonite or zeolite to 85% (Lind et al, 2000). Struvite is a well-known slow release fertiliser.

By closing eco-cycles several goals may be achieved:

- no risk for contaminating the farmland by cadmium from artificial fertilisers or from waste activated sludge;
- less demand for energy and finite resources (phosphorus) for the production of artificial fertilisers;
- less demand on the capabilities of the sewage treatment systems;
- less contribution to the eutrophication by the sewage treatment systems.

Experimental prototypes for struvite crystallisation will be developed based on the fundamental research conducted by Zsofia Ban and Bo Lind at Göteborg University during the last three years. The experimental prototype will be installed in a room specifically prepared for that purpose at the new Science Centre (*Universeum*) in Göteborg. The experimental prototype will be used for the testing of different strategies regarding freezing, dosage and mineral choice, mixing, drying and collecting of struvite, etc.

Based on the experiences from the experimental prototype, full-scale prototypes will be developed. The first one will be installed in a house with six flats with urine separating toilets. This pilot prototype will be in practical operation for the rest of the project time.

The mineral production (struvite) by the prototypes will be tested as fertiliser under different conditions both in the Science Centre (greenhouse plants) and in open field farmlands.

The question concerning micro-pollutants are not addressed in this study.

Nitrification and denitrification of urine in a column upflow macrophyte system
Farahbakhshazad Morrison

This research is aimed at developing treatment methods for household and agricultural effluent in developing countries.

Phragmatis australis (common reed) has the ability to take up large amounts of nutrients. Nitrogen is not lost to the atmosphere, as in nitrification/de-nitrification systems and therefore the N/P ratio in the agricultural runoff is retained. Vertical flow wetlands are proposed for cultivating the wetlands with reeds. Once reed stands show signs of senescence the whole bed can be emptied and composted. Land use for this full-scale systems is much less than conventional horizontal flow wetlands.

The first application of the vertical method in Göteborg is the Preem Refinery's pilot-scale wetland of 5m^2 that was constructed in May 1998. The first pilot-scale system for agricultural wastewater is the next objective of this project.

Studies of a vertical upflow macrophyte system show that luxury plant uptake with rhizome storage dominate nitrogen removal and provide conditions for nitrification.

An unplanted control system was supplied with the same influent. The efficiency of unplanted system for removal of ammonia and nitrate was poorer than the planted system. This difference between planted and unplanted systems indicates the role of phragmites in changing the pattern of oxygenation in the system. Better performance of planted system also shows a combination of microbial processes and plant uptake in this system.

3.4.4 Commercial enterprises

Water Revival Systems (WRS)

WRS work with the design and construction of alternative wastewater and storm water systems, specialising in urine separation systems [i.10].

VERNA, Ekologi och Miljkonsult

VERNA provides consulting services within the field of ecological wastewater treatment with recycling technologies and is competent in urine- and black-water separation [i.11].

3.5   Austria

"Solar City" is a new ecological housing project in the city of Linz. Urine diversion toilets will collect urine separately. Constructed wetlands would be used for treatment of "brown" and "grey" water. The project was preceded by a market study to evaluate various aspects of urine separation and decentralised wastewater treatment. Little information is (yet) available on the project.

3.6   Mexico

In Mexico 50% of the population has no form of sanitation. Of all municipal wastewater generated, only 13% is treated. Furthermore, due to improper pit latrines, ground water is polluted in many areas. This leads to the spreading of gastro-intestinal infections, which is the second most common cause of child mortality.

Conventional sanitation would require enormous quantities of water. There are no water resources available for large-scale development of water borne sanitation.

Dry sanitation was combined with source separated sanitation in Morelos, a state experiencing a population explosion. These toilets are a modification by Csar Anorve, based on the Vietnamese double vault toilet. The double-vault system consists of two chambers and a urine diverter. One chamber is used until two-thirds full, then sealed, and the other chamber is used until two-thirds full. The first chamber is then emptied of what has become clean, dry humus. This is done through an easily accessible door in the rear of the chamber [i.12].

The Mexican modification includes a toilet seat with funnel for urine diversion. The main advantages of the complete system are:
- aquifers are protected;
- grey water is released into local soils to recharge groundwater;
- separated excrement transformed into fertiliser and soil conditioner;
- toilets are relatively inexpensive (US $ 18.00).

3.7 South Africa

There are a few initiatives in the Republic of South Africa to develop sanitation technology as alternatives to water borne sewage systems. These developments are all concerned with supplying a sustainable form of sanitation to millions of rural people who has little or no access to sanitation facilities.

Sustainable sanitation in rural (poor) communities, must comply with two important criteria: low installation as well as maintenance cost;
environmental "friendliness", posing no threat to scarce ground- or surface water sources.

Urine diversion technology in South Africa

The basic level of sanitation in South Africa was defined as ventilated and improved pit latrines. Problems with these latrines are that they have to be emptied from time to time. Owners often don't have the financial means to do this. Geotechnical conditions also complicate the design and construction of pit latrines (e.g., rock, non-cohesive soil, water table).

"Dry box" toilets have been used in Vietnam, China, Mexico and El Salvador. Faeces and urine are collected separately allowing faecal matter to dehydrate and compost much quicker. Saw dust can be used as bulking agent and the "soil" must be removed from time to time. This concept was applied to build a number of urine diversion sanitation units in three rural communities near Umtata. Main research objectives were to test the social acceptability of the technology and to determine the potential for resource reuse. Microbial tests are performed to determine the rate of pathogen destruction.

The project includes community training and workshops. People were not prepared to collect and reuse urine (cultural taboo). Urine is therefor lead into soak pits. It would still be possible to convert the system to collect urine later.

When operated correctly the toilets have no odours or flies. People have already started using desiccated faeces in vegetable gardens and grain fields with positive results (Austin and Duncker, 1999)

3.8 North America, Australia, New Zealand

The United States of America is home to some complicated systems that separate urine and faeces after being mixed in the way conventional toilets do. These methods were not considered further. One of these can be seen on the internet [i.13].

Composting toilets are popular amongst a certain group of people in New Zealand, but do not seem to include mechanisms to collect urine separately.

No dedicated research actions could be encountered.

3.9 The Netherlands

A few examples of "ecological sanitation" can be found in the Netherlands. These are individuals adopting a certain lifestyle, and some include urine separation toilets. One example is briefly discussed below.

1 Schutterstraat, Delft

The house in 1 Schutterstraat was built during the middle of the 1990's by re-using old materials. Walls consist of wooden frame reinforcements filled with a straw and loam mixture. The house in Delft's city centre is not connected to the municipal wastewater network.

The toilet on the first floor allows faeces to drop down a shaft to ground level. Kitchen waste and toilet paper is also dropped into this shaft. A ventilation fan, operated with a time switch, increases the rate of composting and drying. The toilet collects urine separately, flushed with a small amount of water, which is then pumped with kitchen water and water from washbasins to a grease trap on the roof. Immediately downstream of the grease trap is a small and enclosed garden with pebbles and aquatic plants that died the previous winter. These plants take up an amount of nutrients in the yellow/grey water. The effluent from the aquatic garden infiltrates into a sloped roof garden. The water percolates through the soil and then runs down to ground level via gutters and through three small ponds. Water from the third little pond infiltrates into the soil. Table 8 indicates the water quality measured from a spot sample. The concentrations can not be compared to normal wastewater effluent, because in the volumes of water produced are totally different.

Table 8   Water quality measurement, 1 Schutterstraat, Delft

		Pond 1	Pond 2	Pond 3
Phosphorus	gP/m^3	18.2	4.7	2.3
Nitrogen total	gN/m^3	200	10.5	16
COD	gO_2/m^3	<0	<0	<0

4. SEPARATE COLLECTION AND HANDLING OF URINE

4.1 Toilets to collect urine and faeces separately

Collecting and treating human excreta in two separate fractions – urine and faeces – is not a new concept. Figure 4 shows a urine-diverting porcelain toilet from 1880. This toilet is in a museum now, in the city of Skara, Sweden.

Many of these toilets were built and installed to be operated with a bucket system. Human anatomy suggests that urine could be diverted to a bucket in front, while faeces are caught in a bucket at the back. From the point of view of sanitation and communal hygiene, it was a much better system than the failing pit latrines of that time. Nevertheless, it didn't have a significant impact on modern sanitation. Water borne sanitation, once installed in a city, was much more convenient and therefore an obvious choice by the end of the nineteenth century. By 1998, more than 2000 modern urine separation toilets have been installed in Sweden (Höglund *et al*, 1998) following a new line of thought, based on different criteria. Manufacturers of "separation" or "no-mix" toilets are now claiming that their products contribute to sustainability by enabling us to use human urine as fertiliser, thereby closing part of the nutrient cycles.

Figure 4 Museum piece

Since the first "no-mix" toilets were made, the success in collecting two distinctly separate and undiluted waste products has improved a lot. Successful separate urine collection depends on a number of different aspects, such as:

- efficiency and reliability of separation mechanism and minimal dilution of urine;
- robustness to blockages by solids (toilet paper, faeces, precipitates, etc);
- simplicity and durability of design (amount of moving parts, materials of construction, etc);
- social acceptance of changes to well-known and convenient forms of sanitation.

Public acceptance of new concepts in sanitation depends on its comparison with conventional toilets. It requires sophistication in its comfort, ease in handling and modern design. Probably the most crucial aspect in the acceptance of separation toilets is their technically fail-proof performance, while the user almost doesn't notice the differences. The following sections give some insight into the development of modern no-mix toilets and some pitfalls that should be avoided in planning and designing urine separation systems.

Wost Man Ecology[i.14] manufactures two distinctly different systems for toilets with urine separation. The single flush system only uses water to remove urine, while faeces are composted on site. The double flush system has two different flushing mechanisms: One for urine (using much less water) and one for faeces (conventional). The toilets are made out of porcelain.

In the single flush system, urine is flushed, using approximately 0.2 litres of water with each use. Solids fall into a bin housed in an insulated container. Good ventilation is required to prevent odours, but ventilation requires energy. Natural ventilation (chimney draft) could be sufficient, but where a fan is required, the energy in the total cycle could be more than that of conventional wastewater treatment. The solid waste dries and after a drying period of six months, the waste material can be burned, or buried in the ground where it decomposes further. A standard bin (holding 80 litres) will need to be emptied every three months for normal family use. A full bin weights between 10-12 kg, which makes handling difficult.

The separation toilet type DS (figure 5), was a modification of a regular toilet. A low wall was placed inside this toilet bowel to divert urine through three holes into a separate drainage system. Experience with this toilet learned that urine is diluted approximately five times (Udert, 2000). Furthermore, not all urine is separated, because some of it flows over the low wall. This toilet is flushed in the conventional way. An improvement on this toilet was the separation type ES (figure 6). The urine "catchment area" is much larger and elevated to a higher level where separation is more efficient. The system incorporates two flushing mechanisms. Urine is flushed with approximately 0.2 litre and drains down a 50 mm diameter pipe to a storage tank. Solids are flushed with 3-5 litres of water like any standard toilet. However, the urine draining system blocks from time to time. Toilet paper or other solids dropped in the front compartment (meant for urine only) is not flushed away. When some of these solids enter the small holes and pipes, the system blocks.

Figure 5 WM Ecology DS   *Figure 6 WM Ecology ES*   *Figure 7 Dubbletten*

Another Swedish product that is similar to the Wost Man Ecology separation toilet type ES, is the Dubbletten [i.15]. This toilet also comprises two well-separated bowels, each with its own independent flushing mechanism (figure 7). The Dubbletten require only 0.12 – 0.15 litre to flush urine in comparison to the 6 litres required for flushing of a conventional toilet. This difference implies a saving of almost 30 λ/p.d, based on the assumption that the average person uses the toilet around 5 times per day to urinate. The special nozzle for urine flush water is pointed out with an arrow in figure 7. The Dubbletten also differ from the separation toilet type ES in the urine openings. The front bowel is protected with a plug (circled in figure 7) to prevent paper and other solids from entering the orifices, while urine can flow freely into the drainpipe. Since the manufacture of these toilets started in 1994, they have been installed in schools, museums, offices, flat blocks and "eco-villages". These toilets have also been exported to other Scandinavian countries, as well as Switzerland, Italy, Spain and Australia.
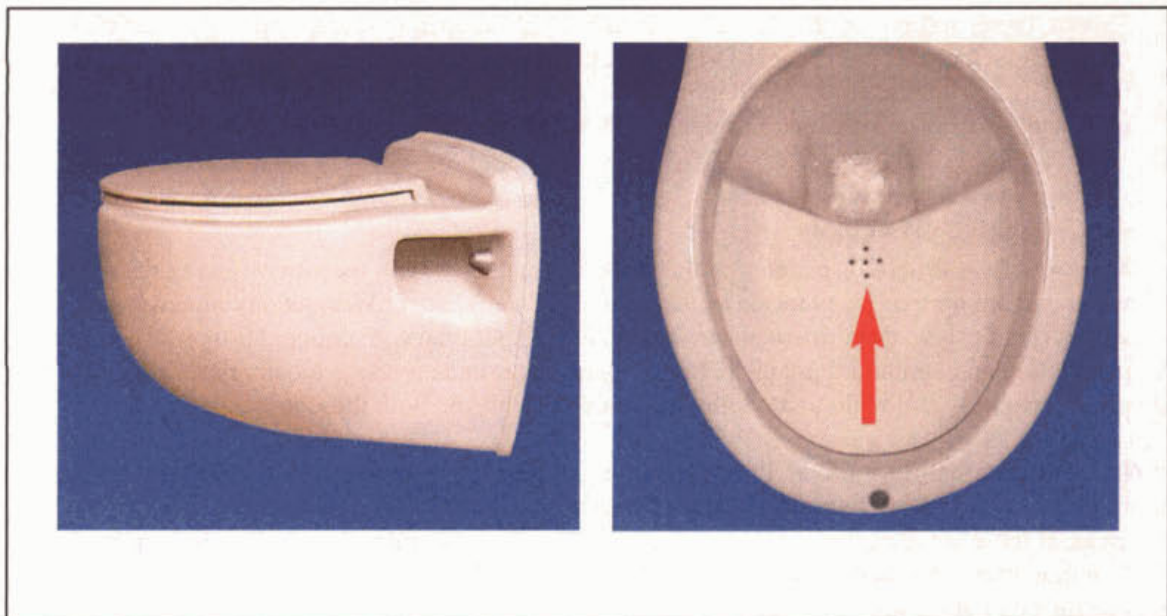
Figure 8 No-mix toilet

Step 1   Step 2   Step 3
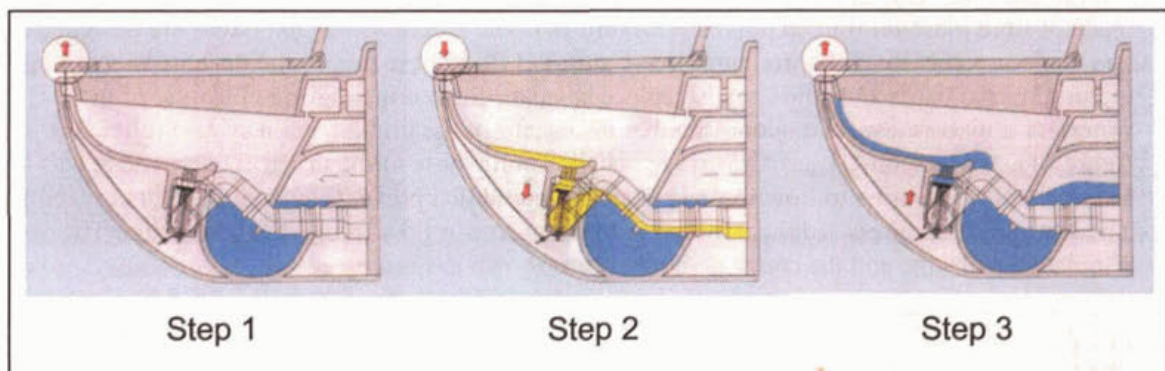
Figure 9 No-mix toilet flushing mechanism

The ROEDIGER [i.16] No-mix toilet gives the impression of being a conventional toilet and it can be used in the same way (figure 8). The only exception, as with all separation toilets, is that men have to sit during use. The toilet is similar to other separation toilets with two separate outlets. Urine is drained through five holes on an elevated platform (pointed out by an arrow in figure 8). A conventional outlet for faeces and paper is located in the back part of the bowl. The outlet for urine is special in its functioning, directed by a movable plug. This plug is normally closed (Step 1 of figure 9). While the toilet is in use, the toilet seat presses down onto a lever that pulls the plug away from the holes. Urine can then flow into these front inlets (Step 2 of figure 9). By standing up, the plug closes the holes again and the toilet can be flushed (Step 3 of figure 9). The two parts of the toilet bowl are unlike other separation toilets, not separated by a barrier (low wall), and flush water would wash away faeces and paper through the rear outlet. The system was designed to recover urine and water saving is not a prime concern, which could be a disadvantage. In contrast to the Dubbletten or separation toilet type ES, a single flush mechanism is used for urine and faecal flushing. However, due to the plug that closes during flushing, urine could be collected with almost no dilution that could benefit the recovery of minerals. This plug also prevents paper or faeces from entering and blocking the urine outlet.

Except for separation toilets, other mechanisms exist to separate faeces and urine. These mechanisms (such as the "Faecal-Urine Separation System" developed at Michigan State University [i.17]) are generally much more complex and much less elegant. It could possibly be used with success in live stock farming, but is not discussed here.

4.2 Waterless urinals

Most waterless urinals, especially in the USA, are marketed with the aim of saving drinking water and are normally connected to the normal sewage system. Without any pipe work and valves (for flush water) the urinals are normally cheaper than conventional urinals. The initial installation and continuing maintenance of waterless urinals is also a lot simpler. Collecting urine separately and without any dilution could be achieved with these urinals.

In a conventional flush urinal, the constant moist conditions provide a habitat for bacteria. Furthermore, flushing fixtures can spread bacterial aerosols, which then seed nearby moist areas. If these conditions are maintained, bacteria will proliferate, eventually resulting in stench. Manufacturers of waterless urinals claim to have special water repellent and disinfectant coatings painted inside the urinals. However, the material of manufacture seems to be of little importance. As long as the finish is very smooth, urine would not deposit and accumulate around or inside small surface cavities. Waterless urinals are therefore obtainable in glass fibre reinforced resins as well as in ordinary vitreous china. The surfaces of the WATERLESS [i.18] are made of fibre glass-reinforced polyester (figure 10). The Falcon-U1 urinal bowls are designed and made specifically with vitreous china (figure 11) Both these urinals are designed according to building standards and could easily replace standard flush urinals. A main feature of most waterless urinals is a special odour trap that forms part of the urinal's drain system (circled in figure 11). The Falcon-U1 has a cartridge installed at the base of the urinal. The cartridge acts as a funnel, allowing urine to flow through a sealant liquid that prevents any odours from escaping. The cartridge also filters sediment, before allowing urine to flow freely down the drain. The removable cartridge and the cartridge housing are shown in Figure 12.

Figure 10 Waterless urinal

Figure 11 Urinal with odour trap

Figure 12 Odour trap cartridge

The cartridge function is illustrated in Figure 13 [i.19]. Urine flows into a central hole, where it drops down into a small reservoir, initially filled with water. This reservoir acts as seal to prevent odours in the drain system from escaping into the bathroom or restroom. A liquid with a lower density than water floats on the surface of the reservoir. This liquid acts as seal to prevent the urine (eventually replacing the water in the small reservoir) from exposure to atmosphere. Urine gravitates through the liquid seal and pushes urine (already present in the reservoir) out to flow over a weir on one side of the cartridge. From there, the urine flows down a pipe. A horizontal baffle limits turbulence of the dropping urine and prevents mixing of the sealant liquid with the draining urine. The reservoir is also sufficiently large to ensure low flow

velocities. From time to time, the sealant liquid is replenished. Figure 14 shows a similar cartridge for the WATERLESS urinal [i.20] (shown in figure 10) where the drainpipe is in the centre of the cartridge.

Figure 13 Odour trap mechanism I
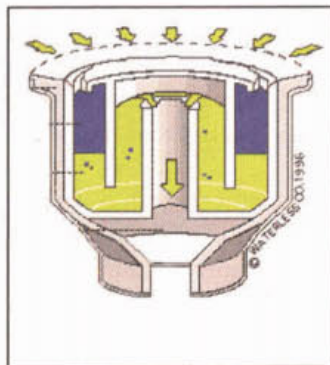
Figure 14 Odour trap mechanism II

Figure 15 Female urinal

Urinals for female use have already been mass-produced 50 years ago. One such example was the "Sanistand" [i.21], manufactured by American Standard from 1950 to 1973 when production stopped. The "Sanistand" had no real advantage over conventional toilets, as it used equal flush volumes and took up the same amount of floor space. Other attempts to introduce urinals to lady's rooms were even less successful, such as the "She-inal" [i.21]. Recently, Sphinx in the Netherlands started to produce urinals for ladies, the "Lady-P" (figure 15). This new design is mostly an attempt to reduce waiting time and long queues in places like night-clubs. This urinal was however not designed to use less flushing water than normal urinals. Where separate and undiluted urine collection is the priority, it is doubtful whether these urinals would be more effective than the separation toilets discussed above. However, it could be made to operate without water.

Men generally prefer to stand while urinating. Persuading men to sit down on the separation toilet (to urinate) has been problematic where the system was introduced to a wider public. Although this statement has not been investigated thoroughly (Hellström and Johansson, 1999), waterless urinals for men could definitely be used effectively to collect urine separate and undiluted.

4.3   Storage, handling and transport of separated urine

4.3.1   Hydrolysis of urea

In fresh urine, nitrogen is mostly present in the form of urea, $CO(NH_2)_2$. Many micro-organisms produce urease, an enzyme that catalyses the hydrolysis of urea (Alexander, 1977):

$$CO(NH_2)_2 + 3H_2O \rightarrow 2NH_4^+ + HCO_3^- + OH^-$$
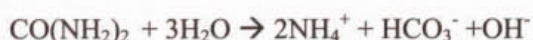
Ammonium in an aqueous solution is always in equilibrium with dissolved ammonia, depending on the pH:

$$NH_4^+ + OH^- \leftrightarrow NH_3(aq) + H_2O$$
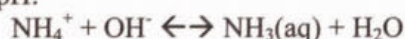
The dissolved ammonia in turn is in equilibrium with gaseous ammonia, depending on temperature and partial pressure above the liquid:

$$NH_3(aq) \leftrightarrow NH_3(g)$$

Urea hydrolysis will therefore increase the ammonia concentration and increase the pH. This will lead to ammonia evaporation if the temperature is high enough and where the solution is exposed to the atmosphere. Apart from the bad odour, ammonia lost to the atmosphere contributes to environmental problems (such as acid rain).

Experience with urine separation systems in stock farming has shown that ammonia losses could be minimised by keeping the storage temperature low, having a low pH value and avoiding aeration above the liquid surface in the storage tank (Rodhe and Johansson, 1996).

Low temperatures (below $10^{\circ}C$) reduce the growth of bacteria as well as the hydrolysis of urea (Wolgast, 1993). However, urea is still slowly hydrolysed at temperatures just above freezing point (Alexander, 1977). The effects of storage temperature (and other factors) on urea hydrolysis were also investigated by Hanæus et al (1996). The alkalinity of urine was used as an indicator of the concentration of hydroxide and ammonium ions. This indicates the extent of hydrolysis that took place in the solution. Urine samples that were stored at $5^{\circ}C$ (original pH = 6.1), underwent almost no hydrolysis in the first half of the experiment (pH = 6.2 after 60 days). Once hydrolysis started, the conversion rate was constant (pH = 7.8 at 120 days). Samples stored under similar conditions but at a temperature of $20^{\circ}C$ showed a fast conversion within the first 25 days (pH = 8.7 at 25 days) and then rose to a final pH = 9. Temperature also effects the equilibrium between dissolved and gaseous ammonia. Higher temperatures would shift the equilibrium towards the gaseous phase

When 4% wastewater was added to the fresh urine samples, there was a relatively fast conversion of urea, even at $T = 5^{\circ}C$. At higher temperatures ($T = 20 - 25^{\circ}C$) the conversion to ammonium was very rapid with addition of 4% wastewater (pH = 9 at 5 days). The increased hydrolysis rate is ascribed to the micro-organisms present in wastewater (Hanæus et al, 1996).

When urine was diluted with pure water, higher dilutions generally led to a higher ammonia conversion rate. Where urine was diluted, hydrolysis started earlier (after 25 days, compared to 60 for undiluted urine) and the final ammonium concentration was higher. Of the initial nitrogen in the urine sample 25% was converted to ammonium in diluted urine, while 15% was converted in undiluted urine after 120 days. Where 4% wastewater was added to diluted urine, the conversion of urea to ammonium was almost complete: After 25 days, 90% of the nitrogen was converted to ammonium (Hanæus et al 1996).

Further experiments showed that the addition of sulphuric acid delays the start of urea hydrolysis. At pH = 3, almost no conversion of urea to ammonium occurred within 120 days in samples with 4% wastewater added.

4.3.2 Storage and handling of urine

"Eco-villages" have experimented with urine transport and storage. In most cases, urine was piped to flow under gravity into storage tanks. From these underground storage tanks, urine was pumped into tanker trucks and taken to farmlands. The material most often used for pipelines and storage tanks was uPVC. In urine separation systems where copper pipes were used, problems were experienced with precipitation which led to blockages and loss of minerals (Hellström and Johansson, 1999).

Urine storage should be aimed at preventing urease and ammonia evaporation. Ammonia will be present in urine storage tanks where it could potentially evaporate. Ammonia evaporation must be prevented to:
- avoid odours from escaping;
- avoid adverse environmental effects caused by vaporised ammonia;
- reclaim as much as possible nitrogen in following processes.

Exposure to atmosphere should be minimised where possible, and urine storage tanks should be kept well closed.

Water in urine should be avoided as much as possible. This means that as little as possible water should be used in separation toilets and no water at all in urinals. Care should be taken to ensure that water doesn't leak into storage tanks. Where this has happened in pilot projects such as Björsbyn, 90% of the nitrogen in urine storage tanks was present as ammonia. Much of this ammonia was lost due to evaporation and in many cases less than half of the expected amounts were collected (Hanæus *et al*, 1997). In their review of Swedish urine separating systems, Hellström and Johansson (1999) mentioned ammonia concentrations of between 2500 and 4500 gN/m^3, measured in 15 different storage tanks from different projects. Fresh urine has a total nitrogen concentration of about 8000 gN/m^3. Apart form problems of hydrolysis and ammonia evaporation, higher volumes of water would obviously reduce the urine storage capacity and increase the transport costs.

The temperatures in the storage tank should be kept as low as possible. For this reason, most urine storage tanks are installed underground.

The issues of hygiene and faecal contamination in urine are discussed in Chapter 6 of this report.

4.3.3 Transport of urine

To some extent, the method by which separately collected urine would be transported depends on the choice of treatment or re-use technology. At least three transport methods are possible:

- Urine could be stored locally or de-centrally[9] in tanks for the short term, from where it is transported to pastures by trucks for use as fertiliser (applied in Swedish pilot projects, Hanæus *et al*, 1997). The benefits that may be gained by collecting urine separately could be lost again if it was transported by trucks. Trucks consume much more fossil fuels than sewer systems, which function by means of gravity flow in combination with electrical pumps. Truck transport of a certain liquid mass requires around 50 times more energy than transportation of the same liquid mass via sewer systems.[10] Except for the environmental costs (consuming finite fossil fuels and emissions of soot particles, CO, CO_2 and NO_x), the costs involved in operation and maintenance of trucks would in general be higher than sewers.

- Urine could be stored in small household tanks temporarily, from where it could flow down existing sewers at night, when there should normally be no other sewage (suggested at EAWAG, Larsen and Gujer, 1996). It is proposed that a pulse in the electricity distribution network could open a valve in the tanks, allowing stored urine to exit and flow into the sewer. The tanks furthest from the final collection point would be opened and emptied first. As the urine flows down the system, more and more tanks along the way are emptied to create a surge of urine flow. This surge of urine could then be diverted from a collection point in the sewer to a central treatment plant. The success of this approach depends mainly on two conditions:
The natural slope of sewers has to be steep enough to drain quickly, as most of the sewers in Switzerland are. In the Netherlands for instance, sewers are much less steep and almost never completely empty.

---

[9] In this case, "locally" refers to households, while "de-centrally" refers to a grouping of households, neighbourhoods, public places etc. For the rest of this report, "de-central" will also include "local"

[10] This figure is based on the assumptions that a 5-ton truck consumes fuel at a rate of 0,04 λ/km.ton, that the average trucking distance is 12 km and that the potential energy contained in one litre of diesel fuel is 34 MJ (CBS, 2001). The average total pump head for sewer systems in the Netherlands is assumed 20m (Evenblij, 2001) and the combined motor and pump efficiency is assumed 0,6.

The risk of rain events at night will have to be assessed continually. Most sewers in Switzerland are combined, draining municipal sewage as well as storm water. This leads to serious ammonia emissions after rainfalls, even from wastewater treatment works where the capacity is too low to treat these pulses. Where urine is transported separately in the combined sewers, a control mechanism would be needed to prevent urine transport during or immediately after rain events. Other problems that this approach must address are contamination of urine by faeces left in sewers and the volatilisation of ammonia in sewers. Additional benefits arising form *not transporting* urine with combined wastewater, such as reduced emissions during combined sewer overflows or normal emissions of endocrine disrupters[11], should also be investigated.

- Urine could flow directly from separation toilets to a local or de-central treatment facility. If one assumes that the nutrients present in urine could be removed on a local or de-central scale, the remaining liquid could flow down normal sewers with combined municipal wastewater. The dried matter (nutrients transformed into crystals or adsorbed into other materials) can then be removed either with normal household garbage, or with special trucks where mineral recovery is important. The combined weight of all minerals present in urine is around 5% of the weight of urine (refer Chapter 2 of this report). In a pilot project at Göteborg's Universeum [i.22], phosphorus is removed from separated urine by treating urine with magnesia to form struvite. Ammonia is removed by adsorbing it in natural zeolites. The granular solids produced are now being tested as fertiliser for different plants, while the remaining liquid (water) could either be used for irrigation, or drained via existing sewers.

There are of course more ways of transporting the urine, one of them being the construction of a new sewer network. This option would be very costly in the Netherlands, especially because the density of existing services and infrastructure render the construction of further infrastructure almost impossible. However, some form of sewer would be required to drain urine from toilets to a central point. Common problems experienced in eco-villages included precipitation of phosphate salts (e.g. struvite) and ammonia evaporation. It is therefore important to ensure sufficient gradients and to prevent ventilation.

It seems that the best technology for treatment of urine would also depend on the most viable method to transport either fresh urine, or the treated product.

4.4   On-site collection and treatment of urine

Sewers in the Netherlands are generally too flat to consider the Swiss approach, discussed above. Furthermore, it doesn't seem viable to construct dedicated sewer systems to transport separated urine over great distances through cities. Truck transport of liquid is more expensive ecologically and economically than sewers. The most feasible option (for the near future) seems to be treating urine on de-central level by removing the nutrients (nitrogen and phosphorus) and draining the treated liquid with the existing sewers. The reclaimed minerals can then be transported as dried matter by trucks. If urine would be collected separately in the Netherlands, one could already target two areas of priority:

- Public places attract a large part of the community. Public toilets are used by many people on a daily basis. In general, these toilets are used and flushed more frequently than household toilets and have the potential to collect urine from many people coming from different places in one small area. Examples of such places include airports, schools, universities, sport stadiums, office blocks, shopping centres or even apartment buildings. The advantages of steering new projects in this direction would be twofold:
Many public places have a constant flux of people that guarantees a near-constant flow of urine. This would imply a continuous operation of any de-central treatment plant as well as large

---

[11] Refer Chapter 6 of this report for a short overview of combined sewer overflows, endocrine disrupters and other micro-pollutants.

quantities of minerals that can potentially be reclaimed. In such a case, the economy of scales would have the benefit over small installations. However, some other places like sport stadiums, can be described as having short periods of peak-attendance and long periods of no-attendance. Where urine from these places is discharged into sewers in concentrated surges, conventional wastewater treatment plants could be overloaded from time to time. This will lead to additional emissions.

Most large buildings have a maintenance team or a janitor and cleaning personnel. The operation and maintenance of de-central urine treatment could easily form part of the job description of such a team or personnel. In contrast, on a household level people would have to take personal responsibility for new technology. This would either lead to some opposition or just simply to negligence.

The main disadvantage of focussing on public places is also related to the great flux of people. People who are not educated on the purpose and functioning of separation toilets are bound to use it incorrectly or even misuse it. The survey being done by Novaquatis [i.05] regarding laymen's opinions and objections to no-mix toilets could give valuable insight.

- Open air mass gatherings in the Netherlands occur mostly during spring or summer festivals. For the duration of these gatherings, temporary mobile toilets and urinals are installed. In 2000, Nijmegen attracted more than 1.2 million visitors during the five-day summer festivals that coincide with the four-day march. According to the town council, all temporary toilets and urinals are connected directly to the municipal sewers with flexible hoses. Nijmegen has a population of around 150000, which means that the wastewater treatment plant would not be able to deal with the load in this period. If the urine could be collected separately, the load on the treatment plant would be more normal and urine could be treated to reclaim minerals. In this case, the use of trucks to transport urine might be justified. Other open-air festivals where many people are gathered in a small area are music festivals and rock shows. The Dynamo and Pink Pop festivals were limited to a maximum of 60 000 people and these shows continue for 3 days, with many of the attendants camping. The question would be whether people at a rock show or in a general festive mood are concerned about recycling minerals and the function of "strange toilets". Waterless urinals might be the perfect answer.

Hellström and Kärrman (1999) studied the variation of nitrogen and phosphorus in fresh urine. Out of a group of ten test persons, it was measured that 75% of the urinary phosphorus excretion and 67% of the urinary nitrogen excretion can be found in household urine. This would make residential areas the most viable starting place to collect urine separately. The same study showed that the time-variation in urinary nutrient concentrations was almost insignificant. Only nitrogen has a higher concentration in the morning (8500 gN/m^3 in the morning and 7250 gN/m^3 during the day). However, in studying the urine separation system of an "eco-village" in Stockholm, the predicted amount of nutrients was exactly 50% of the total daily excretion (Jönsson et al, 1997). The measured nutrients were between 15% and 20% lower than the predicted amounts, due to small inefficiencies. From these figures (based on 160 persons), it would seem that public places and office buildings could be equally suitable to start collecting source separated urine.

Variations in daily urinary excretion could also vary between countries and cultures. Should a pilot project be planned in the Netherlands, amounts of nutrients should also be measured.

4.5   Faecal contamination in urine separation systems

Although separate collection and treatment of urine could significantly benefit the aquatic environment, the primary aim of sanitation remains prevention of diseases carried by bacteria present in human excreta, especially faeces. The health risks associated with separate collection (and re-cycling) of urine should be weighed against its possible environmental benefits.

The possibility of faecal matter entering the urine bowl of source separating toilets is assumed to be small, but it can not be neglected. Faecal contamination was first studied by Jönsson *et al* (1997) who analysed source separated urine for the presence of coprostanol, a faecal sterol produced when cholesterol is degraded (Nichols and Leeming, 1991). Contamination with faecal material was found to be low.

The same group did more detailed research on faecal contamination and microbial die-off in urine separating systems (Höglund *et al*, 1998). The main objectives of their investigation were to:

- Quantify the presence of different groups of faecal indicator organisms from 11 different urine separating systems in Sweden.
- Estimate the die-off rates of these organisms and other pathogens with regard to temperature, pH and concentration.
- Determine whether the found organisms emanated from the faecal material or were indigenous to the system.
- Quantify the load of faecal material in source separated urine based on measurements of faecal sterols.

Escherichia coli (E.coli) were barely present in urine collection tanks, whereas faecal streptococci occurred in large amounts. It was suggested that growth within the system occurred. Survival experiments indicated that higher pH in general reduced the number of organisms: At pH = 6 most of the organisms had a better survival than at pH = 9. Alkaline conditions accelerate the conversion of urea to ammonia or ammonium, which is toxic to most organisms. The concentrations of Salmonellaphages were constant with no reduction after 50 days, except at pH = 4.5, where the die-off rate was higher. Lower temperature and urine dilution lead to longer survival of most organisms.

Sundin *et al* (1999) further studied the degradation of faecal sterols in urine. Over periods of 118 days, coprostanol showed no significant degradation in experiments with either urine or urine tank sediments. This levels out variability between spot samples and makes risk assessments sensible as it can now be based on time integrated loads. Coprostanol is therefor a much better indicator of faecal contamination than the traditionally excepted analyses of indicator organisms.

Although contamination by faecal matter is not considered as a serious problem in separate urine collection, certain health risks can not be excluded. Experiments with salmonella species demonstrated slow or no die-off rates in urine collection tanks in neutral or alkaline conditions. No-mix Toilets, although efficient, do not exclude faecal contamination of urine. Faecal contamination is usually detected by the presence of E.coli. This method is not effective in urine (E.coli die-off) while the presence of faecal sterols give a better indication.

Transport, storage and treatment of urine include some health risks. Faecal contamination or micro-organisms indigenous to sanitary systems may occur in source separated urine. No reference to a microbial risk assessment done for urine collection and storage tanks could be found.

4.6   Conclusions

- Commercial urine separation or "no-mix" toilets were produced and installed successfully in the past decade. Waterless urinals were first introduced to save water, but it could be applied to collect urine undiluted. The technology seems to be developing in the right direction.

- Dilution of urine should be prevented with separate collection, as the hydrolysis rate of urea to ammonia is proportional to the amount of dilution. Ammonia evaporation could lead to a loss of recoverable minerals, but could also lead to problems with odours and corrosion. However, although the rate of urea hydrolysis can be slowed down it will eventually occur. Acidification (to prevent ammonia evaporation) is unwanted, but partial nitrification would also lower the pH and prevent ammonia evaporation (Udert, 2001).

- Questions regarding urine transport have not yet been answered. In Switzerland, Novaquatis considers the option to drain urine via existing sewers late at night (when supposedly there is no wastewater). The benefits of urine separation probably wouldn't "pay" for truck transport. Local or de-central treatment of urine seems to be a solution. The treated liquid could drain through sewers and the recovered minerals could be removed by trucks.

- Research and experience to date have shown that separate urine collection, handling and storage are relatively safe to communities or operators. Methods exist to determine the microbial risks of urine separation systems and to monitor occurrence of faecal contamination.

5. IMPACTS OF SEPARATE URINE COLLECTION ON THE AQUATIC ENVIRONMENT

5.1 Introduction

Lower nutrient loads in wastewater effluent would contribute to prevent eutrophication and to improve surface water quality in general. However, the total load of nutrients routed through sewers does not always reach the wastewater treatment works. In the Netherlands, like most European countries, overflows from combined sewers add to the effect of eutrophication.

Except for nitrogen and phosphorus, many other substances, such as micro-pollutants, could have a negative impact on the aquatic environment. Although not specifically designed to do so, some of these substances might be partly removed in wastewater treatment works. A substantial part of the micro-pollutants, many of which are excreted via urine, can be found in the effluent.

Flush toilets and water borne sewage improved public health in cities immensely. Sanitation has become such an integrated part of modern city life that very few people are conscious of its great importance. Diseases such as cholera don't occur in cities with good sanitary services. However, one should also consider whether or not the long-term consequences of existing technology are harmful. If the urban water cycle requires changes, the new alternatives should not pose new health risks. One such an alternative could be separate urine collection.

5.2 Micro-pollutants and endocrine disrupters

5.2.1 Micro-pollutants and endocrine disrupters in the aquatic environment

Chapter 2 of this report highlighted urine as the body's main route for excretion of metabolic waste products and excess minerals. End products of pharmaceuticals and natural hormones found in wastewater are also believed to be largely excreted with urine (Larsen *et al*, 2001). Researchers differ on the exact concentrations of these substances in wastewater, but it may typically be in the order of micrograms or, in the case of estrogens, nanograms per litre (10^{-9} g/λ). Changes in the hormonal balance of animals may occur at very low concentrations of endocrine disrupters, which might cause chronic effects after sustained exposure. Many hormones and end products of medicines and drugs are not removed in wastewater treatment processes. Significant amounts of these compounds reach rivers and seas, polluting the environment and even drinking water, where surface water is used as source (Beer, 1997). It is still mostly unknown how these compounds interact with organisms or ecosystems in the environment, and what the non-toxic concentrations are. Research is being carried out with the aim to quantify the presence and effects of these substances.

5.2.2 Occurrence of natural and synthetic hormones in wastewater and surface water

The endocrine system in animals is a collection of glands that secretes special chemicals, known as hormones. Hormones integrate the functions of individual organs and the nervous-reproductive- and immune systems. Hormones exert their effects through binding to specific receptors, which then initiate a chain of biochemical events. Endocrine disrupters are agents that interfere with hormone functions such as altering feedback loops involving the brain, pituitary, gonads, thyroid gland, or other organs. Many of the substances believed to be related to endocrine disruption, have been detected, measured and studied in waste- and surface water.

The female hormone oestrogen, and an extended family of related compounds, are accumulating in effluents from sewage treatment works (Berg and Kuss, 1991). At the same time feminisation of male fish and hermaphrodite fish are attributed to the presence of endocrine disrupters

(Purdom *et al*, 1994; Sumpter and Jobling, 1995). One of the prime suspects is the oral contraceptive pill (Sahlberg *et al* 1987). Oral contraceptives contain ethinyl oestradiol, a synthetic compound capable of overriding the natural hormones that control women's reproductive cycles. This excess oestrogen, excreted along with the natural hormone in women's urine, finds its way into rivers via treated wastewater effluent.

Investigations in Germany, Brazil and Canada showed that relatively small loads of natural estrogens, metabolites and contraceptives were discharged with wastewater effluent. More than 60% of some of these substances were removed in activated sludge systems (Ternes, 1999a). The transformation of natural estrogens, contraceptives and estrial glucuronides in contact with activated sludge were studied using aerobic batch experiments (Ternes, 1999b). The experiments showed how different estrogens are transformed in wastewater treatment works. Whether the small concentrations of estrogens encountered in German rivers cause disruptions of the endocrine system was questionable; as fish and other aquatic animals exhibit naturally low concentrations of some of those analysed substances.

In the Netherlands Belfroid *et al* (1999) developed an analytical procedure that enables routine analysis for four estrogenic hormones at concentrations below $1 ng/\lambda$. Data from water samples collected at various locations in the Netherlands have shown that concentrations of (all tested) hormones were higher in municipal wastewater effluents than in industrial effluents (up to 47 ng/l in municipal effluent in the case of estrone).

On the other hand, Flemish rivers have higher estrogenic potency, when compared to treated wastewater effluent (Witters *et al*, 2001). Extracts prepared from 16 selected water samples in Flemish rivers, municipal wastewater effluent and drinking water reservoirs were analysed for estrogenic activity. Half of the samples were believed to contain micro-pollutants adversely effecting the reproduction success of wildlife. This had to be verified *in vivo*.

5.2.3 Occurrence of pharmaceutical end products in wastewater and surface water

Pharmaceuticals found in wastewater effluent have included lipid regulating drugs (e.g., phenazone and fenofibrate), analgesics (e.g., ibuprofen and diclofenac), chemotherapy drugs, beta-blocker heart drugs, epilepsy controlling drugs, drugs for contrast during scans, antiseptics and antibiotics. Kümmerer (2001) gave a recent overview of pharmaceutical drugs in the environment.

In 1999 there was still no comprehensive data on the occurrence of antibiotics in wastewater effluents. Antibiotic residues in the environment are suspected to induce resistances in bacterial strains. This phenomenon could threaten public health if the present antidotes were no longer effective for treating infections. In a study of the occurrence of 18 different antibiotics in various waters, concentrations ranged between $0.1 \mu g/\lambda$ and $1.8 \mu g/\lambda$, depending on the type of antibiotic and water source. The median concentration in surface waters were generally one order of magnitude lower than the median of treated wastewater effluent (Hirsch *et al*, 1999). Hirsch concluded that potential risk for public health couldn't be assessed yet, as too little is known to propose causal relations between occurrence of resistant bacteria and the concentrations of specific antibiotics in aqueous environments.

The same group of researchers investigated the presence of polar drug residues in waste-, surface- and drinking water (Stumpf *et al*, 1999). Composite samples were taken of raw wastewater, the maximum concentrations of 13 different drugs were in the range of $1 \mu g/\lambda$. Removal of these drugs in activated sludge varied greatly, between 34% and 83%.

5.2.4 Effects of micro-pollutants on aquatic ecosystems

The studies mentioned above made it clear that micro-pollutants occur in varying concentrations in surface waters. Furthermore, aquatic ecosystems have shown signs of endocrine disruption. However, the pathways leading to endocrine disruption in organisms are, mechanistically seen, very complex. Some researchers are now trying to unravel the processes and agents causing endocrine disruption. One of the areas of research at EAWAG is Environmental Microbiology and Molecular Eco-toxicology. Projects within this study group include topics such as:

- Involvement of various hormone receptors in endocrine disruption in fish;
- Identification and characterisation of oestrogen-induced genes in fish;
- Assessment of the endocrine disrupting potential of wastewater treatment plants by (sub)cellular and whole organism testing.

This research is carried out in collaboration with the Novaquatis project at EAWAG (Chapter 3 of this report). The special effects of urinary excretion on endocrine disruption, as well as the reduction of eco-toxicity in surface water (where wastewater effluent is discharged) due to separate urine collection and treatment will be evaluated.

Fish biologists also aim to translate the measured endocrine disrupting potential to biological effects on whole organisms. The COMmunity Programme of Research on Environmental Hormones and ENdocrine Disrupters (COMPREHEND) is an initiative between Switzerland, France, Germany, Scandinavia, Belgium and the Netherlands to co-operate with research on endocrine disrupters in fish (and the general aquatic environment). In the Netherlands, RIZA (Rijksinstituut voor Integraal Zoetwaterbeheer en Afvalwaterbehandeling) will assess the estrogenicity of representative sewage and industrial effluents in the Netherlands, Belgium and Germany. This will include an analysis of existing long-term data for evidence of population changes associated with exposure of freshwater bream to estrogenic effluents. In the Netherlands RIZA will also chemically analyse the livers from exposed fish for known estrogenic compounds (alkylphenols and steroids). Sweden (IVL) and Switzerland (EAWAG) will co-develop a Toxicity Identification Evaluation method, using the combined approach of biological testing with traditional analytical chemistry as a generic technology for detecting endocrine disruption. In the Netherlands, RIVO-DLO (Rijksinstituut voor Vis Onderzoek – Wageningen University) will examine and analyse existing, long-term fisheries data (including information on sex ratios, gonadosomatic indices, hepatosomatic indices) for evidence of changes associated with a known history of exposure to effluents.

5.2.5 Micro-pollutants and urine fertiliser

In addition to nutrients (N, P and K), plants might also take up micro-pollutants where urine is used as a fertiliser. Micro-pollutants could possibly also return to the body via meat of animals that grazed on pastures fertilised with urine. The endocrine disrupting potential of untreated urine fertiliser on humans is still unknown. The Novaquatis group at EAWAG is now assessing risks associated with micro-pollutants in urine and the risk that these micro-pollutants may be taken up by plants and enter the human food cycle. The risk will be assessed in both the raw product (urine) and in fractions from the different urine treatment schemes being developed. They aim to ultimately develop methods to use in assessment of the quality of urine based fertiliser products.

Micro-pollutants in wastewater are diluted 300 times in relation to the concentration in urine. If urine were collected separately, whether for use of fertiliser or not, micro-pollutants could be removed from urine much more efficiently than from wastewater.

5.3 Reduction in harmful emissions from combined sewer overflows

The combination of storm water and wastewater run-off in a single drain network has a historical rather than technological explanation. Sewers were initially constructed to drain rainwater run-off. Later, with the introduction of flush toilets, existing infrastructure (storm water drainage) was used to drain wastewater too, which is now treated at the end-of-the-pipe. During or after rainstorms, the flow rate in combined sewers exceeds the hydraulic capacity. When this occurs, the mixture of storm- and wastewater flows over weirs and directly into surface water.

Mullis *et al* (1997) compared the impacts of discharges from two combined sewer overflows (CSOs) on urban watercourses in London. At the onset of overflows, elevated concentrations of both phosphate and total nitrogen were measured. The ammonia concentrations directly after CSOs increased up to around 25 gN/m^3 at both sites. Subsequently pollutant levels fell below dry weather flow concentrations as dilution and exhaustion of supply dominates conditions. In a similar study in France, the ammonia discharged to the Oise River was measured (Betrand-Krajewski *et al*, 1997). Up to 85% of total ammonia emissions resulted from CSOs during or directly after rainstorms. In this case, ammonia removal optimisation (in the treatment works) appeared to be less important in comparison to preventing the occurrence of CSOs.

Traditionally, engineers deal with the problem of combined sewer overflows (CSOs) by building retention basins. The volume of retention basins are still mostly determined by the product of the catchment area and design precipitation: 1ha x 1mm = 10m^3. This is a simple solution for a complex problem, but neither statistical nor deterministic models have been adequate in predicting CSO emissions accurately.

Data has been collected since 1982, on the occurrence and magnitude of CSOs in the Netherlands. Veldkamp and Wiggers (1997) developed a statistical method based on this data. The model treats the sewer network, overflow weirs and wastewater treatment plants as a black box to predict emissions. However, it could give no insight into the behaviour of nutrients or other dissolved pollutants in combined sewers.

Grum *et al* (1997) attempted to "disclose the underlying structure of systematic variations in the event mean concentrations of pollutants in combined sewers during rainfall events". Five combined sewer catchment areas were used in the study (three in the Netherlands and two in Denmark). Statistical analyses showed a similar distribution of events in three "event type groups", where the effluent concentrations were also divided amongst types (low, medium and high particulate and dissolved pollutant concentrations). It was suggested that modelling single sewer overflow events could be improved by considering these differences.

Holzer and Krebs (1998) showed in their model study that ammonium peak concentrations in rivers due to CSOs result largely from the washout effect in the main sewer at the onset of rain events. They modelled the effect of 25 rain events on a total wastewater system (near Zurich) to distinguish between the ammonium load from the wastewater treatment plant and from CSOs. It was assumed that storm water in pipes forms a wave, concentrating pollutants already present in sewers, so that the first overflow of combined sewage is not diluted. This only holds where sewers have steep slopes. Ruan (1999) developed a model for the Netherlands, based on data from the nation wide research programme (NWRW, 1989). The model was based on the traditional assumption that sediments accumulate on hardened surfaces and in sewer networks during dry-weather periods. During rain events, the surface sediments enter sewers and mix with re-suspended solids. Although the CSO discharges (flow rates) from different rain events were modelled effectively, model predictions for pollutant emissions due to CSOs did not reflect measurements.

Apart from the combined sewer overflows, emissions due to decreased efficiency of wastewater treatment works during rainstorms should also be considered. Increased flow rates with a diluted nutrient concentration (equalling the total daily load) increase the effluent ammonium and total nitrogen concentrations. Emptying retention basins back into sewers after storms also increases the effluent concentration from treatment works, due to the increased flow rate and the effective increase in "storm" duration. Rauch and Harremoës (1997) urged the need for a total system evaluation. Water pollution is caused by emissions from both CSOs and wastewater treatment works. Neglecting any of the components gives a wrong impression, such as positive effects of retention basins that could be overestimated.

Several pollutant storage, transport and emission mechanisms have been postulated, but variations in pollution concentrations, following from seemingly similar conditions, remain mostly unexplained. However, most of the nitrogen and phosphorus, and some of the micro-pollutants in wastewater originate from urine, and therefore emissions of these substances to surface waters directly relate to urine. Effects of separate urine collection on the performance of wastewater treatment works under dry weather conditions, is discussed in detail in Chapter 6.

At present, various methods are proposed for reducing the occurrence of CSOs. The approach even varies within countries. One such approach is that of the municipality of Nijmegen. A target has been set to separate 40% of the storm water drainage from wastewater in 2015, estimated at € 4 million. Modern cities, such as Lelystad, have constructed storm and wastewater drainage systems separately from the start. It is impossible to say if this solution would become common practice in the Netherlands, and if so, what the time-scale of such projects would be.

5.4   Conclusions

Wastewater treatment works are not designed to remove micro-pollutants (natural- and synthetic hormones and pharmaceutical end products) under normal conditions. Although measurements indicate increasing micro-pollutant concentrations in surface waters, the quantity and effect of micro-pollutants are not yet fully understood. Studies on aquatic ecosystems have also indicated the occurrence of endocrine disruption. Although the relation is not clear, micro-pollutants are believed to cause endocrine disruption, which might also be responsible for some chronic illnesses and disorders amongst people.

The most of the micro-pollutants in municipal wastewater originate in human urine. Micro-pollutants currently discharged into surface waters would be reduced if urine were collected and treated separately. However, there are also unanswered questions regarding the effect of the micro-pollutants on human and animal health where urine is used as fertiliser.
Sewer systems do not ensure that municipal wastewater always reach the treatment works. During intensive rain events, nutrients and micro-pollutants are discharged directly into surface water with combined sewer overflows. The higher flow rate following rain events also reduce the efficiency of biological nutrient removal and increase the nutrient load in the wastewater effluent. Both statistical and deterministic models fail to accurately predict N and P emissions due to combined sewer overflows. The variety of factors seems to deny the notion of "an underlying structure" according to which combined sewer overflow emissions could be modelled.

Some authorities construct new pipe networks that will drain storm water separately from domestic wastewater to reduce the effects of combined sewer overflows. If the political decision is taken to allow no combined sewer overflows (either by increasing basin capacity or separate storm water sewers) the effect of urine on combined sewer overflow emissions becomes irrelevant.

Urine in wastewater is *believed* to have an effect on combined sewer overflows emissions, micro-pollutants in the aquatic environments and the quality of surface water. These emissions and their effects have not (or can not) be modelled or quantified yet.

6. EFFECTS OF URINE SEPARATION ON BIOLOGICAL NUTRIENT REMOVAL PROCESSES

6.1 Introduction

Nitrogen (N) and phosphorus (P) emitted into surface waters cause problems such as eutrofication (Portielje, 1998 and Hosper, 1998). Throughout the world, wastewater treatment plants are constructed or upgraded to remove more N and P in order to meet stricter effluent standards. Chapter 2 of this report pointed out that urine is the greatest source of N and P in municipal wastewater.

Experience from ecological villages in Sweden shows that urine can be collected separately (refer Chapter 3). The first attempts to collect urine separately were mostly pilot projects and experiments. The efficiency of urine separation systems is expected to improve as more of these projects are commissioned and operated.

Growing bacteria require nutrients (N and P). The nitrogen content of bacteria is approximately 0.1 gN/gVSS and the phosphorus content is 0.03 gP/gVSS (Ekama and Marais, 1984). This means that for wastewater with an influent COD:N:P ratio of approximately 100:5:1, almost all the nitrogen and phosphorus will be used for cell growth. The amount of nitrogen and phosphorus in municipal wastewater exceeds this ratio and treatment must therefore include biological nitrification/denitrification and excess phosphorus removal. If urine would be collected separately, the influent nutrient load could be reduced to match the cell growth requirement. For wastewater containing little or no urine, wastewater treatment processes would be simpler, reactors would be smaller and the effluent quality would be better (Larsen, 1999).

This chapter theoretically evaluates the effects that separate urine collection could have on municipal wastewater and centralised biological nutrient removal processes. The effect of urine separation can best be valued when compared to non-separated (mixed) wastewater. Initially, the possibility of reaching a future effluent quality of $N_{tot} = 2.2$ g/m^3 and $P_{sol} = 0.15$ g/m^3 by collecting urine separately, was considered. However, this future standard is not viable with available technology. New treatment processes must be proposed before such a comparison could be realistic. The scope of this project did not allow for new process development and it was decided to study only the effects that urine separation could have on existing processes.

Where biological nutrient removal is not yet optimised, a decrease in the influent nutrient load will also decrease the final effluent load. However, if the benefit of separate urine collection is insignificant for advanced treatment processes, future investment should rather be in better treatment process than in urine separation technology. If it can be shown that advanced treatment processes could benefit from separate urine collection, then processes with lower removal efficiencies (e.g. including chemical P-removal) would benefit even more from urine separation. Most advanced wastewater treatment works operate according to variants of the modified UCT process. For this study we chose the BCFS® process (Biological/Chemical Phosphorus and Nitrogen removal) as reference process. This process is used in a number of plants in the Netherlands and produces good quality effluent (N < 10 gN/m^3 and P < 0.5 gP/m^3). The possible effects of urine separation on the simplest activated sludge process – an aerated biological reactor with sludge return only - was also investigated.

Only the activated sludge processes were considered in these evaluations. Other processes, such as sludge treatment etc., were not considered. It was assumed that nutrient-rich side streams could be treated with existing processes (such as Sharon Anammox for N-removal or struvite precipitation for P-removal). Furthermore, implications of urine treatment (once it is separated) were also not considered here. Treatment processes for concentrated waste streams are discussed in more detail in Chapter 7 of this report.

6.2   Description of theoretical set-up and computer model

6.2.1   Modelling the BCFS® process at Hardenberg wastewater treatment works

In this study, the BCFS® process at Hardenberg wastewater treatment works was used as a basis of comparison for different scenarios, because a well-calibrated model was already available.

Hao et al (2001) modelled the BCFS® process by combining the metabolically structured bio-P removal model (Smolders et al, 1994, Kuba et al, 1996 and Murnleitner et al, 1997) with parts of the activated sludge model ASM2d for COD and N removal (Henze et al, 1999). This model has been used to simulate several full-scale processes successfully (van Veldhuizen et al, 1999, Brdjanovic et al, 2000 and Meijer et al, 2001). We used the computer software package AQUASIM 2.0 (Reichert, 1998) to implement the dynamic simulation of the BCFS® process.

Figure 16 is a schematic representation of the BCFS® process at Hardenberg. The process comprises four in-line biological reactors with a contact tank between the anaerobic and anoxic zone. The total volume of the five compartments is 10 000 m^3. A secondary settling tank (2 800 m^3) downstream of the final aeration basin completes the process. Three pumps return mixed liquor to different reactors and settled sludge is returned at a ratio of 1:1.2. Waste activated sludge is withdrawn from the clarifier underflow.
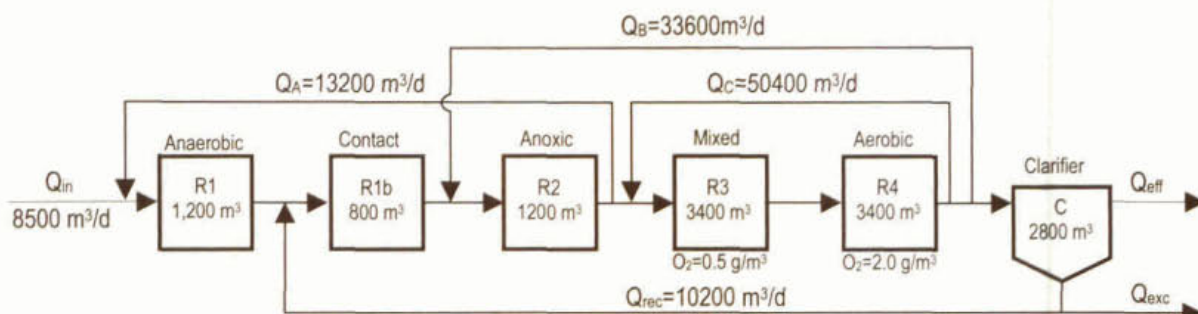
Figure 16: BCFS® process at Hardenberg

The effect of urine separation on a simple bio-reactor was also simulated with this model by simply omitting all zones and return flows except for the aerobic zone and clarifier underflow return.

6.2.2 Influent and effluent characteristics and using the calibrated model

The average Dutch influent characteristics accepted for this study (refer Chapter 2) differ from the actual wastewater in Hardenberg. Meijer et al (2001) calibrated the modelled treatment plant under influent values as shown in table 9. The current average wastewater discharge to Hardenberg (8 500 m^3/d) is less than the design flow rate. At the current flow rate, the total nitrogen effluent concentration is much lower (N_{tot_eff} is approximately 3.5 gN/m^3) than the effluent standard of 10 gN/m^3.

Table 9: Influent characteristics, actual effluent concentrations and simulated effluent concentrations at maximised flow rate

Parameter	Unit	Actual influent Hardenberg (June 1998)	Actual effluent Hardenberg (June 1998)	Simulated effluent for Hardenberg	Average Dutch influent (this study)	Simulated effluent for this study
Flow rate	(m^3/d)	6900	-	-	-	13500
COD_{tot}	(g/m^3)	625	44	43	537	39
P_{tot}	(g/m^3)	9.5	0.4	0.2	8	0.25
NH_4^+-N	(g/m^3)	51	0.4	0.5	40	0.9
N_{tot}	(g/m^3)	60	4.0	5.5	50	6.7

The maximised inflow rate, 13 500 m^3/d with average Dutch influent concentrations, was determined by iteration for this study. Up to this inflow, the effluent ammonium concentration was limited by increasing the dissolved oxygen concentration in the mixed reactor from 0.5 g/m^3 to 2 g/m^3 (sacrificing denitrification capacity for nitrification capacity). For inflow rates above 13 500 m^3/d, the nitrification capacity is too low to maintain the effluent ammonium concentration below 1 gN/m^3. All other simulations and comparisons at different percentages of urine separation were based on this maximum flow rate and influent values. None of the return flow rates was changed when the influent was increased to the simulated maximum (13 500m^3) or with different urine separation scenarios.

6.2.3 Sludge age and solids concentration

In all simulations of the BCFS® process the total suspended solids concentration (TSS) was kept constant at 5 000 g/m^3. In practice, sludge age controls the TSS. In the simulations, the sludge age was adjusted by iteration until a TSS of 5 000 g/m^3 was reached. The sludge volume index of existing BCFS® processes is below 120 ml/g. It was assumed that the sludge will separate and settle well in the secondary settling tank.

Simulations of the process with a pure aerobic reactor were done for low sludge ages (between one and six days). It was investigated whether growing bacteria would utilise all available nutrients at low sludge ages (high production of sludge mass). For these pre-selected sludge ages, the TSS varied from 1 100 g/m^3 to 4800 g/m^3 respectively.

6.2.4 Temperature

A constant temperature of 12°C was used for all simulations of the BCFS® process. Effects of different sludge ages of the completely aerobic reactor were also evaluated at 12 °C.

In a second set of simulations, the effect of temperature on the completely aerobic reactor was investigated at a constant sludge age of two days. Temperatures used in the simulations included 8, 12, 16 and 20°C.

6.3   Effects of separate urine collection on influent characteristics

6.3.1 Modelling urine separation

If urine separation technology were implemented in the Netherlands, it would be phased in over decades. Different scenarios were defined to study the effects of partial urine separation on wastewater and the nutrient removal process.

The effects of urine separation were first studied for 5%, 10%, 33%, 50% and 100% urine separation. The separation percentage indicates the percentage of the total urine production that did not reach the wastewater treatment works, irrespective of the means employed.

It seemed that small separation percentages (5% and 10%) would have no real effect on the treatment process. Non-linear phenomena occurred at high separation percentages and the range between 80% and 100% urine separation was investigated in more detail. Consequently, most of the simulations were done for separation percentages of 0%, 25%, 50%, 75%, 85%, 90%, 95% and 100%.

6.3.2 Reduction in wastewater volume

Urine contributes only a small volume to the total wastewater volume. However, the water used to flush urine is a significant fraction of the total. In this study, total urine flush water was assumed to be 35 λ/p.d including toilet visits away from home, based on measurements in Stockholm (Jönsson *et al*, 1997). It was further assumed that if 100% urine separation could be achieved, wastewater discharge to the treatment works could be reduced by 36.25 λ/p.d (including 1.25λ/p.d urine). Wastewater volume reductions at all other percentages of urine separation are considered linear.

6.3.3 Reduction in COD load

A concentration of COD_{tot} = 537 g/m^3 was assumed (refer Chapter 2). The COD fractions were divided into different model components according to the STOWA method (1996) as shown in Table 10.

Table 10:   Influent COD fractions (g/m^3) of raw wastewater

Soluble acetate	Soluble fermentable	Soluble inert	Particulate inert	Particulate slowly degradable
S_a	S_f	S_i	X_i	X_s
64	70	32	134	236

Separate collection of urine will lead to a small decrease in the COD load discharged to wastewater treatment works. Experiments with activated sludge showed that about 85-90% of the COD in diluted and stored urine is biodegradable and the rest is inert (Udert, 2000). Based on measurements from the same experiments, the COD load in urine was assumed to be 12 gCOD/p.d (the total load in municipal wastewater is 161 gCOD/p.d). It was further assumed that the COD in urine is divided according to the following ratio:

S_f = 0.88*12g/p.d and S_i = 0.12*12g/p.d.

The product of the total urine COD load and separation percentage was subtracted from the COD_{tot} load. The fractional COD loads for the different urine separation percentages were divided by the flow rate to yield the average COD concentrations.

6.3.4 Reduction in nitrogen load

Nitrogen in urine is mainly present as urea, $CO(NH_2)_2$. This soluble compound rapidly hydrolyses to NH_4^+, and HCO_3^+. Addition of wastewater to pure urine significantly increases the decomposition rate of urea (Helström *et al*, 1999). Most of the sewers in the Netherlands are flat and retain wastewater for some time during which urea dissociates. It was assumed that all the nitrogen in urine would reach the wastewater treatment works as ammonium.

It was assumed that 12 gN/p.d (out of 15 gN/p.d in wastewater) is excreted via urine. The influent soluble ammonium concentration (S_NH4) is then 40 gN/m^3 at zero urine separation. This means that 80% of the nitrogen in wastewater originate from urine and that at complete urine separation, the total nitrogen influent concentration drops from 50 gN/m^3 to 10 gN/m^3. The different nitrogen fractions from sources other than urine, was divided as shown in table 11. The default value was used for i_NXI, while the other fractions were lowered slightly to yield a sum equal to 10 gN/m^3 (20% of total) that is *not* ammonium.

Table 11: N-Content of different influent fractions (g N/g COD) of raw wastewater

Acetate	Fermentable substrates	Inert soluble material	Inert particulate material	Slowly bio-degradable substrates
i_NSA	i_NSF	i_NSI	i_NXI	i_NXS
0	0,010	0,010	0,030	0,023

The nitrogen load related to influent COD load (table 11) was not changed for increasing urine separation. Only the influent ammonium concentration (S_NH4$_{in}$) was reduced with increasing percentages of urine separation. The products of the total nitrogen load excreted via urine and the separation percentages were subtracted from the total nitrogen load at zero urine separation. The concentrations of influent ammonium and total nitrogen were determined with the reduced flow rates (due to flush water saving) for different separation percentages. The concentration of S_NH4$_{in}$ varies linearly between 40 gN/m^3 and zero gN/m^3, for no separation and 100% separation respectively.

6.3.5 Reduction in phosphorus load

All the phosphates in the influent were assumed soluble. This means the variables i_PSF, i_PSI, i_PXI and i_PXS were all set to zero during the simulations. The inflow concentration of S_PO4 was assumed 8 gP/m^3 in wastewater with zero urine separation and the phosphorus load from urine 1gP/p.d. The products of the total phosphorus load excreted via urine and the separation percentages were subtracted from the total phosphorus load at zero urine separation. The influent phosphate concentrations were determined with the reduced flow rates (due to flush water saving) for different separation percentages. At 100% urine separation, the total influent phosphorus concentration is 5.3 gP/m^3. All other concentrations were considered linear with the urine separation efficiency.

6.4 Effects of urine separation on treatment of raw wastewater

The optimised process model was used to evaluate the effects of different influent values as shown in Appendix 3 for raw sewage. Recycle flow rates were equal to those shown in figure 16. At a sludge retention time (SRT) of 13.5 days, the simulated total suspended solids concentration (X_{tss}) was 5 000 g/m^3.

This model was used to simulate different urine separation scenarios by changing the influent concentrations. The SRT had to be reduced to maintain X_{tss} at 5 000 g/m^3 (from 13.5 days at zero separation to 12 days at 100% separation). The flow rate refers to the zero separation scenario. The most important results from the simulations are presented in figure 17.
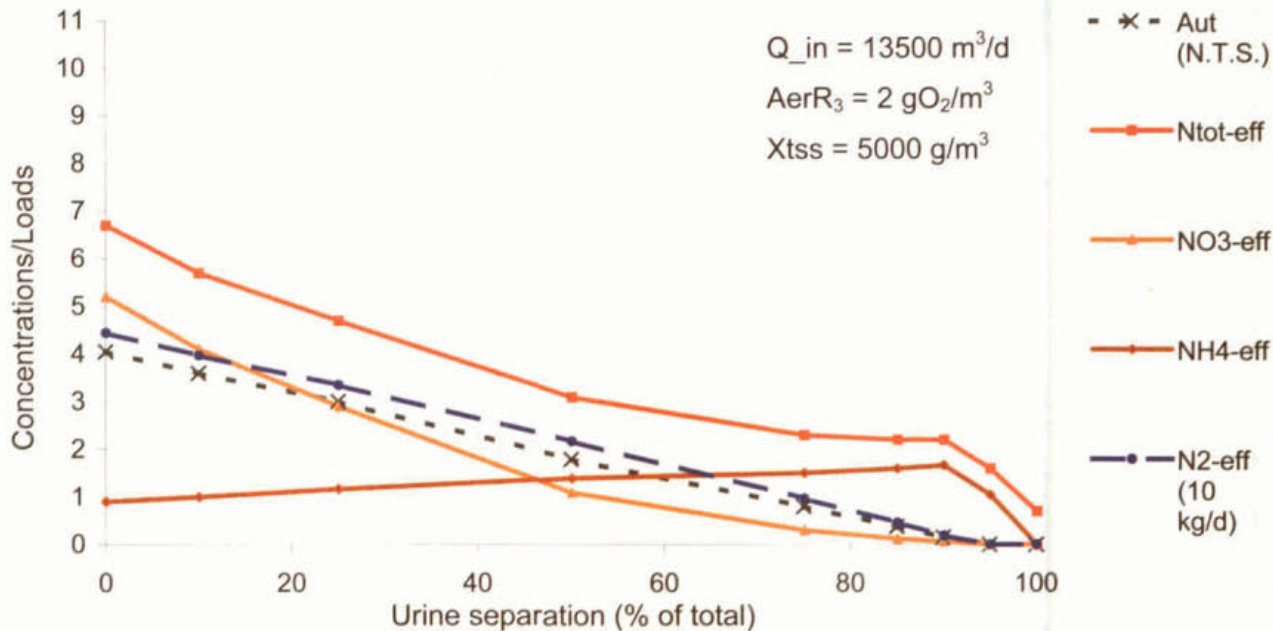

Figure 17: Effects of urine separation on N removal in advanced UCT-type wastewater treatment processes

The effluent ammonium concentration (NH$_4^+$$_{eff}$) seems to increase with increasing urine separation (which means a reduction in the influent ammonium load). This is ascribed to the decrease in the autotrophic bacteria population (nitrifiers) that depends on ammonium as substrate. Effluent nitrate concentration (NO$_3$$_{eff}$) decreases with increasing urine separation. At zero separation, NO$_3$$_{eff}$ > 5 g/m^3, while at 50% separation NO$_3$$_{eff}$ = 1 g/m^3. The denitrification potential of a treatment plant is proportional to the influent biodegradable COD concentration. A decrease in the ammonium load at a near constant COD load increases the COD/TKN ratio. Relatively more nitrate can be denitrified with increasing urine separation efficiency, but at the same time less nitrate is produced. While NO$_3$$_{eff}$ decreases non-linearly, the amount of dinitrogen gas produced decreases linearly with increasing urine separation. The decrease in nitrification shapes the profile of the N$_{tot\_eff}$ with increasing urine separation.

The total nitrogen in the effluent (N$_{tot\_eff}$) decreases with increasing percentages of urine separation. N$_{tot\_eff}$ is the sum of ammonium, nitrate and nitrogen contained in suspended solids (not settled in the clarifier). This explains why N$_{tot\_eff}$ = 0.7 g/m^3 at 100% urine separation, while effluent concentrations of ammonium and nitrate are zero at this level. An interesting point is that the model predicts N$_{tot\_eff}$ = 3.2 g/m^3 at 50% urine separation. This is the current effluent concentration at Hardenberg, at a much lower influent load (8 500 m^3/d at current compared to 13 500 m^3/d in the model). This suggests that the capacity of a wastewater treatment plant can be increased by 60% (not hydraulically) with 50% urine separation, while maintaining the same effluent quality. One observes a substantial decrease in N$_{tot\_eff}$ with urine separation up to 50%. The decrease is less obvious for urine separation efficiencies above 50%.

Virtually all phosphate had already been removed in the case of zero urine separation. No change was expected in the effluent phosphate concentration and this was confirmed by the simulations of increasing urine separation. The only effect will be a lower P-content of the

sludge. However, this is only true for biological excess phosphorus removal. Section 6.11 of this chapter deals with chemical phosphorus removal.

The complete spreadsheet with nutrient loads and concentrations for different urine separation scenarios is attached as Appendix 3 of this report.

6.5   Effects of urine separation on treatment of settled wastewater

The main advantage of pre-settling is the reduction of influent COD, saving energy due to reduced aeration requirement. However, removing COD from the influent by pre-settling reduces the denitrification capacity in general. With a reduction in the total influent nitrogen (urine separation), the denitrification capacity need not be as high. The combined effects of urine separation and pre-settling were therefore simulated.

It was assumed that 60% of the particulate COD could be removed in a primary settling tank. This was modelled by multiplying X_S and X_I by a factor of 0.4.

The flow rates, total suspended solids concentration, aeration capacity, temperature etc., were kept equal to that of the model without pre-settling. The sludge age was increased to 30 days in order to maintain X_{tss} = 5 000 g/m^3. Reductions in the influent loads due to urine separation were determined as described in the previous sections.

Figure 18: Effects of urine separation and pre-settling on N removal in advanced UCT-type wastewater treatment processes

The reduction in the influent COD concentration increases the nitrification capacity. The increased sludge age could also increase the nitrification capacity. The effluent ammonium concentration is therefore lower than in the case of raw wastewater.

A reduction in denitrification capacity is predicted for treatment of pre-settled wastewater. This can be seen by comparing the N_{tot_eff} for zero urine separation from figure 17 and figure 18. N_{tot_eff} = 6.7 g/m^3 with raw wastewater, while N_{tot_eff} = 8.7 g/m^3 with pre-settled wastewater, which is still within the effluent standard. Except for the profile's gradient, the N_{tot_eff} for pre-

settled wastewater show the same general pattern as for raw wastewater. At 50% urine separation, N_{tot_eff} for settled wastewater is equal to the N_{tot_eff} for raw wastewater. A comparison between figures 17 and 18 shows that, although the denitrification capacity is reduced with pre-settling, the N_{tot_eff} from pre-settled wastewater is lower than the N_{tot_eff} from raw wastewater for urine separation greater than 50%.

6.6   Effects of urine separation on treatment of pre-precipitated wastewater

Normal primary sedimentation of wastewater removes most particles with a nominal size of more than 65 µm. Particles with a nominal size of more than 65 µm will settle directly, while smaller particles (down to approximately 25 µm) have to be entrapped in flocs, before they can be removed. With pre-precipitation, 90% of the COD_{part} can potentially be removed by addition of high weight cationic organic polymers (Mels and van Nieuwenhuijzen, 2001).

Removing the COD_{part} from the influent with cationic organic polymer was modelled by multiplying the influent load of X_I and X_S with a factor of 0.1. The effects of urine separation on pre-precipitated wastewater treatment were simulated in the same way as described in the sections above. The TSS concentration of the activated sludge process was maintained at 5 000 g/m^3, but at such a low COD load, the sludge age had to be increased to 90 days. This means that the portion of dead cell matter in the sludge would be higher than normal. Figure 19 illustrates the most important effects of urine separation on wastewater treatment after pre-precipitation.
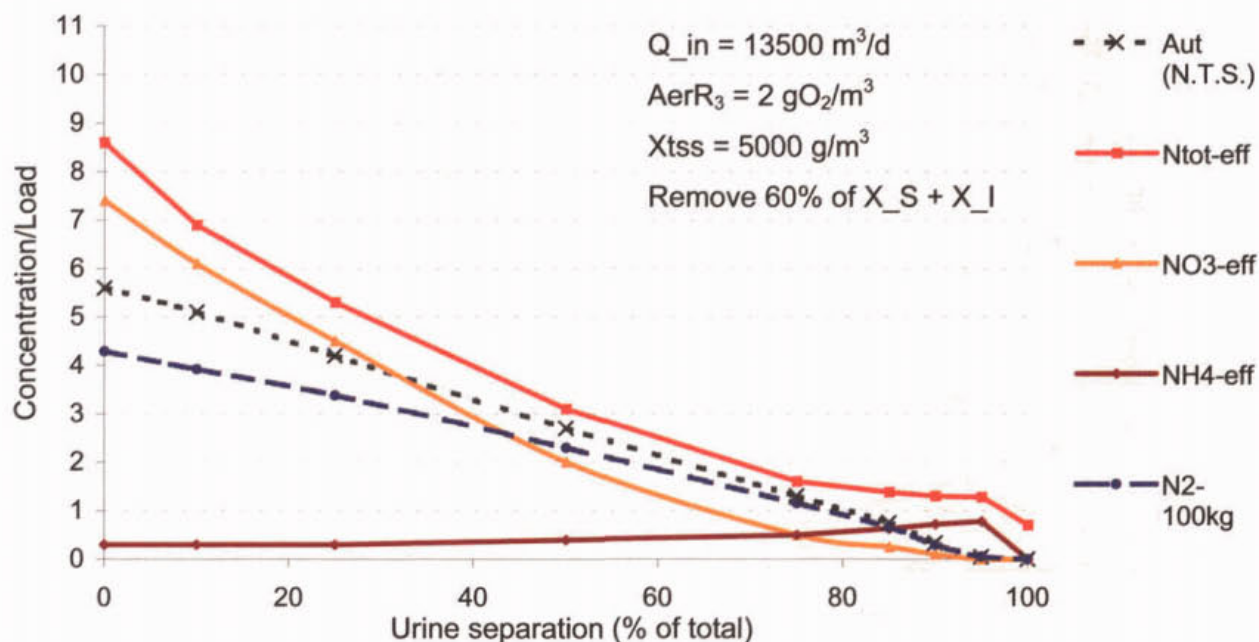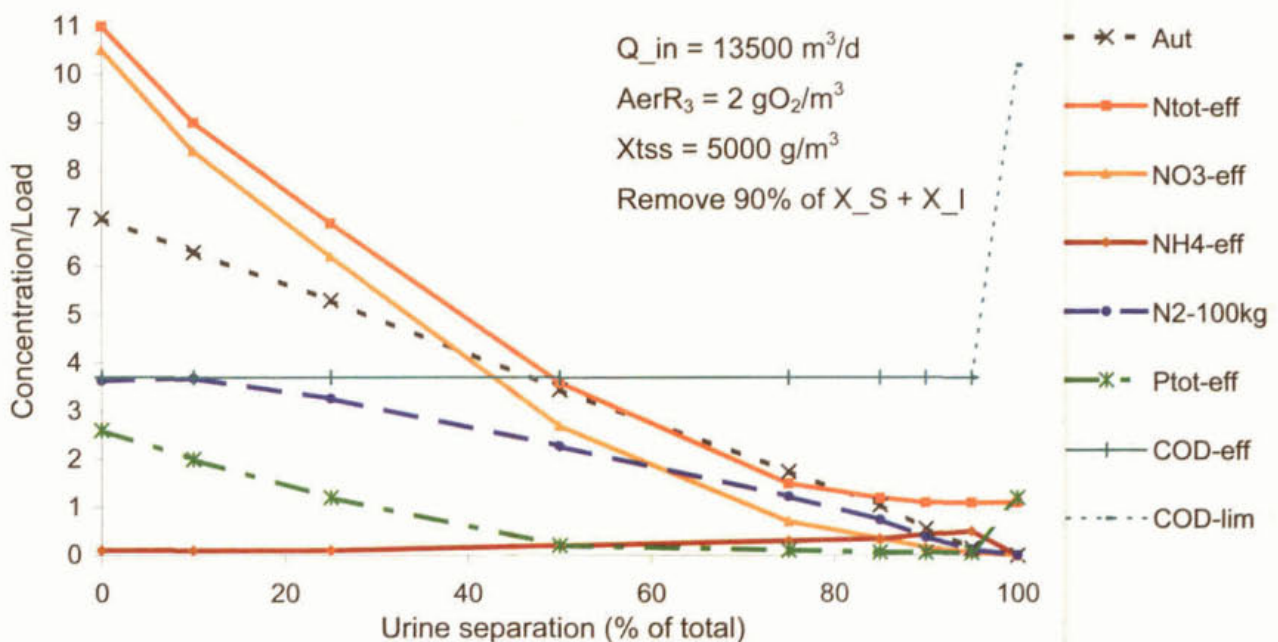

Figure 19: Effects of urine separation and pre-precipitation on N and P removal in advanced UCT-type wastewater treatment processes

The total nitrogen concentration in the effluent ($N_{tot} = 11$ g/m^3) for the zero separation scenario is above the allowed limit. The high nitrate concentration ($NO_3 = 10.5$ g/m^3) results from insufficient denitrification due to the absence of an electron donor (COD removed from influent) in the anoxic zone. With increasing urine separation, the total nitrogen concentration in the effluent is greatly reduced. At 50% urine separation, $N_{tot} < 4$ g/m^3. The effect of urine separation of more than 60% on the nitrogen effluent concentration is less obvious. The autotrophic population is larger than in the case of raw or settled wastewater treatment. The nitrifying bacteria remove virtually all the influent ammonium as well as ammonium released from dead cell matter.

Biological excess phosphate removal is not effective at zero and low urine separation percentages (P_{sol} = 2.6 g P/m^3 with zero separation). Although the amount of phosphate accumulating organisms (PAOs) in the sludge was almost equal to that of the process for treatment conditions for raw or settled wastewater, the amount of phosphorus removed from wastewater is ultimately dependent on the sludge age. An increase in the phosphorus removal was observed for simulations with sludge age of 70 days, due to higher sludge withdrawal rate. At higher urine separation percentages, the phosphorus removal remains the same, but a lower influent load also leads to a lower effluent concentration.

At high separation percentages (around 90%), nitrogen availability becomes limiting for bio-mass production. At 100% urine separation, the nutrients are insufficient to build cell mass, leading to increased effluent concentrations of chemical oxygen demand and phosphorus.

6.7   Increased denitrification capacity

In the previous sections, the dissolved oxygen concentration in reactor 3 was maintained at 2 gO$_2$/m^3. However, figures 18 and 19 made clear that the ammonia effluent concentration for settled and pre-precipitated wastewater is lower than that of raw wastewater, due to the relative increase in nitrification capacity. The dissolved oxygen concentration in reactor 3 could therefore be lowered to 0,5 gO$_2$/m^3. The "mixed compartment" (neither fully aerobic, nor anaerobic) would increase the denitrification capacity of the works and the total nitrogen effluent concentration would decrease. Figure 20 shows the effects of urine separation and different dissolved oxygen concentrations in reactor 3 on the total nitrogen effluent concentration.
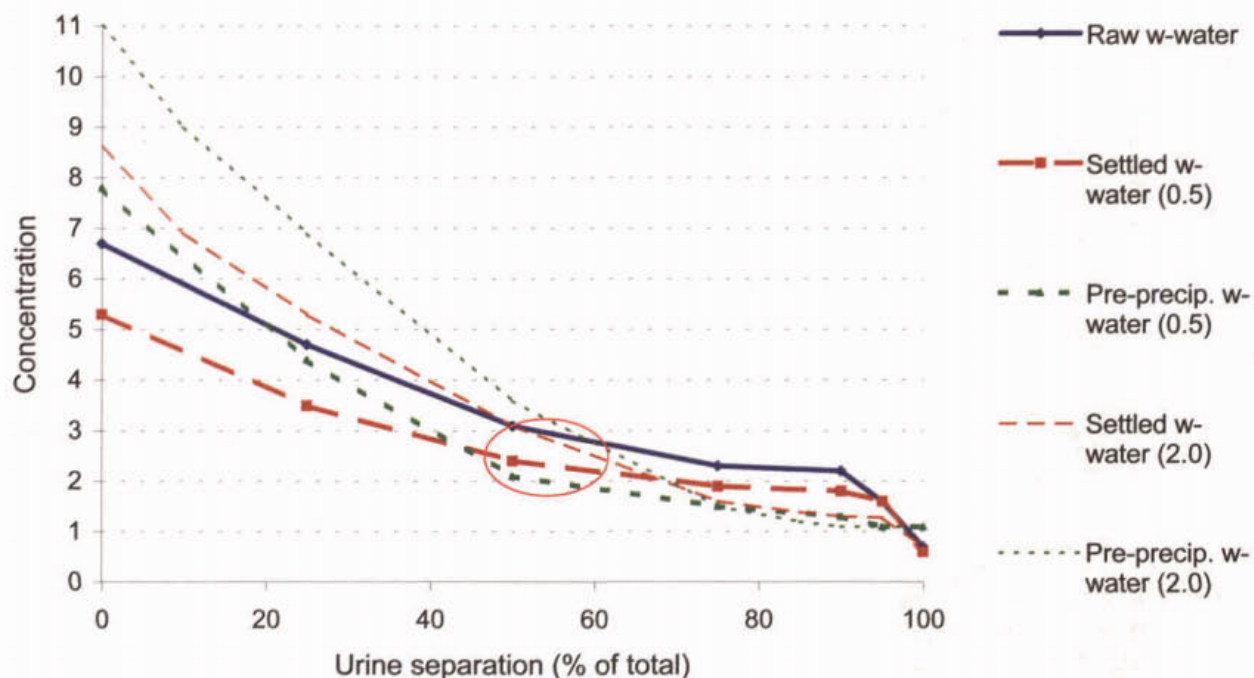

Figure 20:Effect of urine separation on total nitrogen effluent concentration in different scenarios for advanced UCT-type processes

Although the increased denitrification lowers the nitrogen effluent concentration, separate urine collection could reduce the effluent concentration even further to between 2 and 3 gN_{tot}/m^3 at around 50 - 60% urine separation (refer oval in figure 20). There is again no real difference in nitrogen removal performance for separation efficiencies above 60%, regardless of denitrification capacity.

6.8   Effects of urine separation on oxygen consumption in treatment processes

Stoichiometrically the oxygen requirement for nitrification is 4.57 gO_2/gN. When denitrification is complete, 2.86 gO_2/gN can be recovered relative to COD oxidation. However, when the ammonium influent load is reduced (through urine separation) the nitrification requirement is also reduced. Therefore, for each gram of ammonia collected separately, 1.71 gram of the total oxygen demand could be saved. Figure 21 illustrates the effect of urine separation on the oxygen demand of an advanced modified UCT-type process.

The saving in oxygen consumption at 50% urine separation is around 25%. Simulations showed no significant difference in oxygen consumption for the mixed zone of the BCFS® process maintained at 0.5 or 2.0 gO_2/m^3. The sudden drop in oxygen consumption at 100% separation is due to nutrient limitation and reduced substrate (COD) oxidation. The production of waste activated sludge (WAS) is nearly constant for each scenario at different separation percentages. The slight increase in WAS production with increased urine separation results from decreasing sludge age to maintain a constant X_TSS concentration.
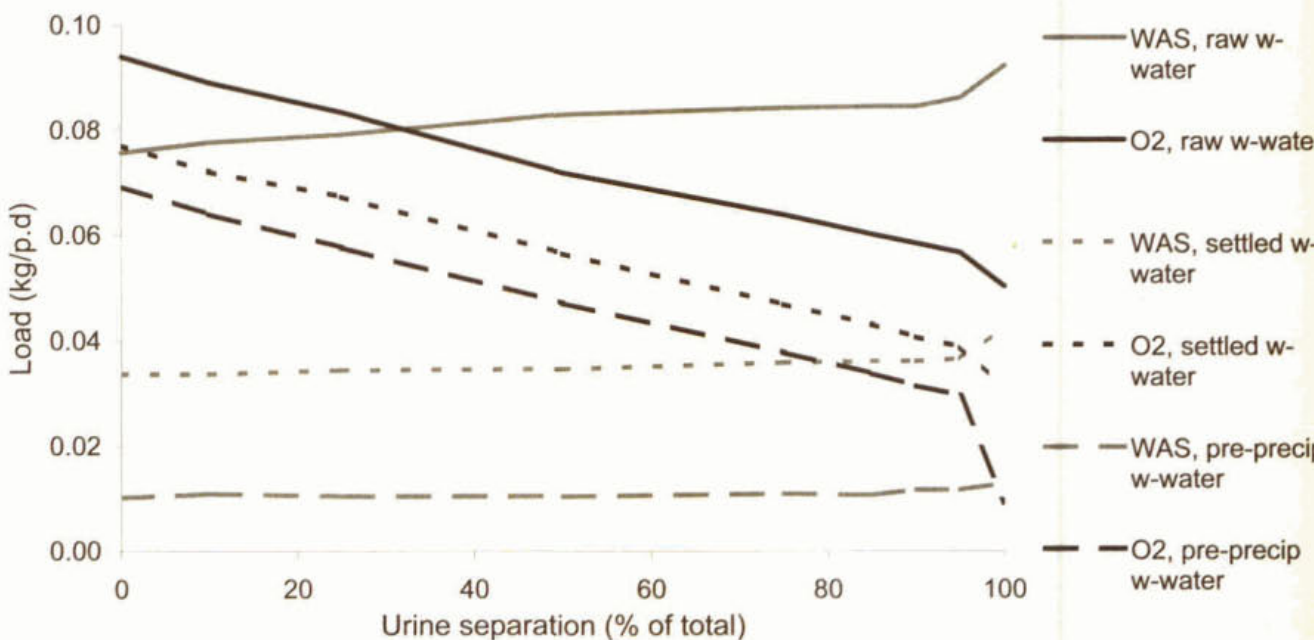

Figure 21:Effects of urine separation and pre-settling/pre-precipitation on oxygen demand and WAS production in advanced UCT-type wastewater treatment processes

6.9 Potential increase in methane production and decrease in energy consumption

Methane (CH_4) can be produced from settled primary sludge. Anaerobic sludge digestion and methane production can potentially supply all energy required for wastewater treatment (Hao *et al*, 2001). It is assumed that methanogenic bacteria convert 0.25 $gCH_4/gCOD$ of the biodegradable COD fraction. Energy conversion is assumed to be 20 MJ/kg methane gas (van Loosdrecht et al, 1997). Based on the COD removal from settling or pre-precipitation, between 210 kJ/p.d and 320 kJ/p.d can be generated through methane production. Figure 22 shows the potential energy production from settled and pre-precipitated primary sludge as 0.058 kWh/p.d and 0.088 kWh/p.d respectively.
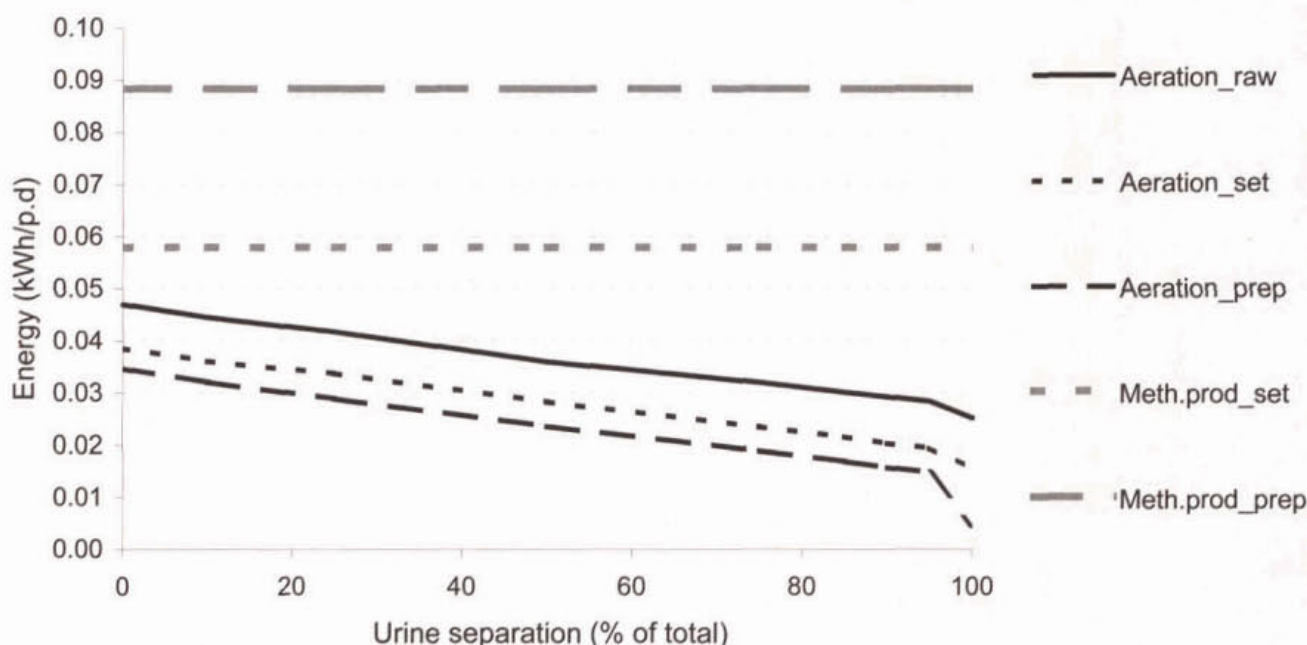
Figure 22:Energy consumption and potential energy production in advanced UCT-type processes

Another advantage of primary sedimentation is the direct removal of COD_{part} that would otherwise increase the aeration requirement. Most of the energy consumed in wastewater treatment is due to aeration. The energy consumption of aeration is assumed 1.8 MJ/kgO_2 (Tchobanoglous, 1991).

Although many still believe that primary sedimentation reduces the denitrification capacity, section 6.7 showed that this is not the case for advanced biological nutrient removal processes. Separate urine collection could decrease the nitrogen effluent concentration even further. With urine separation, low effluent concentrations of nitrate and ammonia could be maintained while a large amount of the COD load could be removed, either with pre-settling or with pre-precipitation. It is clear from figure 22 that urine separation could lead to an overall energy saving in wastewater treatment.

6.10 Possibilities of nitrogen removal with sludge production only

If all the nutrients present in wastewater would be required for cell growth, excess biological nutrient removal would no longer be necessary. The effects of urine separation were evaluated for a simple biological reactor consisting of a pure aerobic zone with a clarifier and return flow to the inlet. Urine separation and changes to the influent characteristics were modelled in exactly the same way as for simulation of the BCFS® process.
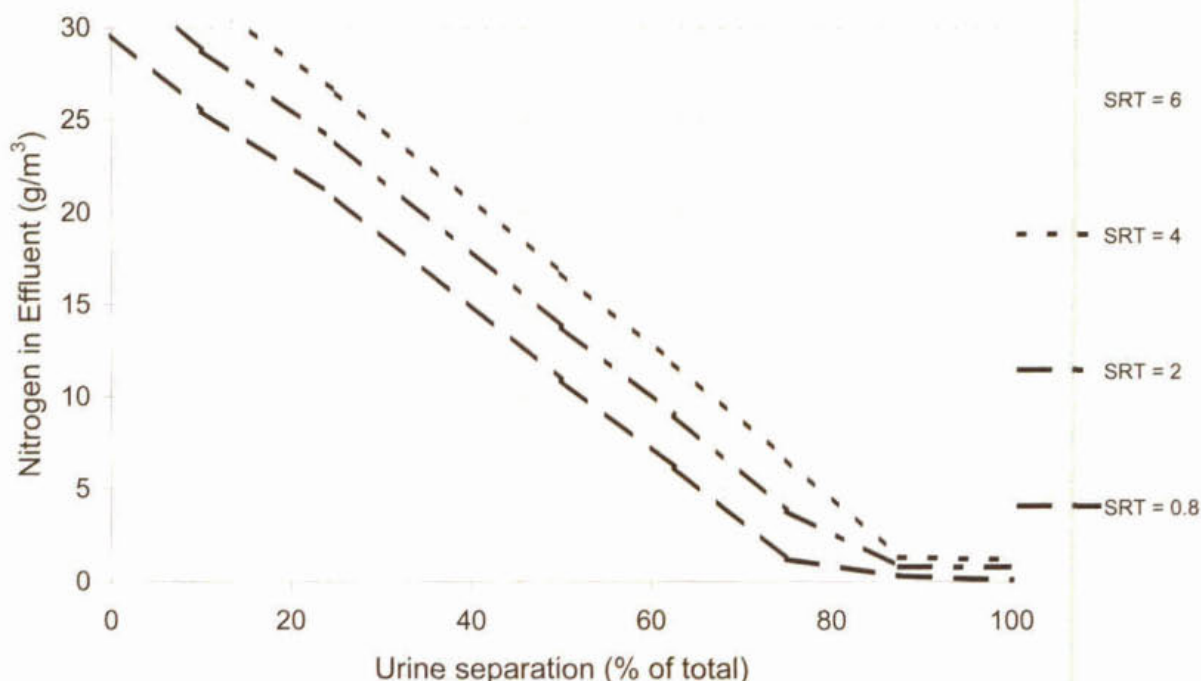
Figure 23: Effect of urine separation and sludge age (SRT) on nitrogen uptake in sludge at 12°C

The simulation was done for very short sludge ages to increase the sludge production. Figure 23 illustrates the effects of urine separation and different sludge ages on the total nitrogen effluent concentration. Some biological nitrogen removal seems to occur at sludge ages of 6 days and higher. Nitrification probably occurs in the aerobic zone and some denitrification in the sludge compartment of the clarifier. With short sludge ages, complete nitrogen removal through sludge withdrawal would only be possible at around 80% - 90% urine separation. However, at the same time, the nutrient concentrations would then be limiting for sufficient sludge production, resulting in a sharp increase in the effluent COD concentration.

Figure 24 illustrates the effect of different temperatures on nitrogen uptake by sludge at SRT = 2d. It seems that different temperatures would have little effect on the bio- mass uptake of nitrogen.

Figure 24:Effects of temperature on nitrogen uptake in sludge at SRT = 2d

The relative influence of the nitrogen influent fractionation on N-uptake by active sludge was also investigated. Tchobanoglous (1991) mentioned an activated sludge nitrogen content of between 2.5% and 5% of the total solids. The default values used in this study (refer table 11) results in a model prediction of approximately 5% nitrogen uptake.

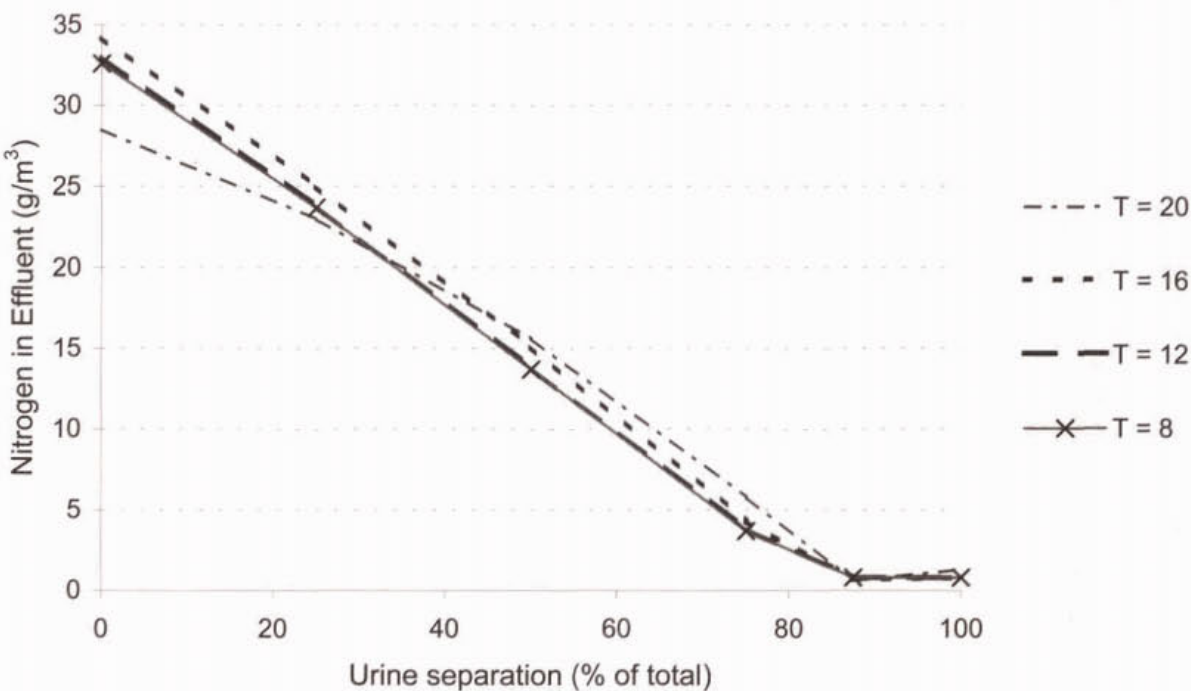However, these values are variable and the effect of an even higher sludge content was evaluated by increasing the N-content of inert particulate COD. Roughly, 50% of the total solids in waste activated sludge (simulated in this study) consist of inert matter. Table 12 shows how default values for nitrogen content of different wastewater fractions (table 11) were changed, while the sum of the total nitrogen fractions *not* including ammonia remained at 10 g/m^3.

Table 12: N-Content of different influent fractions (gN/gCOD) of raw wastewater with maximum inert concentration

Acetate	Fermentable substrates	Inert soluble material	Inert particulate material	Slowly bio-degrad-able substrates
i_NSA	i_NSF	i_NSI	i_NXI	i_NXS
0	0,000	0,000	0,060	0,008

Figure 25 shows that the predicted percentage of urine separation, at which nitrogen would be completely removed by sludge growth, is not too sensitive to variations in nitrogen influent fractionation. With approximately 55% urine separation, the effluent limit of 10 gN/m^3 could be reached. One should keep in mind that the N_{tot_eff} and the ammonium effluent concentrations are virtually the same in this case. With approximately 70% urine separation, an effluent concentration of 5 gN/m^3 could be reached.
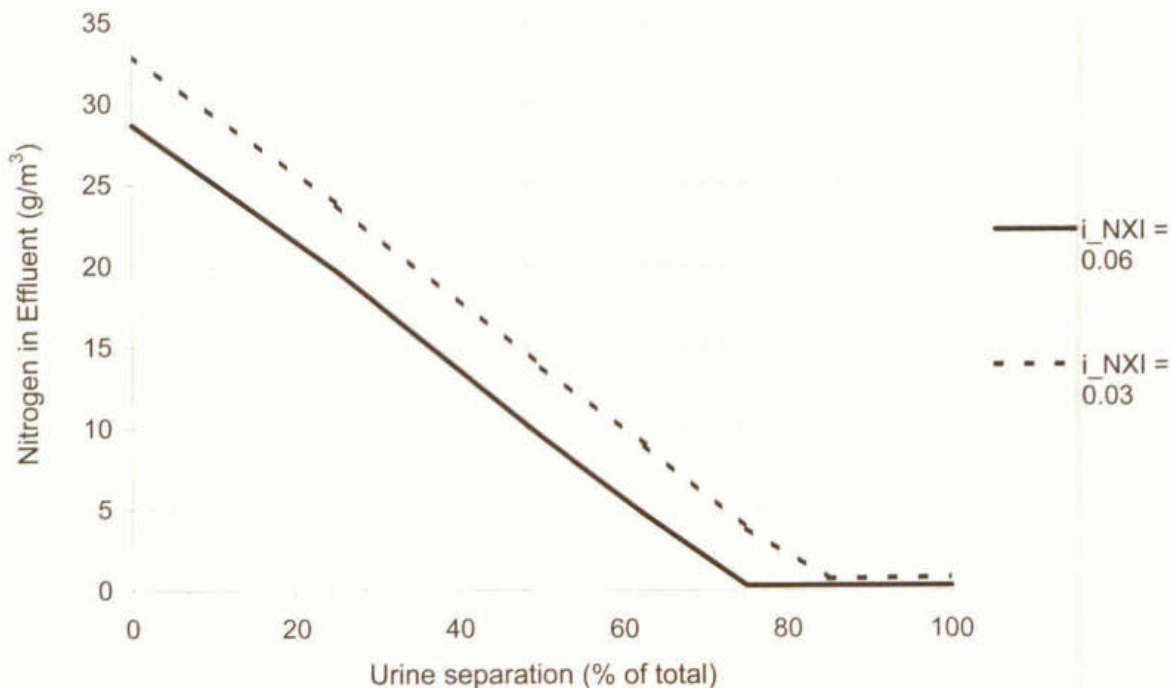
Figure 25:Effect of influent fractionation on nitrogen removal at different urine separation percentages (SRT = 2d and T = 12°C)

As expected, no biological excess phosphorus removal occurred in these simulations. Nitrogen removal with sludge production only (at around 50%) would have to include some form of chemical phosphorus removal.

6.11   Chemical phosphorus removal and urine separation

The disadvantages of chemical P-removal from wastewater include inefficient metal use, difficult sludge dewatering, low pH toxicity and metal leakage into effluent. The chemicals used for phosphorus removal include metal salts and lime. The effect of urine separation on the use of ferric chloride ($FeCl_3$) can be determined from the stoichiometry of phosphorus precipitation:

$Fe^{3+} + PO_4^{3-} \rightarrow FePO_4$

In many cases an unwanted by-product also forms:

$Fe^{3+} + 3\ OH^- \rightarrow Fe(OH)_3$

Based on these two equations, two moles Fe^{3+} are required for the precipitation of one mole P. However, in practice the ratio is somewhat higher and depends on the required level of phosphorus removal. For 95% P-removal, Tchobanoglous (1991) recommends a molar ratio of Fe:P = 3:1. Chemical pre-precipitation is also widely applied in the Netherlands. STOWA (1996) reported that on average, 15 $gFe^{3+}/m^3_{influent}$ is dosed for phosphorus precipitation. For a removal efficiency between 60% and 80%, the average molar ratio for metal dosing was Fe:P = 0.9:1.0.

Based on the influent data and nutrient concentrations in urine used in this study, and an assumed ratio of Fe:P = 1, table 13 illustrates the reduction in ferric chloride use with increasing urine separation. The masses presented in table 13 are based on the atomic mass of elements only. It was assumed that the precipitate consisted of 80% ferric phosphate and 20% ferric

hydroxide. Only dried matter was considered and no substances other than ferric phosphate or ferric hydroxide were taken into account.

Table 13: Effect of urine separation on phosphorus precipitation with ferric chloride

Urine separation (%)	0	10	25	50	75	100
kg PO_4-P/d_influent	108	104	97	86	69	63
kg Fe^{3+}/d	195	187	175	154	124	114
Chemical sludge dry matter, kg/d ($FePO_4$ and $FeOH_3$)	473	453	424	375	301	276

The reduction in ferric chloride and the mass of chemical sludge produced daily is linear with increasing urine separation. Although phosphorus would still have to be precipitated from urine (requiring some metal salt), the disadvantages of chemical P-removal from wastewater may be prevented

6.12 Conclusions and suggestions

1. With 50% of all urine collected separately, the BCFS® process requires between 20% and 30% less oxygen than in the zero separation scenario. Urine separation decrease the requirement for denitrification and more aeration energy could be saved through settling or pre-precipitation, without an increase in effluent concentrations.

2. The Dutch population is still increasing, leading to an increased wastewater production. If 50% or more of the urine from new urban developments were collected separately, grey water (or even grey- and black water) could be treated in existing wastewater treatment plants, without upgrading the biological reactor. However, it would probably require improvement of the hydraulic capacity of the treatment works.

3. Urine separation could reduce the total nitrogen concentration in the effluent of advanced wastewater treatment processes, such as the BCFS® processes. The improvement in effluent quality will be even more substantial in less advanced treatment processes. Urine separation could make very low effluent concentrations possible in future. Total nitrogen effluent concentrations of 2.2 gN/m^3 could be possible with settled or pre-precipitated wastewater where more than 50% of all urine was collected separately.

4. Advanced wastewater treatment processes (such as the BCFS® process) remove most nutrients very well and are not optimal to remove nutrients at more than 60% urine separation efficiency. The effluent concentration of nitrogen and ammonium would decrease greatly up to around 60% urine separation efficiency.

5. A reduction in influent nutrients could reduce the size of future wastewater treatment works. Complete nitrification requires sufficient sludge age and aeration capacity (both factors increasing reactor size). If all nutrients were consumed in cell metabolism, very small treatment works, with low sludge ages (0.8 - 2 days), could be possible. Simulation results suggest that with between 50% and 60% separate urine collection, a process could be operated to achieve $N_{tot\_eff} < 10$ gN/m^3.

7. RECOVERING MINERALS FROM WASTEWATER AND URINE

7.1 Introduction

Most of the inorganic nutrients present in wastewater, such as N, P, K, S, Mg, Ca, Fe, Na and Cl are required for microbial cell synthesis (Tchobanoglous, 1991). Waste activated sludge therefore presents a source of minerals, but it also contains heavy metals. Around 25% of the total heavy metal load in wastewater (including Cd, Cr, Cu, Hg, Ni, Pb, Zn and As) is discharged with the effluent and 75% is absorbed and/or adsorbed by the sludge mass (RIVM, 2001). These percentages differ greatly between individual treatment works. Accumulation of heavy metals could occur in the food chain if sludge were used as fertiliser. Preventive and expensive quality control procedures were implemented and consequently, most of the waste sludge is now incinerated. In 1998, 14 kt (as dried matter) was treated in composting facilities, 101 kt was dumped and 162 kt was incinerated in the Netherlands (RIVM, 2001). Researchers in Japan and Sweden found that it is technically possible to recover phosphorus from incinerated sludge ash (Matsuo, 1995; Karlsson, 2001; Hultman *et al*, 2001 and Jozuka, 2001). These techniques are not considered further in this report.

During anaerobic sludge digestion, considerable amounts of ammonia and phosphate are released. Liquor from various digested sludge de-watering processes therefore has a high mineral concentration. STOWA already evaluated techniques to remove or recover ammonia from concentrated wastewater (STOWA 1995-08, STOWA 1995-09, STOWA 1995-11, STOWA 1995-12 and STOWA 1996-01).

Due to increased pH and mineral concentrations, various phosphate salts easily precipitate in side stream processes and pipelines. This natural process could be used to recover phosphate under controlled conditions. Effluent from the anaerobic zone of UCT-type processes also contains high phosphate concentrations. This stream could be used for calcium phosphate or struvite recovery. A more detailed description of the "phosphate stripper" in the anaerobic zone of a modified UCT type process was given by Brandse and van Loosdrecht (2001).

This chapter gives a short account of existing techniques to recover ammonium and phosphate from aqueous waste streams. These techniques could be applied to develop processes for mineral removal and recovery from urine.

7.2 Recovery of ammonia from solution

7.2.1 Ion exchange and ammonia adsorption

Ion exchange is a technique where a specific ion (normally in low concentration) replaces another ion in a resin or mineral bed. In recovering ammonium, other cations are therefore always introduced into the effluent. The "Coupled columns" technique consists of three columns filled with acid resin, where counter currents of a regenerant (36% nitric acid) exchange ions with wastewater. Higher alkalinity (pH > 9) and higher influent ammonia concentration was found to increase the capacity of the resins and the ion exchange efficiency. Ammonia can be recovered from industrial wastewater, containing ammonia concentrations of between 0.34 kg/m^3 and 3.5 kg/m^3. An experimental set-up proved that this process could concentrate NH_4NO_3 to between 50 and 65 kgN/m^3 (Buday, 1994). This concentration, which is sufficient for utilisation in the production of granulated or liquid nitrogen fertilisers, had also been achieved in a full-scale plant.

However, nitric acid is normally produced by burning ammonia and it therefore seems more efficient to recover ammonia through more direct techniques.

Van der Gaag (1987) studied the phenomena of crystalline aluminosilicates (zeolites) acting as a "molecular sieves" to adsorb ammonia, air, hydrogen, carbon hydrogen sulphide and disulphide. Ammonium adsorption in zeolites is in fact a special case of ion exchange. Special types of zeolites can be synthesised to optimise the adsorption of ammonia.

Lind *et al* (1999) investigated the possibilities of recovering ammonia from "fresh" and synthetic urine. A combination of two processes, struvite crystallisation and ammonia adsorption, was developed. It was possible to recover the majority of nutrients in urine as solids. By adding MgO, the alkalinity increased to pH =10 and struvite precipitated as the main component, with smaller quantities of newberyite, brucite and epsonite. By mixing the remaining urine with natural zeolites (especially clinoptilolite), up to 80% of the total ammonia load was removed by adsorption. Ammonia-clinoptilolite mixed with apatite is a good slow release fertiliser. Struvite and ammonia-clinoptilolite may have the same fertilising properties, which is now being investigated by the same group. This combination of techniques could be very useful for recovering both nitrogen-ammonia and phosphorus from urine. A pilot installation is operating at an interactive science museum, the Universeum in Göteborg.

Ion exchange and adsorption are both techniques to concentrate an ammonium solution. It is in many cases used as a primary process step before stripping.

7.2.2 Ammonia volatilisation and stripping

Ammonia molecules in solution are always in equilibrium with ammonium and hydroxide ions. An increase in pH and/or temperature displace the equilibrium to the right (according to the equation below) and increase the evaporation of ammonia gas.

$$NH_4^+ + OH^- \quad NH_3 \uparrow + H_2O$$

Chapter 4 of this report gave a simplified description of urea hydrolysis and ammonia volatilisation. A more detailed description of ammonia volatilisation and evaporation and the dynamics in relation to pH and temperature is given in STOWA (1995-12). Ammonia volatilisation and evaporation are natural processes occurring in solutions with high concentrations. Shilton (1996), for example, investigated the significance of these mechanisms in ponds treating piggery wastewater and found that a large amount of ammonia (up to 8 $gN/m^3.d$) was removed by natural evaporation.

The process illustrated in figure 26 consists of a stripper column through which ammonium rich liquid flows against a counter air stream. Due to the increased surface area, ammonia is transferred from the liquid phase into the air. The spent air is then forced through an absorber where ammonia is dissolved in a strong sulphuric acid solution. The air was recycled back into the stripper, and the ammonia salt solution was drained from the absorber. Instead of air, steam can also be used in strippers. Ammonia is ten transferred from the liquid phase into the vapour. By condensing the steam/ammonia mix, a highly concentrated ammonia solution can be formed.
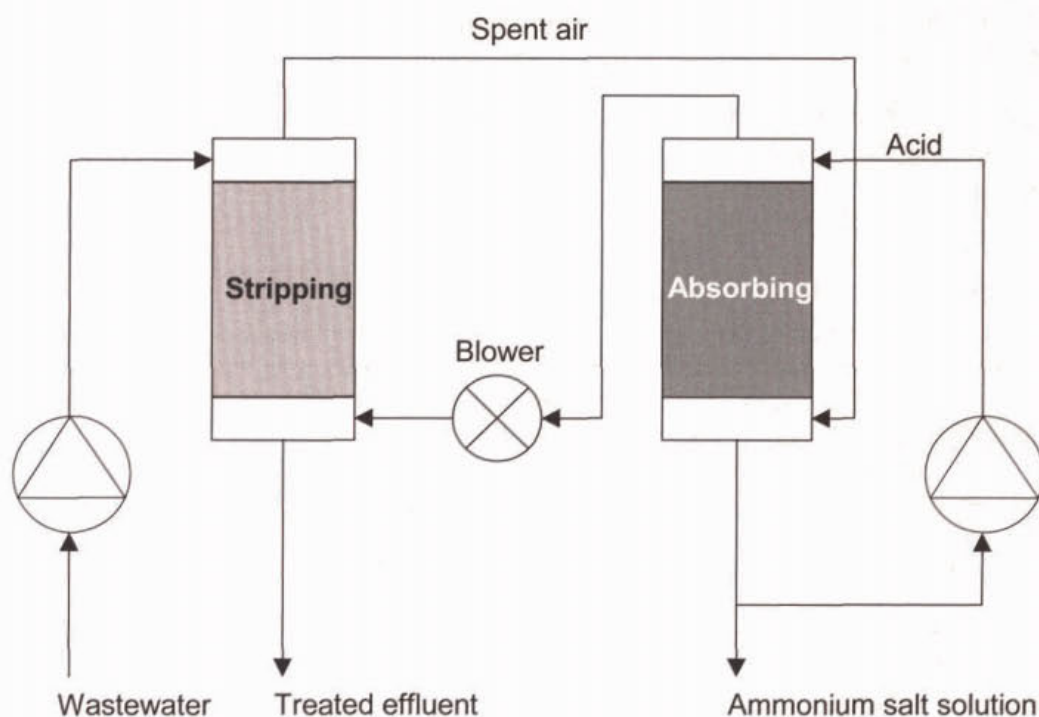
Figure 26 Recovering ammonia from solution by air stripping and acid absorption

Supernatant and centrate (the flow from centrifugation processes) from anaerobic digesters contain high concentrations of ammonium. Kathesis *et al* (1998) collected centrate from eight different wastewater treatment works, had average TKN concentrations varying from 560 gN/m^3 up to 1800 gN/m^3. The TKN concentration was as high as 3000 gN/m^3 in isolated cases. Ammonium accounted for around 75% of the TKN. The effects of temperature and pH on the effectiveness of ammonia stripping were evaluated. It was shown that for six different temperatures (between 23°C and 75°C) a pH value exists beyond which a further increase in pH has little effect on the removal efficiency. These pH values at 23°C and 75°C were 11 and 9.5 respectively. Ammonia removal efficiencies were 45% and 95% respectively.

A pilot steam stripping installation was built at the wastewater treatment plant Amsterdam-Oost (STOWA, 95-11). Ammonium was stripped from the sludge centrate at 92°C (after heating with steam) and pH = 11 (after addition of lime). The ammonia gas mix was condensed and rectified to yield a constant 25% ammonium-solution. An average removal efficiency of 87% was achieved, irrespective of concentrations between 500 and 2000 gN/m^3. The cost efficiency of ammonia stripping was found to increase with increased load (kgN/d). The work by Kathesis *et al* (1998) suggests that the same removal efficiency could have been achieved at much lower pH.

Different fillings and flow dividers can be used in stripping columns. Specific mass transfer coefficients depend on the type of filling, but at different loading rates, the effect that a specific type of filling has on mass transfer also differs. The relation between efficiency and the size of a stripping installation was investigated by Schomaker *et al* (1984). Apart from temperature, pH and ammonia concentration, the specific design of the installation also effects the efficiency of ammonia stripping.

The choice of a specific ammonia stripping process depends on the available resources or installations. For instance, where steam is already available from another process, steam stripping would probably be a favourable option. On the other hand, steam generation and stripping demand much more energy than air stripping. Temperature, concentration and pH of

the liquid would also determine the choice. Some form of hydroxide is usually required to raise the pH to around 9 or 11, depending on the liquid temperature. The relative amounts of required chemicals and energy in different stripping processes were summarised in (STOWA, 96-01). The cost of stripping installations is for a large extent determined by the capital investment. Small-scale ammonia stripping, operating at the same efficiency, could be expected to be much more expensive.

7.2.3 Transport of concentrated ammonium solution

The transport of urine through gravity sewers poses the problem of ammonia evaporation. If this occurs, not only would some recoverable ammonia be lost, but ammonia in the atmosphere can also be harmful. Ammonia also evaporates when separately collected urine is applied directly onto fields as fertiliser. Ammonia evaporation can be prevented by acidification to pH ≅ 3. However, acid addition could lead to other problems and does not offer a solution where the issue of sustainability is prevalent. Partial nitrification of urine can be used to generate acidity to stabilise and preserve ammonium for further treatment or re-use. Oxidation of ammonium to nitrite is an acidifying process, as shown in the reaction equation below:

$$2NH_4^+ + 3O_2^- \rightarrow 2NO_2^- + 2H_2O + 4H^+$$

This reaction occurs in a suspended growth process at pH = 6. In a subsequent attached growth process, the pH is lowered to around pH = 3. After the two process steps, 50% of the nitrogen is oxidised to NO_3^- while 50% remain in the form of NH_4^+. The ions contained in this liquid are stable and suitable for further handling, transport and possible recovery of ammonia or nitrate. Another advantage is that phosphate precipitation does not occur at such a low pH, limiting the accumulation of insoluble phosphate salts on pipe walls.

The product could be used directly as fertiliser, on condition of meeting health and safety requirements related to micro-pollutants in urine.

7.3   Recovery of phosphate from solution

7.3.1 Alternative phosphate removal processes

Phosphate precipitation with iron or aluminium generally excludes recycling. Iron or aluminium phosphates are incompatible with the current technology of the phosphate industry and would require excessive energy input to recover phosphorus.

Alternative processes have been developed to remove phosphate from wastewater, including the DHV Crystalactor®, the CSIR fluidised bed reactor, the Kurita fixed bed crystallisation column, the RIM-NUT ion exchange system, the Unitika Phosnix and the Sydney Water Board reactor (Brett *et al*, 1997). Most of these processes operate according to the same basic principles. A phosphate solution is injected into a fluidised bed reactor with seeding material (mostly sand grains). Chemicals are added to increase the pH and provide cations so that either calcium phosphate or magnesium ammonium phosphate (*struvite*) precipitates. The influent phosphate concentration for most of these processes varies between 10 gP/m^3 and 100 gP/m^3. The solid product from these processes contain between 10% – 15% P/dried mass. A summary of important publications regarding phosphorus recovery from wastewater appeared in a special edition of the SCOPE newsletter, number 41. The following sections present a summary of basic phosphate precipitation kinetics.

7.3.2 Calcium phosphate crystallisation

Calcium phosphate precipitates from solution under certain conditions. Solubility is a measure of ion concentrations in a liquid. When an ideal solution becomes saturated with its component ions, precipitation can commence. Wastewater has significant ion associations (due to high ionic strength) and the free ion concentrations contributing to saturation will be lower than the total ion concentration. Supersauration of the liquid is necessary for precipitation to occur and the measure of supersaturation accounts for the efficiency of precipitation (Song *et al*, 2001). Koutsoukos (2001) described supersaturation as the quotient of the ion activity product and thermodynamic solubility product. Calcium phosphate crystals would grow in a supersaturated solution only when the newly formed nuclei exceed a critical size. Homogenous nucleation occurs when individual crystallites grow spontaneously from solution. Heterogeneous nucleation describes the process whereby crystallites aggregate on existing surfaces. Heterogeneous nucleation requires less energy and is more likely in wastewater treatment environments with large surface areas, such as suspended particles or pipe walls.

Calcium phosphate seems to deposit in amorphous structures around seed materials, rather than in crystals. Crystal growth (following initial nucleation) takes place in several steps, of which neither the chemical nor the physical parameters are fully understood. The chemistry is made more complex because calcium phosphate is not one molecule but potentially a number of different compounds with different hydration complexes (calcium hydroxylapatite, dicalcium phosphate dihydrate, octacalcium phosphate, and tricalcium phosphate). Kinetic forces determine the nature of the solid deposits formed during the precipitation process, more so than equilibrium considerations. The most stable thermodynamic condition may therefore not be reached immediately. Less stable phases may precipitate initially, precipitation may not happen at all, or not until an appropriate "trigger" (e.g. seeding) becomes available. A solution departing form equilibrium (by a change in pH, temperature or concentration) is bound to return to this state by precipitation of the excess solute. The influence of temperature and pH on hydroxylapatite precipitation was investigated by Zoltek (1974) who demonstrated that supersaturation may be achieved by simply increasing the pH. Song *et al* (2001) found experimentally that both an increase in the pH and an increase in the Ca/P ratio, favour the precipitation efficiency and precipitation rate of calcium phosphate. However, the precipitated phase will most likely eventually transform into the more stable hydroxylapatite (Kibalczyc, 1989). However, for phosphate salt solutions, supersaturated solutions may be stable for practically infinite periods. The rate of crystallisation in pure calcium phosphate solutions was described empirically by Koutsoukos (2001):

$$R_g = k_g\, f(S)\, \sigma^n$$

Where, k_g = crystallisation rate constant, $f(S)$ = function of the number of available crystal growth sites, σ = relative super-saturation and n = apparent order of crystal growth process.

In wastewater, the calcium phosphate precipitation rate also depends on the concentrations of phosphate, calcium, the pH, the temperature and the presence of other ions (organic and inorganic). Impurities such as inorganic ions and ionised organic material reduce the growth rate of crystals. A number of ions, such as carbonate and magnesium, are known to act as inhibitors to calcium phosphate precipitation by forming a surface complex on the crystal and blocking further precipitation (Abbona *et al*, 1990). Natural amino acids with hydrophobic non-polar groups inhibit calcium phosphate growth, blocking active growth sites, and relates to the adsorption affinity of each amino acid to the crystal surface (Koutsopoulos *et al*, 2000). In an experiment by Koutsoukos (2001), the rate of calcium phosphate precipitation was studied in the presence and in the absence of ionised organic molecules. Even at low concentrations of organic ions, the crystal growth was limited due to blocking of active growth sites.

The Geestmerambacht wastewater treatment works include a DHV Crystalactor® process to remove phosphate from the anaerobic zone's return flow (figure 27). Phosphate-rich water is pumped up through a fluidised bed of sand grains. The presence of carbonate requires a de-gassifier to remove CO_2 with addition of H_2SO_4, reducing the pH to around 3. Efficient calcium phosphate crystallisation requires pH $\cong$ 9, which necessitates NaOH addition. By adding milk of lime ($Ca(OH)_2$) and controlling alkalinity, calcium phosphate crystallises on nuclei. Crystals that reach a certain size, settle in the reactor and can be removed. The dried product contains almost no water and is virtually identical to rock apatite. Three of these installations have been built in the Netherlands. The first Crystalactor® installation was a pilot plant at the Westerbork wastewater treatment plant (12 000 pe). It treated wastewater effluent where the influent had a phosphorus concentration of around 20 gP/m^3. Discontinuation of phosphate addition to detergents has been blamed for the closure of this installation.

The second installation was built at Heemstede (35,000 pe) and the third at Geesmerambacht (230,000 pe) which has been operational since 1994. The calcium phosphate Crystalactor® plant at Geestmerambacht produces 200 – 300 tonnes/year of phosphate pellets (13% P).

The Crystalactor® process has not become popular due to carbon dioxide stripping, the high operational pH to achieve precipitation, complexity of the process, the overdosing of calcium ions and investment costs is still not favourable. More research topics regarding calcium phosphate precipitation were summarised by Valsami-Jones (2001). Reactor design has all been centred on fluidised bed reactors, but it is not clear whether or why this is the most efficient technique of phosphate precipitation.

Figure 27 DHV Crystalactor® at Geestmerambacht

7.3.3   Struvite crystallisation

Struvite (magnesium ammonium phosphate) was discovered in Hamburg in 1846. In contrast to calcium phosphate, struvite precipitates naturally in wastewater treatment installations. In a solution where both Mg and Ca ions are present, nearly all phosphate will precipitate as struvite (WR Grace & Co, 1969). This is mostly true for wastewater and leads to problems such as scaling and pipe blockages. Rawn (1939) discovered struvite crystals in supernatant pipelines. Borgerding (1972) described struvite crystal growth in a pipeline reducing the diameter from twelve to six inches. Many other wastewater treatment works experienced the same problem (Mohajit et al, 1989, Pitman et al, 1991, Webb and Ho, 1992, Mamais et al, 1994). The Slough wastewater treatment plant in England experienced problems with struvite scaling in centrifugal pumps used for centrate pumping. Pumps had to be decommissioned regularly to chisel and chip away struvite scales (Williams, 1999). Struvite has never been a common problem in Dutch wastewater treatment works due to differences in the pH and chemical constitution of wastewater. However, accumulated struvite crystals have been found during excavations in Amsterdam. These struvite deposits date from the 13th century, where it formed in pit latrines (Wevers et al, 1981 and Kars et al, 1983). According to Schuiling and Anrade (1999), "if nature can do it, we can also try it."

Struvite crystallises according to the following reaction:

$$Mg^{2+} + NH_4^+ + HPO_4^{2-} + 6H_2O \rightarrow MgNH_4PO_4.6(H_2O) + H^+$$

The precipitation is analogous to that of calcium phosphate. Crystal growth follows the formation of small nuclei when the solubility product of the solution is exceeded. An important difference from calcium phosphate is that struvite nucleation usually occurs homogeneously. Struvite precipitation is related to the solution's pH, temperature supersaturation and the presence of other ions (such as calcium). Momberg and Oellerman (1992) showed that the nucleation rate was reduced from a few days at pH = 7 to just 1 hour at pH = 8.5. The increase in pH is normally done with addition of chemicals $Mg(OH)_2$ and/or NaOH. The level of supersaturation (of phosphate, magnesium and ammonium) effects both nucleation and crystallisation rates. Ohlinger *et al* (1998) showed that the rate of struvite scale accumulation increased from 6 g/m^2.d to 48 g/m^2.d when the level of supersaturation was doubled. Ohlinger *et al* (1999) also showed a linear relation between mixing rate and rate of struvite crystallisation.

A few wastewater treatment works in Japan recover struvite from different streams. Ueno and Fujii (2001) summarised the operation of the Unitika process three years after commissioning. In this process, dewatering liquid from anaerobic sludge digestion is fed into a fluidised bed reactor with $Mg(OH)_2$ in a Mg:P ratio of 1:1 and NaOH to increase the pH to between 8.2 and 8.8. The influent phosphate concentration is around 44 gP/m^3 and after struvite is precipitated, the effluent concentration is around 5 gP/m^3. The struvite has a phosphate content of 30% (P_2O_5) and around 500 kg/d of struvite is produced on average. A similar process is used by Kurita Water Industries (Ishiduka *et al* 1998). Changes to this process were described by Mitani *et al* (2001) to recover struvite from lower concentrations. One of the main differences is that the hydraulic retention time in the fluidised bed reactor must be increased to allow sufficient crystal growth. Seed material was also required to form struvite particles. Instead of Mg(OH)2, the more soluble MgCl was used for addition of magnesium ions. Addition of chlorine increases the salinity of water, which should be prevented where possible.

The Brisbane city council evaluated struvite recovery from side streams as an option to decrease effluent phosphate concentration. The struvite grain size was proportional to the influent phosphorus concentration and the struvite particle retention time in the reactor based on the Unitika process (von Münch and Barr, 2001).

It has been shown that struvite can be precipitated under controlled conditions without the addition of any chemicals. Battistoni *et al* (1997) removed 80% of the phosphorus form dewatering liquid with a high Mg:P ratio of 3.7:1. The pH was risen to around 8.5 by ageing the liquor and removing CO_2 by air stripping. This work was further developed and a full-scale fluidised bed reactor is now operating in Treviso, Italy. Up to 85% P-removal form dewatering liquid is attained in an 85 000 pe wastewater treatment works, using only aeration to increase the pH (Battistoni *et al* 2000).

The continuously stirred tank reactor at Putten produces potassium struvite ($MgKPO_4.6H_2O$) from calf manure. The first step of the treatment process is conventional biological nutrient removal. The liquid calf manure is not diluted as with municipal wastewater and after nitrification and denitrification, the effluent contains little ammonia, but still around 300 gP/m^3. After settling, the phosphate rich effluent is treated with magnesia (MgO) to form struvite in a series of three tanks (figure 28). Mechanical

Figure 28 Struvite stirred tank reactors at Putten

mixers keep solids in suspension, mix the liquor to bring ions in contact with nuclei and dissolve MgO, which is rather poorly soluble. The dissolving MgO increases alkalinity to pH = 8.5 - 9, where struvite starts to precipitate (Schuiling and Andrade, 1999). The number and size of the tanks were determined based on laboratory experiments. However, in the prototype installation, dephosphatation takes only 30 minutes to be essentially complete, making the last two tanks almost obsolete (Verhoek, 2001). The clarified effluent contains between 15 gP/m^3 and 30 gP/m^3 (95% removal efficiency). Approximately 125 kg P is recovered per day This is one of the only techniques to recover potassium. Although potassium removal or recovery is not a priority in wastewater treatment, it is a valuable fertiliser component.

Fittschen and Hahn (1998) stated that liquid cattle manure and human urine show different chemical compositions. However, chapter 2 of this report showed that urine contains all the ions required for struvite precipitation. If potassium struvite is to be precipitated from urine (after say, ammonia stripping), the average molar ratios of these ions in urine are Mg:K:P = 5:44:18. The amount of potassium is sufficient, but magnesium would have to be added to remove all phosphate. Furthermore, struvite is a common form of kidney stone. Infection of the urinary tract with urea hydrolysing organisms increases the ammonia concentration and pH of urine, which leads to dissociation of phosphate and the supersaturation and precipitation of struvite (Griffith, 1978 and Pak, 1998). Reactions occurring naturally in the human body could be repeated outside the body. In a feasibility evaluation for phosphorus recovery, Woods et al (1999) concluded, "phosphorus recovery is most viable in cases with high phosphorus concentration, low BOD concentrations and high sludge handling costs." Urine has all the attributes required for struvite recovery from liquid waste.

7.3.4 Miscellaneous research on phosphorus recovery

A number of alternative recovery techniques were listed by Brett et al (1997). Recovery of phosphate as red mud, through filters, with activated alumina, half burned dolomite and other techniques are discussed shortly. The internet site of the Natural History Museum in London keeps track of some developments the recovery and recycling of phosphates from sewage and animal wastes[1.23].

7.4 Conclusions

- Ammonia can be recovered from solution by a number of techniques. A promising technique is ammonia stripping with either air or steam. Higher ammonia concentrations, liquid pH and temperature lead to more efficient stripping. Ion exchange reactions could be used to recover ammonia, but some other mineral is also introduced (and lost). Moreover, the ammonia product has a lower ammonia concentration.

- Phosphate can be recovered from solution as various precipitates. The fundamental physical/chemical reaction conditions of phosphate crystallisation, such as solubility phases of different phosphates, supersaturation conditions and presence of other materials, are not yet well understood and an area for further research.

- Calcium phosphate can be recovered from wastewater. Installations to recover calcium phosphate are expensive (high investment cost) and the complex process makes operation and maintenance costs high. Prerequisites for good calcium phosphate precipitation are supersaturation of calcium and phosphate, a seed material to trigger nucleation and absence of other ions. Therefore, a complex systems is necessary where pH is lowered to around 3 (to remove carbon dioxide) and then increased to around 9 (to cause supersaturation). The energy demand of keeping the seeding material in suspension is also quite high. The capital cost, chemical dosing and energy demand makes recovered calcium phosphate an expensive raw material.

- Magnesium ammonium/potassium phosphate (struvite) precipitation is a simple process in comparison to calcium phosphate precipitation. Struvite is a form of kidney stone and occurs naturally in supernatant pipelines. Controlled struvite recovery from digester supernatant, anaerobic return flow and from urine has been done successfully. Although MgO is less soluble than MgCl, dissolved MgO increases the pH, while the use of MgCl requires additional NaOH. This will not only increase the pH, but will also increase the effluent salinity.

- Experience with existing installations shows that higher mineral concentrations make recovery more efficient and reduce costs such as energy requirements and chemical dosing.

- Urine is a concentrated mineral solution, even when compared to some industrial waste streams. The volume of liquid to be treated is therefore only a fraction of most other waste liquid with the same mineral load. The requirement for storage tanks, piping installations, pumping capacity and treatment costs could therefore be less than the requirements for mineral recovery from supernatant, for instance.

8. EVALUATING THE RECOVERY AND RECYCLING OF MINERALS FROM WASTE PRODUCTS

8.1 Introduction

This chapter briefly describes industrial processes and compares the energy requirement to that of recovery processes. Where more energy is consumed in mineral recovery than in mineral production and refinement from renewable sources, the recovery can certainly not be considered sustainable. In the general introduction to this report, it was argued that recovery of minerals is a prerequisite for sustainability. The previous chapter showed that mineral recovery from wastewater is technically possible. Furthermore, it seemed that an inverse relationship exists between the concentration of the minerals and the amount of energy required in the recovery process. The recovery of minerals (finite resources) can only be realised by consumption of energy (another finite resource).

The total energy of any system and its surroundings is always conserved. Energy can neither be created nor destroyed, but can only change from one form to another. The change in Gibbs free energy (dG) equals the heat flow (dH) minus the product of the absolute temperature of the system (T) and the change in entropy (dS): $dG = dH - T.dS$. Furthermore, the second law of thermodynamics states *"The entropy change for any system and its surroundings, is positive and approaches a limiting value of zero for any process that approaches reversibility."* This means dS > 0. As the entropy of a system increases, the amount of useful energy decreases. Although energy can not be lost, the quality of energy can be lost. The quality of energy can be measured as the capacity for doing work. Szargut *et al* (1988) explains the concept and application of exergy as the capacity for doing work. Materials (chemical compounds) also contain exergy. *"Exergy is the amount of work obtainable when some matter is brought to a state of thermodynamic equilibrium with the common components of the natural surroundings by means of a reversible process."* The exergy contained in minerals represents millenia's stored solar energy that shaped the earth and formed complex chemical compounds.

The most common and least reactive elements and substances in the atmosphere, the sea and the earth's crust are defined as the reference state. Each other element or substance then has an exergy value related to that reference state. For instance, the following reference species and reactions determine the standard chemical exergy of carbon and phosphorus:

$$C + O_2 \rightarrow CO_2$$

$$P + \frac{5}{4}O_2 + \frac{3}{2}CaCO_3 \rightarrow \frac{1}{2}Ca_3(PO_4)_2 + \frac{3}{2}CO_2$$
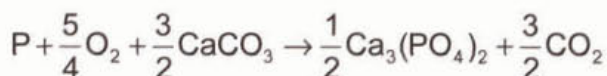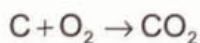
In this framework, the standard chemical exergy of pure carbon is 410 kJ/mol and of pure phosphorus 863kJ/mol. The standard chemical exergy of these compounds in their reference state (CO_2 and $Ca_3(PO_4)_2$) is zero. The reference state of nitrogen is N_2 (0 kJ/mol), while the standard chemical exergy of ammonia is 338 kJ/mol (20 kJ/gNH_3). The practical implication of these definitions is that their standard chemical exergy is the amount of shaft work or electrical energy necessary to produce a specific substance from materials with zero exergy. It is therefore a natural fact that ammonia can not be produced with less than 20 kJ/gNH_3 in any way. Where minerals are removed from wastewater and disposed of, an amount of useful energy is therefore lost. Recovery of minerals also represents a recovery of energy.

Dilution of minerals (e.g. due to flushing water or rain water) also destroys some exergy, as energy would be required to reconcentrate the minerals. The Gibb's free energy of mixing is represented by the following equation:

$$G_b = RT \sum \frac{N_i}{N} \ln \frac{N_i}{N}$$

Where R = the universal constant (8.314 J/mol.K), T = temperature (298 K), N_i = the number of moles of one of the substances, either ammonia or water, N = the total number of moles.

This amount of energy is destroyed upon diluting, which is an irreversible process. However, this loss is insignificant in comparison to the energy consumption of industrial ammonia production. The effect of losses occurring during dilution has to be evaluated with the energy demand of specific processes to re-concentrate the substance.

The potential economic value of recovered minerals is also important. The sentiment is usually not only towards "sustainability", but more often cost minimisation is essential. It could be argued that high economic costs of recovery processes represents a "hidden energy consumption", because all economic activity is sustained by energy consumption. However, there are no good conversion factors between energy and economic cost available from literature at present.

8.2   Ammonia production and recovery

8.2.1   Ammonia production

Approximately 80% of earth's atmosphere consist of nitrogen. Nitrogen (N_2) is an inert gas and by far the most abundant form of nitrogen. Other compounds are NO_2, N_2O, NO_3 and NH_3. Nitrogen is used in the industrial production of ammonia. The Haber process for ammonia synthesis is the most effective and widely used. In this process, natural gas (methane, CH_4) reacts with steam to produce carbon dioxide and hydrogen gas (H_2). Ammonia gas is then synthesised by the reaction of hydrogen gas with nitrogen gas under pressures ranging from 10 MPa to 25 MPa. Current energy consumption is between 32 and 38 kJ/g NH_3. The efficiency of the process has increased over the past 25 years and may soon be around 28 kJ/g NH_3. A theoretical minimum for energy required in production is around 25 kJ/gNH_3, based on the high heat value of ammonia, and assuming no energy export and no energy import except for feedstock and fuel (UNEP, 1998). This amount isn't much higher than the standard chemical exergy (20 kJ/gNH_3).

Liquefied ammonia is used as a nitrogen fertiliser. Ammonia is also used in the production of urea, $CO(NH_2)_2$, which is used as a fertiliser, in the plastic industry, and in the livestock industry as a feed supplement. Ammonia is often the starting compound for many other substances, such as ammonium nitrate, ammonium sulphate and ammonium phosphate.

Hydro Agri Sluiskil BV is the most important ammonia producer in the Netherlands. Out of the total 3.5 million ton ammonia produced in Europe in 1997, 1.5 million ton were produced at Hydro Agri Sluiskil[i.24]. The yearly natural gas consumption at Hydro Agri Sluiskil is around 2 billion m^3, which is five percent of the total gas consumption in the Netherlands. Other important ammonia producers in the Netherlands include DSM Agro[i.25], Rotem Amfert[i.26] and Kemira[i.27]. The European ammonia market is saturated and competition is aimed at cost control and improved energy efficiency, which is difficult, because ammonia production has almost reached its theoretical maximum efficiency.

8.2.2 Ammonia removal and recovery

Nitrogen in wastewater is predominantly present as ammonia and ammonium. The ammonia industry has very little (if any) incentive to recycle ammonia, as their core business is centred on production. Moreover, the economic margins are small and only large-scale processes are economically viable.

The cost of anhydrous ammonia is around 350 €/t. Ammonium nitrate fertiliser (34% N) costs around 200 €/t. and ammonium sulphate costs around 250 €/t (21% N and 24% S). The actual production cost of ammonia is only around 150 €/t (100% NH_3) [i.28].

Conventional biological nitrogen removal (Refer chapter 6) first consumes energy to oxidise ammonia to nitrate (nitrification) and then reduces nitrate to nitrogen gas (denitrification). STOWA investigated alternative techniques to remove or recover nitrogen from wastewater side streams with high ammonia concentrations (STOWA 95-08, 95-09, 95-11, and 95-12). These techniques (hot and cold air stripping, steam stripping with spray evaporation or a steam generator, struvite recovery, an airlift reactor and a membrane bio-reactor) were compared with the Sharon process (Single reactor for High activity Ammonia Removal Over Nitrate) based on their energy consumption, chemical consumption and operational costs STOWA (96-01). Stripping renders a high ammonia yield (95% recovery). The average cost of ammonia stripping, comprising not only operational costs (energy and chemical consumption), but also the payment of capital investment, was estimated at around 8 €/kg N recovered. The cost range for steam stripping is between 4.50 €/kg N and 11 €/kg N, depending only on the size of the installation and assuming a concentration of 1 000 gNH_4-N/m^3 (STOWA 95-11). The energy consumption of the cold air stripping process could be between 6 and 26 kJ/gN removed. Energy consumption in steam stripping is between 34 and 60 kJ/gN. Processes with a lower energy requirement generally require larger amounts of chemicals, e.g. as with ammonia recovery as struvite, where phosphoric acid is added. Given that phosphorus is one of the minerals to be recovered from wastewater, this does not seem sensible.

The Sharon process requires more or less the same amount of energy per kg N removed as the most energy efficient stripping processes. In comparison to stripping, much less chemicals are used in the Sharon process and the operational costs is around 2 €/kg N removed. The energy consumption of the combined Sharon/Anammox process is in the order of 4 kJ/gN removed. The operational costs of the combined Sharon/Anammox process was estimated between 0.70 and 1.00 €/kgN removed (STOWA 2000-25). Figure 29 compares direct recovery of ammonia via stripping with ammonium removal via Sharon/Anammox and industrial production. Although direct recovery consumes 30% less energy than ammonium removal followed by industrial production, direct recovery is eight times as expensive.

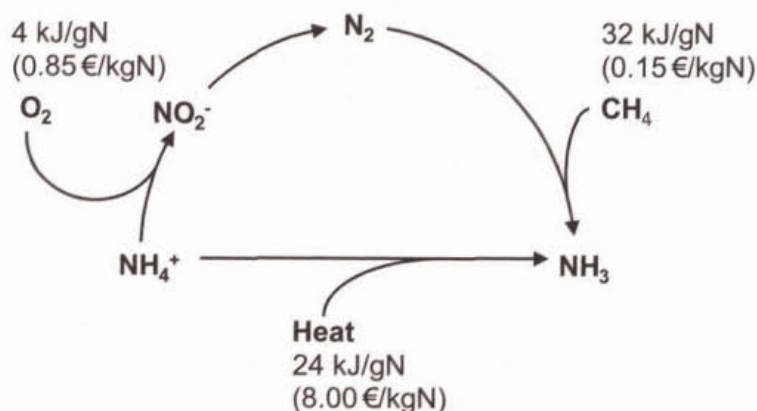
Figure 29: Ammonia recovery: energy and economic costs[12]

---

[12] The cost and energy consumption of direct ammonia recovery through steam stripping were based on a typical concentration (1 000 gNH_4-N/m^3) and capacity (250 000 people). The energy efficiency of the Sharon/Anammox process is insensitive to changes in concentration, although too high ammonium concentrations are toxic.

The cost of direct ammonia recovery is high due to the limited production capacity that still require the same operation, control, safety measures, etc. as large scale processes. Most processes to *recover* ammonia also imply considerable investment costs. Investment costs for new *removal* processes are relatively low, compared to that of stripping installations.

8.3   Phosphorus production and recovery

8.3.1   *Industrial phosphorus production*

Phosphorus is the eleventh most common element in the lithosphere, but concentrated natural phosphate resources are finite. Phosphates are found in sedimentary, igneous and metamorphic rocks. Commercial phosphates are mostly produced from sedimentary apatite that occurs as different composite minerals, namely fluorapatite $Ca_5F(PO_4)_3$, chloroapatite $Ca_5Cl(PO_4)_3$ and hydroxylapatite $Ca_5(OH)(PO_4)_3$. The mechanism for this sedimentary rock formation is generally believed to be the deposition of phosphate with the movement of ocean currents over geologic time periods (Sheldon, 1964). Sedimentary apatite exists as pellets, nodules, grains, replaced calcium in animal skeletal remains and animal excrement. Good quality apatite is more scarce than phosphate in general. Due to the relative reactivity of phosphorus, many of the ores are also rich in arsenic and other heavy metals, such as cadmium (Cd) and uranium (U). The risks associated with heavy metals and radioactive materials will restrict future phosphate production.

Apatite is mostly mined from open pit mines. The USA, Russia and Morocco lead the apatite mining industry. Global annual phosphate production is around 40 million ton (18 million ton P), originating from roughly 140 million ton appetite [i.29]. The phosphorus production industry is divided into two main sectors, as summarised below.

"Wet acid" process

Crude phosphoric acid (also known as green acid) is produced by mixing crushed apatite rocks with sulphuric acid. This is an exothermic reaction and phosphate production requires no additional energy. The green acid is further purified by filtration before the final production of various phosphate salts. The reaction between phosphate rock and sulphuric acid is self-limiting because an insoluble layer of calcium sulphate forms on the surface of the rock particles. Phosphoric acid is therefore re-circulated to convert as much rock as possible to the soluble mono calcium phosphate and then precipitating calcium sulphate with sulphuric acid. One of the by-products removed in the filtration process is calcium sulphate, or phosphor-gypsum. Around 1,5 ton of gypsum is produced per ton appetite used. The gypsum has little economical value and is usually dumped on stockpiles. Approximately 85% of all mined phosphate rock is consumed in this sector and primarily used to produce fertiliser (Driver, 2000). All "wet-acid" production plants in the Netherlands were closed due to increasing environmental control and stagnation in fertiliser demand.

Thermal process

In the thermal process, hydroxylapatite is granulated and mixed with a clay slurry to form round pellets. The pellets are dried, sintered at $800°C$ and then mixed with cokes (reducing agent) and pebbles (SiO_2, for slag formation). This mixture is fed into a furnace, heated to $1500°C$ by three carbon electrodes. Phosphate is reduced to P_4, leaving the furnace as a gas, together with the by-products CO and some dust. The dust (including volatile metals such as zinc) is removed in an electrostatic precipitator. The CO gas is used as a fuel to produce heat for sintering. The P_4 is diluted in water. The thermal production process is energy intensive and requires around 50 kJ/gP produced [i.29]. The high purity phosphorus or phosphoric acid produced in this way is used

in foods, detergents and as additive in many applications (medicine, crop protection agents, paper, cosmetics, diesel oil, lubricants, flame retardant). Thermphos BV is situated in Vlissingen (the Netherlands) and is the only factory of its kind in Europe, importing apatite rock and exporting high grade phosphoric acid.

8.3.2 Phosphate removal and recovery

In Europe, both sectors of the phosphate producing industry are now exploring the recycling potential of phosphate. By the year 2010, the European phosphate industry wants to recover 25 percent of its annual phosphate rock supply from secondary sources, such as recovered minerals from wastewater and manure. Thermphos, who is currently producing approximately 80 ktP/year[i.30], has set itself the target of replacing 15 – 16kt of phosphorus (20% of the total production) by secondary materials, within 5 years.

Under certain conditions, reclaimed phosphates can be mixed with mined phosphate rock for further processing. Although both production processes are robust, there are some strict quality requirements for raw materials. The requirements are more or less the same for both processes. Some constraints on the use of recovered phosphates are summarised below (Schipper, 2000 and Driver, 2000):

1. Phosphate content
   The typical phosphorus content in rock apatite is between 13 and 17% P. If reclaimed materials with a lower P content is used, a higher percentage of slag containing heat will be produced, decreasing the energy efficiency. The lower the phosphate content the more phospho-gypsum is produced in the wet acid process. Aluminium compounds can replace SiO_2 addition in slag formation. Aluminium phosphates would therefore be a good alternative raw material.

2. Impurities
   The phosphorus furnace also reduces all elements that are more easily reduced than phosphorus. The most unwanted elements are mentioned below:
   Zinc (and other volatile metals such as lead, cadmium and tin) is volatilised and precipitated on furnace dust and recycled to the slurry mixture where it builds up.
   Chloride causes corrosion damage to the sintering plant where metal is exposed to raw materials at high temperatures.
   Iron forms a separate slag that removes phosphorus from the process.
   Copper binds to the iron phosphorus and a significant increase in copper content may limit its application potential.
   Approximately 2% iron is the maximum amount which can be dealt with in the "wet acid" phosphate industry.

3. Water content
   The thermal process operates with a supply of dry material. Technical limitations to the use of "wet" materials include mixing, milling, interference with internal sludge processing capacity, etc. The drying of materials on-site would be costly and implies that materials with a high water content must be transported. In the wet acid process, water is disposed with the gypsum, where it could lead to problems of leaching or even sink holes.

4. Organic material and pellet strength
   Significant amounts of organic material decrease the strength of pellets used in the thermal process. The reclaimed material should be limited to a maximum of 5% carbon.

5. Nitrogen and ammonia content
   Ammonia in reclaimed materials would cause gas scrubbing problems in the thermal process. In addition, ammonia will be volatilised and oxidised to nitrous/nitric oxides.

From the phosphate industry's point of view, recovered aluminium or calcium phosphate is ideal for further processing. Recovered calcium phosphate is in most respects indistinguishable from mined apatite. Thermphos recycles some recovered calcium phosphate from the Crystalactor® at Geestmerambacht, which is, according to them, the only technology suitable to yield good quality secondary products. Samples of the recovered materials were pellet-like solids which drain readily to below 5-10% water and can offer 5-15% phosphorus content[i.31].

However, it has not yet been conclusively proven that the income from selling calcium phosphate could pay off the investment in crystallising equipment. The disuse of phosphates in household detergents has been blamed for the low success economic disappointment of the granular sludge bed crystalliser. The process is expensive due to its general complexity and the addition of chemicals. Furthermore, apatite has a rather low market value with prices varying around €40/ton (for good quality rock). Without a subsidy, economic viability of calcium phosphate recovery would be unlikely.

Struvite recovery can be much cheaper, but the phosphate industry finds struvite less attractive. There is no prospect of recycling struvite in a traditional wet acid route (Driver, 2000). The presence of ammonia in struvite would lead to problems of corrosion and NOx scrubbing and excludes the recycling of struvite in the thermal process. The recycling of potassium struvite (produced at Putten for instance) may be possible in the thermal process (very low ammonia content, high phosphate content), but it is uncertain whether this could in fact realise.

Struvite in itself is a valuable slow release fertiliser, considering its phosphate, magnesium, ammonium and/or potassium content. In the 1960's, the large-scale industrial production of struvite fertiliser was evaluated. Further interest and development ceased due to high production costs and little demand.

In Göteborg, struvite from a pilot plant is tested as fertiliser for municipal gardens and parks (Lind et al, 2000). Brisbane Water plans to precipitate struvite from return liquors that could be sold to golf courses. The process to be employed by Brisbane Water is based on the Japanese Unitika process. Struvite has a commercial value of between € 200/t and € 300/t, depending on the region and use. The price of magnesium hydroxide to be added to the process could be as high as € 350/t. Other operating costs are negligible (Von Munch et al, 2001). The dosing ration of magnesium is around 5g magnesium hydroxide to 8g struvite produced. In this case, a sales price of around € 225/t for struvite would be the break-even point. The Shimane wastewater treatment plant produces struvite from low phosphorus concentrations (Unitika process) and sells the product to fertiliser companies who then mix it with other organic and inorganic materials. It is suspected to be an expensive process due to its complexity. Recovered struvite is currently sold to the fertiliser industry at around € 300/ton (13.5% P) who markets it as fertiliser for rice farming (Ueno et al, 2001).

The "Mestvereniging Gelderland" (a provincial association treating animal manure) produces struvite in a much simpler process at Putten (Schuiling 1999 and Verhoek, 2000). Presently farmers can collect the struvite (used as fertiliser) free of charge, because it constitutes a considerable saving in sludge treatment.

Urine is the most concentrated and largest source of phosphorus in municipal wastewater, but still contributes only around 45% of the total phosphorus load in wastewater. If urine were collected separately for reasons related to improved wastewater treatment, recycling the phosphate in urine would be an added benefit.

8.4   Potassium production and recovery

Elemental potassium (K) is highly reactive and doesn't naturally exist in its pure state. Potash deposits occur as beds of solid salts beneath the earth's surface and as brines in drying lakes and seas. Potassium is mined from a number of minerals. Sylvinite, sylvite, and langbeinite are the most important mineral sources. World reserves of potash are very unevenly distributed; 75% are located in the former USSR and Canada, which together account for some 60% of the current global production capacity. The concentration of output among a limited number of countries has led to a high level international trade in potash.

About 95% of the commercially produced K are used in fertiliser production, while small amounts are used for industrial purposes and for common household products. Annual global production of K fertilisers is in excess of 25 million tons [i.32]. Potassium is used in different fertilisers in the following forms:
- Potassium Chloride (Most agricultural KCl is produced by the flotation processes);
- Potassium Sulphate;
- Double Sulphate of Potash and Magnesium or langbeinite ($K_2SO_4.2MgSO_4$);
- Potassium Nitrate (KNO_3) contains little or no chloride or sulphur.

Processing potash for the fertiliser market is relatively simple. The ore is finely ground and the potassium chloride (muriate of potash) is recovered in a flotation process. The froth is skimmed off, dried and screened to produce different grades of potassium chloride. Although the refining processing is relatively simple, potash mining could be expensive. Potash is mined by three methods:
- Conventional shaft mining, similar to coal mining;
- Specialised continuous shaft mining, removing ore directly from the vein;
- Solution mining, dissolving potash ore bed with hot water and pumping the potash brine to the surface for refining.

Potash has a low compressive strength and therefore the solution mining technique is used below 1500m. This method makes recovery more expensive, because natural gas has to be used to evaporate the water. As potash reserves become scarce, mines will become deeper and increase the production cost and value of the mineral. The value of potassium in fertiliser is currently around € 400/tK. Raw minerals are worth much less, in the order of € 100/tK.

8.5   Potential for removing, recovering and recycling minerals form wastewater

Recovering minerals from wastewater, rather than *removal* and disposal thereof, would be motivated by the value that they have as fertiliser. More than 80% of the N, P and K produced in industry, are used in fertiliser. The most important factor determining fertiliser price is energy [i.33]. Fertiliser production currently accounts for about 2% of total global energy consumption (UNEP, 1998). Recovering minerals from wastewater could therefore also represent a "recovery" of consumed energy.

Table 14 gives a summary of the energy requirement in the production of minerals used in fertilisers. The values are based on average values from UNEP (1998). If these minerals were recovered and recycled from wastewater, the energy required for industrial production would be saved. The energy "contained" in the minerals in wastewater therefore gives an indication of the amount of energy that could be invested to make recovery more sustainable than industrial production.

Table 14 Production energy and energy contained in minerals present in wastewater

Minerals		Concentration in wastewater g/m^3	Production energy KJ/g	Energy contained in wastewater MJ/m^3
Nitrogen	(NH$_4$-N)	40	38.9	1.55
Phosphate	(P$_2$O$_5$-P)	8	7.8	0.06
Potash	(K$_2$O-K)	14	8.8	0.12
Total				1.74

Mineral recovery would also change the energy consumption of wastewater treatment. The total energy consumption of an advanced wastewater treatment plant (including aeration, sludge digestion, de-watering and sludge incineration) is between 0.7 and 1.2 MJ/m^3, depending on whether it is raw or settled wastewater (van Loosdrecht *et al*, 1997).

Table 15 Maximum potential economic value of minerals in wastewater

Minerals		Concentration (g/m^3)	Production Cost (€/t)	Potential Value (€/m^3)
Ammonia-nitrogen	(NH$_4$-N)	40	182	0.007
Phosphorus	(P$_2$O$_5$-P)	8	344	0.003
Potassium	(K$_2$O-K)	14	181	0.003
Total	(NH$_4$, P, K)			0.01
Struvite	(MgKPO$_4$)	69	300	0.02

Table 15 gives an indication of the maximum economic potential of minerals present in wastewater. The production costs of phosphate and potassium vary greatly and companies are reluctant to make the information available for competitive reasons. In the case of ammonia, the production cost is well known due to the uniformity of the process used. Based on fertiliser prices, containing different concentrations phosphate and potash, the prices for phosphate and potash was assumed the same as that of ammonia. The price of struvite was estimated by von Münch (2001). The sales price of these minerals increase in the form of fertiliser, but the production costs (used in this study) should give an indication of their value.

In the Netherlands, water treatment boards charge an annual tariff of around € 50 per pollution unit, which is based on one person's wastewater production. Based on this tariff and wastewater production of 300 λ/p.d, the cost of wastewater treatment is around € 0,46/m^3 [i.34]. Based on the value of raw materials produced in industry, the total value of minerals in wastewater is insignificant when compared to the cost of wastewater treatment. If struvite would be precipitated, the potential value could be around 5% of the cost of wastewater treatment. Even if urine was collected separately, and phosphate is recovered through struvite precipitation, the potential income would still be low: around € 1,50/m^3 urine.

The water industry will generally choose the cheapest available treatment option that meets regulatory requirements. Economic boundary conditions of mineral recovery will be defined by operating and maintenance costs of advanced wastewater treatment. These costs include aeration costs (energy), sludge treatment, sludge disposal and chemical use. This means that mineral recovery will only be attractive if the cost of wastewater treatment could be lowered.

In the early 1990s, the prospects of recovering minerals from concentrated waste streams were rather poor. The net recovery cost (operational costs less potential income from recovered minerals) would not have been less than the best available removal technology's operational cost. De-nitrification with addition of methanol would have been the cheapest solution of ammonia removal with the methanol price of that time. The chemicals required, as well as costs of complex processes, made precipitation and air stripping more expensive (Siegrist, 1996). This dynamic situation might in future change in favour of mineral recovery.

8.6   Recycling minerals from Dutch waste products

The mineral load in municipal wastewater is only a small fraction of the total mineral load in the Netherlands. Table 16 shows approximate figures for annual production and consumption of nitrogen, phosphorus and potassium in industry, agriculture and wastewater. Livestock farming in the Netherlands involves intensive bio-industries with cattle, pigs and poultry. Manure from these animals contains much more minerals than wastewater. Manure disposal has become a serious environmental concern. Not all animal manure can be used for fertiliser as it rarely supplies the exact mineral composition required by commercial crop farming. Furthermore, minerals from manure are often not readily available and especially in the Netherlands, plants need to take up minerals quickly during the short growing season. Industrial fertiliser is therefore still necessary to yield good harvests. However, since 1986 legislation in the Netherlands has forced farmers to limit the amount of minerals applied to land. The annual mass of applied fertilisers therefore increased little or none. There has also been very little or no increase in the annual manure load over the last few years.

Table 16 shows that the amount of nitrogen in wastewater is almost insignificant (less than 10%) when compared to the combined amounts in fertiliser and animal manure.

Table 16 Annual mass flux of nitrogen phosphorus and potassium in the Netherlands

All figures are for 1998. Recent figures for some cases indicate little increase.	wastewater [i-35]	industry [i-30]	thermphos [i-30]	hydro-agri [i-36]	fertiliser [i-36]	animal manure	cattle (PR, 2001)	calfs (PR, 2001)	pigs [i-37, i-38]	poultry [i-37, i-38]
Nitrogen (kt N/year)	85	-	-	1500	360	515	337	8	114	56
Phosphorus (kt P/year)	14	283	90	-	14	80	42	1	24	13
Potassium (kt K/year)	24	-	-	-	30	200	178	-	22	-

Only a very small amount of secondary (reclaimed) raw material is presently being used by the phosphate industry. Thermphos could therefore theoretically recycle all the phosphorus from Dutch municipal wastewater. The phosphorus load in Dutch municipal wastewater was approximately 14 ktP/a in 1997, which is only 5% of the total load of around 264 ktP/a consumed in the Netherlands (Schipper, 2001). The annual amount of phosphorus in Dutch animal manure is almost equal to the Thermphos production. Although this is potentially a much bigger source of secondary sources than human urine, there are many practical problems (such as high content of organic material, high water content, low P-concentration). Other sources, such as ash from incinerated poultry manure is now being investigated. Poultry manure, for instance, contains the same amount of phosphorus as the total wastewater load. Recovery from bio-industry seems more feasible as it presents large and concentrated sources at specific locations, thereby strongly reducing recovery costs.

The Netherlands has no potash production industry (unlike ammonia and phosphate). The amount of potassium in municipal wastewater is also small in relation to that in animal manure (table 16). The amount of potassium in wastewater is similar to that in fertiliser.

8.7 Conclusions

- Some raw materials, such as phosphate and potash, are finite resources. In order to meet the aims of sustainability, these minerals would have to be recycled, even if recycling is more expensive than production.

- Minerals in wastewater constitute only a fraction of the minerals in other waste products. The amount of minerals and higher concentrations would probably make recovery from animal manure more efficient. If separate collection and treatment of urine leads to overall energy saving in wastewater, it would still present a highly concentrated source of recoverable minerals.

- Recovery of minerals should not be focussed on producing secondary raw materials for industry. Phosphate could be recovered as struvite and used directly as fertiliser or in combination with other fertilisers. Controlled struvite precipitation from side streams could also be an economic alternative to more conventional phosphate removal.

- Recovery of "valuable" minerals from aqueous waste streams should be preceded by careful economic and thermodynamic analysis. Minerals in waste streams are valuable because they originate from finite resources and require energy in the production processes. While the economic value of minerals is as low as it is, processes to recover minerals would have to be more economical than traditional removal processes.

- Nitrogen is not a finite mineral and direct recovery is not the only option. Ammonia removal with the Sharon/Anammox process followed by ammonia production in the Haber process cost only 12.5% of the recovery from wastewater (via air stripping). The energy required for direct recovery (air stripping) is 30% less than the energy combined in removal and production.

- Proper evaluation of sustainability requires a method to relate economic cost to energy consumption of a process.

9. GENERAL CONCLUSIONS AND RECOMMENDATIONS

9.1 Introduction

Human faeces in cities could cause diseases. Modern sanitation is contingent to the history of cities, which initiated current wastewater technology. Water is the most efficient medium to transport faeces (solids) and minerals (dissolved), but wastewater treatment is complex and requires energy. More sustainable societies will *recover minerals* rather than *remove nutrients*. Recycling of minerals, by utilising minimal resources, should occur at the same rate as consumption. A healthier environment would be a consequence of sustainable economics.

9.2 Origins of minerals in wastewater

Most of the minerals in wastewater originate in urine. Nitrogen in urine occurs as urea, which rapidly hydrolysis to ammonia when diluted in wastewater. On average, urine contributes 12 gN/p.d (80%) of the total nitrogen load in wastewater. The average ammonia concentration in Dutch wastewater is 40 gN/m^3 and the total nitrogen concentration is 50 gN/m^3.

Phosphorus in urine occurs as different phosphates. On average, urine contributes 1 gP/p.d (45%) of the total phosphorus load in wastewater. The amount of phosphorus in faeces is 0.7 gP/p.d (30% of the total). All other phosphorus originates in various household sources. The phosphorus concentration in urine is variable, unlike the urea concentration that is more constant.

On average, the contribution of urine to the total potassium concentration in wastewater is around 3 gK/p.d. (70% of the total). Although potassium is not an important pollutant, it contributes to the salinity of the effluent.

On average, 1.25λ/p.d. urine is produced. Average daily wastewater production is 300λ/p.d. (130λ/p.d. domestic sewage, 100λ/p.d. rain and infiltration and 70λ/p.d. industrial wastewater). The mineral concentration in urine is 100 –200 times as high as in municipal wastewater.

9.3 Initiatives to collect urine separately

Sweden initiated separate urine collection. A few "ecological villages" were built as attempts to establish an alternative life style that recognises the finite nature of fossil energy and mineral resources. Ecological villages collect urine separately to be used as fertiliser by farmers. *After* implementation of these systems, many studies were conducted to solve technical problems and to compare the technology to existing technology, based on energy or exergy analysis. The research in Sweden is rather fragmentised.

Novaquatis (EAWAG, Switzerland) considers the complete cycle of urine separation technology. The research covers diverse issues, such as human excretion, sanitation, urine collection, -transport and -treatment, wastewater treatment, surface water quality, agriculture and socio-political aspects and conceptions of urine technology. The research is conducted with bold vision in an integrated fashion, but the lack of researchers is a great obstacle.

Some projects to collect urine separately have also been started in Germany and Austria. In Western Europe, separate urine collection provides an alternative form of sanitation that has a "green" image. This image is used in trying to sway opinions in favour of developments.

In the Third World, urine separation (and composting faeces) could provide inexpensive and uncomplicated sanitation. This technology requires no water and little infrastructure.

9.4   Separate collection and handling of urine

Dilution of urine should be prevented with separate collection, as the hydrolysis rate of urea to ammonia is proportional to the amount of dilution. Ammonia evaporation could lead to a loss of recoverable minerals, but could also lead to problems with odours and corrosion.

Commercial urine separation or "no-mix" toilets were produced and installed successfully in the past decade. Waterless urinals were first introduced to save water, but it could be applied to collect urine undiluted. The technology seems to be developing in the right direction.

Questions regarding urine transport have not yet been answered. In Switzerland, Novaquatis considers the option to drain urine via existing sewers late at night (when supposedly there is no wastewater). The benefits of urine separation probably wouldn't "pay" for truck transport. Local or de-central treatment of urine seems to be a solution. The treated liquid could drain through sewers and the recovered minerals could be removed by trucks.

Research and experience have shown that separate urine collection, handling and storage are relatively safe to communities or operators. Methods exist to determine microbial risks in urine separation systems and to monitor occurrence of faecal contamination.

9.5   Effects of separate urine collection on environmental health

During rainstorms, nutrients and micro-pollutants are discharged directly into surface water with combined sewer overflows. Although models fail to quantify this discharged accurately, urine separation would most likely reduce the direct discharge of nutrients and micro-pollutants.

Wastewater treatment works are not designed to remove micro-pollutants, such as hormones and pharmaceutical end products. Studies on aquatic ecosystems indicated the occurrence of endocrine disrupters. It is believed that micro-pollutants cause these effects, which might also be responsible for some chronic illnesses and disorders amongst people.

Micro-pollutants in municipal wastewater mainly originate in human urine. The load of micro-pollutants and nutrients (N and P) currently discharged into surface waters would be reduced if urine were collected and treated separately.

9.6   Effects of separate urine collection on biological nutrient removal processes

Modified UCT-type wastewater treatment processes, such as the BCFS® process, would require between 20% and 30% less oxygen if 50% of all urine were collected separately. Total nitrogen effluent concentrations of 2.2 gN/m^3 could be possible in these processes with settled or pre-precipitated wastewater and where more than 50% of all urine was collected separately. The relative improvement in effluent quality could be even more significant in less advanced treatment processes.

If all nutrients were consumed in cell metabolism, very small treatment works with low sludge ages (SRT = 1) could be possible. Simulation results suggest that with between 50% and 60% separate urine collection, such a process could still achieve $N_{tot\_eff} < 10$ gN/m^3.

9.7 Techniques for reclaiming minerals from wastewater or urine

Ammonia can be recovered from solution by a number of techniques, such as ammonia stripping. Higher ammonia concentrations, liquid pH and temperature lead to more efficient stripping. Ion exchange reactions or ammonia adsorption could also be used to recover ammonia.

Calcium phosphate can be recovered from wastewater with complex and expensive processes. Recovered calcium phosphate is virtually indistinguishable from mined apatite.

Struvite (magnesium ammonium/potassium phosphate) is a form of kidney stone. Struvite precipitation is a simple process in comparison to calcium phosphate precipitation. Controlled struvite precipitation from side streams could be economically beneficial when compared to more conventional chemical phosphate removal. Struvite can be recovered from urine by addition of magnesium.

Higher mineral concentrations reduce the energy demand and chemical dosing required for recovery. Urine is a concentrated mineral solution, even when compared to some industrial waste streams and it should be possible to recover minerals from urine in cost effective ways.

Nitrogen is not a finite resource. If nitrogen removal (including the energy required for industrial ammonia production) requires less energy than ammonia recovery, it could be more sustainable. Ammonia can be removed efficiently in the Sharon/Anammox process.

9.8 Possibilities of recycling minerals in wastewater through agriculture or industry

Raw materials, such as phosphate and potash, are finite resources. In order to meet the aims of sustainability, some of these minerals must be recycled, even if recycling costs are higher than current production costs. However, it is not obvious whether these minerals should be recovered from wastewater. The amount and concentration of minerals in wastewater are much lower than in animal manure, for instance. Still, if separate urine collection and treatment could lead to considerable benefits for conventional wastewater treatment, it would present a highly concentrated source of minerals that could be recovered.

The price of apatite is only around € 40/ton, but minerals don't have to be recovered as a secondary raw material for industry. Struvite, for instance, could be used directly as fertiliser or in combination with other fertilisers.

Recovery of "valuable" minerals from aqueous waste streams should be preceded by careful economic and thermodynamic analysis. Minerals in waste streams are valuable because they originate from finite resources and require energy in the production processes. Mineral resources and energy have not reached critically low levels and exploitation of natural resources is therefore still economically attractive.

It seems that thermodynamics indicate nutrient recovery and reuse as more sustainable than nutrient removal followed by industrial production. However, techniques for nutrient recovery are still more expensive than removal. Economic factors can not be disregarded in terms of the objectives of sustainability.

9.9   Recommendations

Prior to pilot installations and implementation of urine separation techniques, a clear quantification of all aspects of separate urine collection must be made. The following areas of interest need to be investigated in more detail.

1. Effects of urine separation on **existing** advanced wastewater treatment processes
   The effect of urine separation on modified UCT-type process and other less advanced wastewater treatment processes must be evaluated regarding:
- Oxygen and energy requirements
- Effluent quality
- Land-use and the accommodation of larger populations in existing WWTWs

2. **New** combinations of existing processes to treat total wastewater delivered in separate streams
   Combinations of existing treatment processes must be evaluated to determine if separate urine collection and treatment could improve the total wastewater system:
- Nitrification of concentrated ammonia in urine with Sharon/Anammox
- Treatment of COD in "normal" concentration and nutrients in grey and black water (aerobic reactor with SRT = 0,5 to 1d, trickling filter/bio-filter, sand filtration, etc)
- Chemical phosphate removal from urine and wastewater (side streams)
- Integrating all process units and side streams to present a balanced mass flow diagram of nitrogen, phosphorus and COD

3. Theoretical and experimental evaluation of mineral recovery from urine
   Existing techniques could be applied to recover minerals from urine. Central wastewater treatment plants already employ some of these techniques for side stream treatment. A qualitative analysis on how to apply these techniques for mineral recovery from urine must:
- Optimise techniques to remove and/or recover phosphate through crystallisation of minerals such as struvite, newberyite or apatite
- Optimise ammonia stripping or adsorption and recovery

Recovery processes must be compared to recovery processes with existing advanced wastewater treatment (refer points 1 & 2) in terms of energy consumption.

4. De-central urine treatment and mineral recovery
   The techniques evaluated for large-scale systems should also be applicable to a smaller scale, with the aim of eventual local or de-central treatment. Efficient collection and storage of urine in large buildings must be investigated to prevent practical problems such as odours, precipitation and excessive dilution.

5. Comparison of different wastewater treatment techniques where urine is handled separately based on an analysis of **exergy**.
   Three or four different scenarios should be compared:
- Existing wastewater treatment processes with less urine in influent, and urine being transported and treated separately
- New central wastewater treatment process to treat urine and grey/black water separately
- New de-central wastewater treatment process
- "Ecological" sanitation with urine separation and faecal composting on household scale

6. Social aspects

Technical and economic feasibility does not guarantee social acceptance of recycling minerals from wastewater. A better understanding of the socio-political aspects of mineral recycling from wastewater requires further study.

7. Future co-operation with other research institutions

The Novaquatis group in Switzerland expressed their interest in working with the TU Delft on some aspects of urine separation technology. Co-operation must be managed in an integrated way.

8. Pilot project

Future research and co-operation with other researchers should be done with the aim to set up a pilot installation, including:

- toilets and urinals to collect urine separately
- storage tank and reactor to treat urine and recover minerals
- equipment to separate and de-water recovered minerals

List of references

Abbona F., Boistelle R. and Ludager H.E. (1982) "Crystallisation of two magnesium phosphates, struvite and newberyite: effect of pH and concentration", Journal of Crystal Growth, Vol. 57 pp. 6 – 14.

Abbona F. and Baronnet A. (1996) "A XRD and TEM study on the transformation of amorphous calcium phosphate in the presence of magnesium", Journal of Crystal Growth, Vol. 165 pp. 98-105.

Alexander M. (1977) "Introduction to soil microbiology", 2nd ed. John Wiley, New York.

Altman P.L. and Dittmer D.S. (1968) "METABOLISM", Federation of American Societies for Experimental Biology, Biological handbooks.

Aspergen H., Hellström B.-G. and Olsson, G. (1997) "The urban water system - A future Swedish perspective", Water Science and Technology, Vol. 35 (9) pp. 33 – 43.

Austin A. and Duncan L. (1999) "Urine diversion technology in South Africa: Background and case study", IMIESA, Vol. 24 (3).

Battistoni P., Fava G., Pavan P., Musacco A. and Cecchi F. (1997) "Phosphate removal in anaerobic liquors by struvite crystallisation without addition of chemicals: preliminary results", Water Research, Vol. 31 (11) pp. 2925 – 2929.

Battistoni P., Pavan P. Cecchi F. and MataAlvarez J. (2000) "Struvite crystallisation: a feasible and reliable way to fix phosphorus in anaerobic supernatants", Water Research, Vol. 34 (11) pp. 3033 – 3041.

Beer A.-J. (1997) "Something in the water", Biologist, Vol. 44 (2) pp. 296.

Beler Baykal B. and Akca Guven D. (1997) "Performance of clinoptilolite alone and in combination with sand filters for the removal of ammonia peaks from domestic wastewater", Water Science and Technology, Vol. 35 (7), pp. 47 - 54.

Beler Baykal B. (1998) "Clinoptilolite and multipurpose filters for upgrading effluent ammonia quality under peak loads", Water Science and Technology, Vol. 37 (9) pp. 235 - 242.

Beck, W.S. (1971) "Human Design", Harcourt Brace Javanovich Inc.

Belfroid A.C., Van der Horst A., Vethaak A.D., Schäfer A.J., Rijs G.B.J., Wegener J. and Cofino W.P. (1999) "Analysis and occurrence of estrogenic hormones and their glucuronides in surface water and waste water in The Netherlands", The Science of the Total Environment, Vol. 225 pp. 101-108.

Berg F.D. and Kuss E. (1991) "Urinary-excretion of catecholestrogens, 2-methoxyestrogens and classical estrogens throughout the normal menstrual-cycle", Archives of Gynecology And Obstetrics, Vol. 249 pp. 201-207.

Berne R.M. and Levy M.N. (1998) "Physiology", 4th edition, St. Louis Mosby Inc.

Bertrand-Krajewski J-L., Lefebvre M. and Barker J. (1997) "Ammonia removal and discharges during storm events: integrated approach for a small WWTP and associated CSOs", Water Science and Technology, Vol. 36 (8-9) pp. 229–234.

Bijker W.E., Hughes T.P., Pinch T.F. and Trevor J. (1987) "The social construction of technological systems; new directions in the sociology and history of technology" MIT Press.

Boer, H. (1995) Afvalwaterzuivering in (-) balans, Een balansmatige analyse van de stofstromen betrokken bij waterzuivering in Nederland, Interfacultaire Vakgroep Energie en Milieukunde, IVEM-doctoraalverlsag nr.14, Rijksuniversiteit Groningen.

Borgerding J. (1972) "Phosphate deposits in digestion systems", Journal of Water Pollution Control Federation, Vol. 44 pp. 813 – 819.

Brandse F. and van Loosdrecht M.C.M. (2001) "Sewage treatment and methods for phosphate recovery in BCFS process", Proceedings from the 2nd International Conference on Phosphate Recovery, Noordwijkerhout, The Netherlands.

Brdjanovic D., van Loosdrecht M.C.M., Versteeg P., Hooijmans C.M., Alaerts G.J. and Heijnen J.J. (2000) "Modelling COD, N and P removal in a full-scale WWTP Haarlem Waarderpolder", Water Research, Vol. 34 (3), pp. 846 – 858.

Brett S., Guy J., Morse G. K. and Lester J. N. (1997) "Phosphorus removal and recovery technologies", Centre Européen D'Études des Polyphosphates, Selper Publications, London.

Buday M. (1994) "Reutilization of ammonia from wastewater using cation exchange resins", Water Science and Technology, Vol. 30 (9) pp. 111–119.

Carlander, Hoglund and Vinnerås (1999) "Field experiment at Stockholm on the fertilising value of source separated human urine" Stockholm Water Company Report

Cheung K.C., Chu L.M. and Wong M.H. (1997) "Ammonia stripping as a pretreatment for landfill leachate", Water Air and Soil Pollution, Vol. 94 (1-2) pp. 209-221.

Ciba Geigy (1977) "Wissenschaftliche Tabellen Geigy", Körperflüssigkeiten, 8th ed, Basel.

Collivignarelli C., Bertanza G., Baldi M. and Avezzu F. (1998) "Ammonia stripping from MSW landfill leachate in bubble reactors: Process modeling and optimization", Waste Management and Research, Vol. 16 (5) pp. 455-466.

Davenport, H.W. (1966) "Physiology of the digestive tract", Yearbook medical publishers Inc. Chicago.

Davson H. and Segal M.B. (1975) "Introduction to physiology", London Academic press.

Dimova G., Mihailov G. and Tzankov T. (1999) "Combined filter for ammonia removal - part I: Minimal zeolite contact time and requirements for desorption", Water Science and Technology, Vol. 39 (8) pp. 123 – 129.

Driver J., Lijmbach D. and Steen I. (1999) "Why recover phosphorus for recycling, and how?" Environmental Technology Vol. 20 (7) pp. 651 – 662.

Evenblij, H. (2001) Personal communication, Department of Civil Engineering, TU Delft.

Farahbakhshazad N. and Morrison G.M. (2000) "Nitrification and de-nitrification of urine in a column upflow macrophyte system", Department of Applied Environmental Science, Göteborg University (http://www.miljo.chalmers.se/fug/neda.htm).

Fittschen I. and Hahn H.H. (1998) "Characterization of the municipal wastewaterpart human urine and a preliminary comparison with liquid cattle excretion", Water Science and Technology, Vol. 38 (6) pp. 9 – 16.

Giesen A. (1999) "Crystallisation process enables environmental friendly phosphate removal at low costs", Environmental Technology, Vol. 20 (7) pp. 769 – 775.

Govers M.J.A.P. (1993) "Dietary calcium and phosphate in the prevention of colorectal cancer: mechanism and nutritional implications", PhD Thesis, Rijks Universiteit Groningen.

Griffith D.P. (1978) "Struvite stones", Kidney International, Vol. 13 pp. 372 – 382.

Grum M., Aalderink R.H., Lijklema L. and Spliid H. (1997) "The underlying structure of systematic variations in the event mean concentrations of pollutants in urban runoff." Water Science and Technology, Vol. 36 (8-9) pp. 135–140.

Gujer W., Henze M., Mino T., Matsuo T., Wentzel M.C. and Marais G. v. R. (1995) "The activated sludge model no 2: biological phosphorus removal", Water Science and Technology, Vol. 31 (2), pp. 1 – 11.

Guyton A.C. (1991) "Textbook of Medical Physiology", 8th edition, W.B. Saunders Company.

Hanæus, Å., Hellström, D. and Johansson, E. (1996) "Conversion of urea during storage of human urine" Vatten, Vol. 52 (4) pp. 263-270.

Hanæus J., Hellström D. and Johansson E. (1997) "A study of a urine separation system in an ecological village in northern Sweden", Water Science and Technology, Vol. 35 (9) pp. 153-160.

Hao X.D. (2001) "Model based optimisation of sustainable biological nutrient removal processes", PhD Thesis, Delft University of Technology.

Hao, X. Heijnen, J.J., Qian, Y. and van Loosdrecht, M.C.M. (2001) "Contribution of P-bacteria in BNR processes to overall effects on the environment", Water Science and Technology, Vol. 44 (1) pp. 67 – 76.

Harremoës P. (1997) "Integrated water and waste management", Water Science and Technology, Vol. 35 (9) pp. 11 – 20.

Hellström D. (1999) "Exergy Analysis: A comparison of source separation systems and conventional treatment systems", Water Environment Research, Vol. 71 (7).

Hellström D. (1998) "Nutrient Management in Sewerage Systems: Investigations of Components and Exergy Analysis", PhD Thesis, Lulea University of Technology, 1998:02.

Hellström D. and Johansson E. (1999) "Swedish experience with urine separating systems", Wasser & Boden, Vol. 51 (12) pp. 26 – 29.

Hellström D., Johansson E. and Grennberg K. (1999) "Storage of human urine: acidification as a method to inhibit decomposition of urea", Ecological Engineering, Vol. 12 pp. 253 – 269.

Hellström D. and Kärrman E. (1996) "Nitrogen and Phosphorus in Fresh and Stored Urine", Environmental Research Forum, Vols. 5-6, pp. 221-226.

Hellström D. and Thurdin J. (1998) "Drying as a urine dewatering method" Vatten, Vol. 54 (2) pp. 109-114.

Henze M., Gujer W., Mino T., Matsuo T., Wentzel M.C., Marais G.v.R. and van Loosdrecht M.C.M. (1999) "Activated sludge model No. 2D", Water Science and Technology, Vol. 39 (1), pp. 165 – 182.

Hirsch R., Ternes T., Haberer K. and Kratz K-L. (1999) "Occurrence of antibiotics in the aquatic environment", The Science of the Total Environment, Vol. 225 pp. 109 -118.

Höglund C., Stenström T.A., Jönsson H. and Sundin A. (1998) "Evaluation of faecal contamination and microbial die-off in urine separating sewage systems". Water Science and Technology, Vol. 38 (6) pp. 17 - 25.

Holzer P. and Krebs P. (1998) "Modelling the total ammonia impact of CSO and WWTP effluent on the receiving water." Water Science and Technology, Vol. 38 (10) pp. 31–39.

Houssay, B.A. (1955) "Human Physiology", McGraw Hill Book Company, Inc.

Hosper, S. (1998) "Stable states, buffers and switches: an ecosystem approach to the restoration and management of shallow lakes in the Netherlands", Water Science and Technology, Vol. 37 (3), pp. 151 - 164.

Hultman B., Levlin E., and Stark K. (2001) "Phosphorus recovery form sewage sludges: Research and experiences in Nordic countries", SCOPE Newsletter No. 41 pp. 29 – 32.

Ishiduka S., Sato S., Shibata M. and Togami Y. (1998) "Recovery of fluoride and phosphate from wastewater", Symposium of the Society of Chemical Engineers in Japan.

Jenssen P.D. (1999) "An overview of source separation systems", Proceedings of the 4th International Conference on Ecological Engineering, Norway.

Jenssen, P.D., "A full-scale source separation system using vacuum toilets and grey water treatment in a wetland", Department of Agricultural Engineering, Agricultural University of Norway.

Jönsson H., Stenström T.-A., Svensson J. and Sundin A. (1997) "Source separated urine-nutrient and heavy metal content, water savings and faecal contamination", Water Science and Technology, Vol. 35 (9) pp. 145 – 152.

Jönsson H., Dalemo M., Sonesson U. and Vinnerås B. (1998) "Modelling the sewage system - evaluating urine separation as a complementary function to the conventional sewage system", Systems engineering models for waste management International workshop in Göteborg, Sweden.

Jozuka, T. (2001) "Phosphorus recovery from sewage sludge incineration ash", SCOPE Newsletter, No. 43 pp. 7 – 8.

Kabdasli I., Tunay O., Ozturk I., Yilmaz S. and Arikan O. (2000) "Ammonia removal from young landfill leachate by magnesium ammonium phosphate precipitation and air stripping", Water Science and Technology, Vol. 41 (1) pp. 237 – 240.

Karlsson I. (2001) "Full scale plant recovering iron phosphate at Helsingborg Sweden", Proceedings from the 2nd International Conference on Phosphate Recovery, Noordwijkerhout, The Netherlands.

Karrman E. (1997) "Analysis of wastewater systems, with respect to environmental impact and the use of resources", Internal report, Chalmers University of Technology.

Kars H., Wevers J.M.A.R. and Schuiling R.D. (1983) "Struvite, a mineralogical curiosity from an archeological site", ROB (Berichten Rijksdienst Oudheidkundig Bodemonderzoek) Vol. 30 pp. 423 – 426.

Kathesis D., Diyamandoglu V. and Fillos J. (1998) "Stripping and recovery of ammonia from centrate of anaerobically digested biosolids at elevated temperatures", Water Environment Research, Vol. 70 (2) pp. 231 – 240.

Keel C.A. and Neil E. (1965) "Samson Wright's Applied Physiology, Oxford Medical Publications", 11th edition, Oxford University Press.

Kibalczyc W., Zielenkiewicz A. and Zielenkiewicz W. (1988) "Calorimetric investigations of calcium phosphate precipitation in relation to solution composition and temperature", Thermochimica Acta, Vol. 131 Pages 47-55

Kongshaug G. (1998) "Energy consumption and greenhouse gas emissions in fertiliser production", Proceedings from IFA (International Fertiliser Industry Association) Technical Conference, Marrakech, Morocco.

Körner W., Hanf V., Schuller W., Kempter C., Metzger J. and Hagenmaier H. (1999) "Development of a sensitive E-screen assay for quantitative analysis of estrogenic activity in municipal sewage plant effluents", The Science of the Total Environment, Vol. 225 pp. 48.

Koutsopoulos S. and Dalas E. (2000) "The effect of acidic amino acids on hydroxyapatite crystallization", Journal of Crystal Growth, Vol. 217 (4) pp. 410-415.

Koutsoukos P.G. and Bouropoulos N.C. (2000) "Spontaneous precipitation of struvite from aqueous solutions", Journal of Crystal Growth, Vol. 213 (3-4) pp. 381-388

Koutsoukos P.G. (2001) "Current knowledge of calcium phosphate chemistry and in particular solid surface-water interface reactions", Proceedings from the 2nd International Conference on Phosphate Recovery, Noordwijkerhout, The Netherlands.

Kothari A.C. (1999) "Recovery of ammonia from affluents", Chemical Engineering World, Vol. 34 (2) pp. 101 – 103.

Kuba T., van Loosdrecht M.C.M. and Heijnen J.J. (1996) "Phosphorus and nitrogen removal with minimal COD requirement by integration of denitrifying dephosphatation and nitrification in a two-sludge system", Water Research, Vol. 30 (7), pp. 1702 – 1710.

Kümmerer, K. (2001) "Drugs in the environment: emission of drugs, diagnostic aids and disinfectants into wastewater by hospitals in relation to other sources - a review", Chemosphere, Vol. 45 (6-7) pp. 957-969.

Kvarnström E. and Nilsson M. (1999) "Reusing Phosphorus: Engineering Possibilities and Economic Realities", Journal of Economic Issues, Vol. 33 (2) pp. 393 – 402.

Langeveld J. G. (1999) "Historische ontwikkelingen in de omgang met water in de stad: welke lessen leren we uit het verleden?" PAO cursus "Water in de stad", Technische Universiteit Delft.

Larsen T.A. and Gujer W. (1996) "Separate management of anthropogenic nutrient solutions (human urine)", Water Science and Technology, Vol. 34 (3-4) pp. 87 - 94.

Larsen T.A. and Gujer W. (1997) "The concept of sustainable urban water management", Water Science and Technology, Vol. 35 (9) pp. 3 – 10.

Larsen T.A., Peters I., Alder A., Eggen R., Maurer M. and Muncke J. (2001) "Re-engineering the toilet for sustainable wastewater management", Environmental Science and Technology, Vol. 35 (9) pp. 192 – 197.

Larsen T.A. and Udert K. (1999) "Urinseparierung - ein Konzept zur Schließung der Nährstoffkreisläufe", Wasser & Boden, Vol. 51 (11) pp. 6-9.

Lehnartz E. (1943) "Physiological and pathological chemistry. Including Nutrition", Dieterich Wiesbaden.

Lehnartz E. (1959) "Einführung in die chemische physiologie", 11th edition, Springer Berlin.

Levitt R., Therivel R. and Carlton-Smith J. (1998) "Sustainability", Paper prepared as background to the study, Royal Commission Environmental Pollution Study on Energy and the Environment.

Lind B.-B., Ban Z. and Bydén S. (2000) "Nutrient recovery from human urine by struvite crystallization with ammonia adsorption on zeolite and wollastonite", Bioresource Technology, Vol. 73 (2) pp. 169 – 174.

Lind B-.B., Ban Z. and Bydén S. (2001), "Volume reduction and concentration of nutrients in human urine", Ecological Engineering", Vol. 16 (4) pp. 561-566.

Lutz I. and Kloas W. (1999) "Amphibians as a model to study endocrine disruptors: I. Environmental pollution and estrogen receptor binding", The Science of the total environment, Vol. 225 pp. 49-57.

Lutz I. and Kloas W. (1999) "Amphibians as a model to study endocrine disruptors: II. Estrogenic activity of environmental chemicals in vitro and in vivo", The Science of the Total Environment, Vol. 225 pp. 59-68.

Mamais D., Pitt P.A., Cheng W.Y., Loiacono J. and Jenkins D. (1994) "Determination of ferric chloride dose to control struvite precipitation in anaerobic digesters", Water Environment Research, Vol. 66 pp. 411 – 416.

Matsuo Y. (1995) "Release of phosphorus from ash produced by incinerating waste activated sludge from enhanced biological phosphorus removal", Publisher unknown.

Meijer S.C.F., van Loosdrecht M.C.M. and Heijnen J.J. (2001) "Metabolic modelling of a full-scale enhanced biological phosphorus removing WWTP", Water Research, Vol. 35 (11), pp. 2711-2723.

Meierer M., Eimer N. and Bloechl H. (1999) "Novel process combination for treating ammonia-containing power plant effluent", VGB PowerTech Vol. 79 (2) pp. 67-70.

Merkel A. (1998) "The role of science in sustainable development", Science, Vol 281 pp. 336 – 337.

Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer, 2000, De Kwalitiet van het Drinkwater in Nederland in 1998, Inspectiereeks 2000/13.

Mohajit K., Bhattarai K., Taiganides E.P. and Yap B.C. (1989) "Struvite deposits in pipes and aerators" Biological Wastes Vol. 32 pp. 133 –147.

Momberg G.A. and Oellerman R.A. (1992) "The removal of phsophate by hydroxyapatite and struvite crystallisation in South Africa", Water Science and Technology, Vol. 26 pp. 987 –996.

Mulliss R., Revitt D.M. and Shutes R.B.E. (1997) "The impacts of discharges from two combined sewer overflows on the water quality of an urban watercourse", Water Science and Technology, Vol. 36 (8-9) pp. 195-199.

Murnleitner E., Kuba T., van Loosdrecht M.C.M. and Heijnen J.J. (1997) "An integrated model for the aerobic and denitrifying biological phosphorus removal process", Biotechnology and Bioengineering, Vol. 54, pp. 434 – 450.

Nichols P.D., and Leeming R. (1991) "Tracing sewage in the marine environment." Chemistry in Australia, July, pp. 211 – 276.

Ohlinger K. N., Young T. M. and Schroeder E. D. (1998) "Predicting struvite formation in digestion", Water Research, Vol. 32 (12) pp. 3607-3614.

Oosterom R. and Wulkan R. (2000) Urologists - Personal communications.

Otterpohl, R. (2000) "Design of highly efficient source control sanitation and practical experiences", EURO-Summer School, Desar, Wageningen, The Netherlands.

Otterpohl R., Albold A. and Oldenburg M. (1999) "Source control in urban sanitation and waste management: ten systems with reuse of resources", Water Science and Technology, Vol. 39 (5) pp. 153 – 160.

Otterpohl R., Grottker M. and Lange J. (1997) "Sustainable water and waste management in urban areas", Water Science and Technology, Vol. 35 (9) pp. 121 – 134.

Pak C.Y.C. (1998) "Kidney Stones", The Lancet, Vol. 351 pp. 1797 – 1801.

Portielje R. and van der Molen D.T. (1998) "Trend-analysis of eutrophication variables in lakes in The Netherlands", Water Science and Technology, Vol. 37 (3), pp. 235–240.

Rauch W. and Harremoës P. (1997) "Acute pollution of recipients in urban areas." Water Science and Technology, Vol. 36 (8-9) pp. 179-184.

Reichert, P. (1998) "AQUASIM 2.0 – Computer Program for the Identification and Simulation of Aquatic Systems", EAWAG, ISBN 3 906484-16-5.

Rodhe L. and Johansson S. (1996) "Ammonia emission and nutrient utilization from urine using different techniques for application", Swedish Institute of Agricultural Engineering (JTI) report 217.

Ruan M. (1999) "Computational modelling of emission from combined sewer overflows", PhD Thesis, Delft University of Technology.

Sagberg P. and Berg K.G. (2000) "Cost optimisation of nitrogen removal in a compact nitrogen and phosphorus WWTP", Water Science and Technology, Vol. 41 (9) pp. 147 – 154.

Sahlberg B.L., Landgren B.M. and Axelson M. "Metabolic profiles of endogenous and ethynyl steroids in plasma and urine from women during administration of oral-contraceptives", (1987) Journal of Steroid Biochemistry and Molecular Biology, Vol. 26 (5) pp. 609-617.

Sawyer C.N. (1960) "Chemistry for sanitary engineers", McGraw Hill Book Company Inc.

Schipper W. (2001) Personal communication based on a phosphorus cycle compiled by Thermphos in 1999.

Schipper W., Klpawijk A., Potjer B., Rulkens W.H., Temmink B.G., Kiestra F.D.G. and Lijmbach A.C.M. (2001) "Phosphate recycling in the phsophate industry", Environmental Technology, Vol. 22 (11) pp. 1337 – 1346.

Schomaker A.H.H.M., Beverloo W.A. and van Velsen A.F.M. (1984) "Procestechnologisch onderzoek van het NH₃ luchtstrip/absorptieproces" Vakgroep Waterzuivering, Landbouwhogeschool, Wageningen.

Schuiling R.D. and Andrade A. (1999) "Recovery of struvite from calf manure", Environmental Technology Vol. 20 (7) pp. 765 – 768.

Sheldon R.P. (1964) "Paleolatitudinal and Paleogeographic Distribution of Phosphorite", U.S. Geological Survey Professional Paper 501–c, pp. 106-113.

Shilton A. (1996) "Ammonia volatilization from a piggery pond", Water Science and Technology Vol. 33 (7) pp. 183 – 189.

Siegrist H. (1996) "Nitrogen removal from digester supernatant - comparison of chemical and biological methods", Water Science and Technology, Vol. 34 (1-2) pp. 399–406.

Smolders G.J.F., van der Meij J., van Loosdrecht M.C.M. and Heijnen J.J. (1994) "Stoichiometric model of the aerobic metabolism of the biological phosphorus removal process", Biotechnology and Bioengineering, Vol. 44 pp. 837 – 848.

Snow J. (1856) "Cholera and the water supply in the south districts of London, in 1854", Journal of Public Health and Sanitary Rev, Vol. 2 pp. 239-257.

Snow J. (1855) "On the mode of communication of cholera. London" 2nd edition, John Churchill.

Song Y., Hahn H.H. and Hoffmann E. (2001) "The effects of pH and Ca/P ration on the precipitation of calcium phosphate", Proceedings from the 2nd International Conference on Phosphate Recovery, Noordwijkerhout, The Netherlands.

Stacey R.W. and Santolucito J.A. (1966) "Modern college physiology", St. Louis Mosby.

STOWA (1995) "Behandeling van stikstofrijke retourstromen op rioolwaterzuiveringsinrichtingen: Evaluatie van Nederlandse praktijkonderzoeken", STOWA report 95-08.

STOWA (1995) "Behandeling van stikstofrijke retourstromen op rioolwaterzuiveringsinrichtingen: Praktijkonderzoek aan de membraan-bioreactor bij het slibverwerkingsbedrijf Sluijesdijk", STOWA report 95-09.

STOWA (1995) "Behandeling van stikstofrijke retourstromen op rioolwaterzuiveringsinrichtingen: Praktijkonderzoek aan de driefasen-airliftreactor bij de RWZI Utrecht", STOWA report 95-10

STOWA (1995) "Behandeling van stikstofrijke retourstromen op rioolwaterzuiveringsinrichtingen: Praktijkonderzoek aan stroomstrippen met dampverdichting bij de rwzi Amsterdam Oost", STOWA report 95-11.

STOWA (1995) "Behandeling van stikstofrijke retourstromen op rioolwaterzuiveringsinrichtingen: Praktikonderzoek aan lucht –en stoomstripinstallaties bij de RWZI Utrecht", STOWA report 95-12.

STOWA (1995) "Behandeling van stikstofrijke retourstromen op rioolwaterzuiveringsinrichtingen: Praktijkonderzoek aan het MAP/CAFR-process bij de RWZI Utrecht", STOWA report 95-13.

STOWA (1996) "Behandeling van stikstofrijke retourstromen op rioolwaterzuiveringsinrichtingen: Enkelvoudig reactorsysteem voor ammoniumverwijdering via nitriet", STOWA report 96-01.

STOWA (1996) "Methoden voor influentkarakterisering - inventarisatie en richtlijnen", STOWA report 96-08.

STOWA (1996) "Varianten op voorbezinking: Een haalbaarheidsstudie", STOWA report 96 –20.

STOWA (1996) "Verwijdering van ammonium uit slibgistingswater met het Anammox-proces: Haalbaarheidsstudie", STOWA report 96-21.

STOWA (2000) "Het gecombinmeerde Sharon/Anammoxproces: Een duurzame methode voor N-verwijdering uit slibgistingswater", STOWA report 2000-25.

Strand, F.L. (1978) "Physiology, A regulatory systems approach", MacMillan Publishing Co. Inc, New York

Stumpf M. Ternes T.A. Wilken R-D., Rodrigues S.V. and Baumann W. (1999) "Polar drug residues in sewage and natural waters in the state of Rio de Janeiro, Brazil", The Science of the Total Environment, Vol. 225 pp. 135-141.

Sundin K.A., Leeming R.L., and Stenstrom T.A.B. (1999) "Degradation of faecal sterols in urine for assessment of faecal cross-contamination in source-separated human urine and urine storage tank sediment", Water Research, Vol. 33 (9) pp. 1975-1980.

Szargut J., Morris D. R. and Steward F.R. (1988) "Exergy analysis of thermal, chemical, and metallurgical processes" Hemisphere – New York, ISBN 0-89116-574-6.

Tagesanzeiger, 2001, Swiss daily newspaper May 10 2001 (http://www.tages-anzeiger.ch/)

Tchobanoglous, G. (1991) "Wastewater Engineering: Treatment disposal and reuse (Metcalf &Eddy Inc.)", 3rd ed, McGraw-Hill.

Ternes T. A., Stumpf M., Mueller J., Haberer K., Wilken R.-D. and Servos M. (1999) "Behavior and occurrence of estrogens in municipal sewage treatment plants I. Investigations in Germany, Canada and Brazil." The Science of the Total Environment, Vol. 225 pp. 81- 90.

Ternes T.A., Kreckel P. and Mueller J. (1999) "Behaviour and occurrence of estrogens in municipal sewage treatment plants II. Aerobic batch experiments with activated sludge." The Science of the Total Environment, Vol. 225 pp. 91-99.

Udert, K, (2000) Personal communication.

Ueno Y. and Fujii M. (2001) "Three years experience of operating and selling recovered struvite from full-scale plant", Environmental Technology, Vol. 22 (11) pp. 1373 – 1381.

UNEP - United Nations Environment Programme Industry and the Environment (1998), "Mineral Fertilizer Production and the Environment" Technical report No 26 part 1, pp. 17.

Valsami-Jones E. (2001) "Calcium phosphate precipitation" SCOPE Newsletter No. 41 pp. 8 – 15.

Van Veldhuizen H.M., van Loosdrecht M.C.M and Heijnen J.J. (1999) "Modelling biological phosphorus and nitrogen removal in a full-scale activated sludge process", Water Research, Vol. 33 (16) pp. 3459 – 3468.

Van Loosdrecht M.C.M., Kuba T., Van Veldhuizen H.M., Brandse F.A. and Heijnen J.J (1997) "Environmental impacts of nutrient removal processes: Case study", Journal of Environmental Engineering, Vol. 123 (1) pp. 33 – 40.

Van der Gaag F.J. (1987) "ZSM-5 type zeolites: Synthesis and use in gasphase reactions with ammonia", PhD Thesis, Delft University of Technology.

Van Nieuwenhuizen A.F., van der Graaf J.H.J.M. and Mels A.R. (2000) "Direct influent filtration as pretreatment step for more sustainable wastewater treatment systems", Proceedings of the first World Water Congress of the IWA, Paris.

Varis O. and Somlyódy L. (1997) "Global urbanization and Urban Water: Can sustainability be afforded?", Water Science and Technology, Vol. 35 (9) pp. 21 – 32.

Verhoek J.A. (2001) Gelderland manure society - Personal communication.

Veldkamp R.G. and Wiggers J.B.M. (1997) "A statistical approach to pollutant emissions from combined sewer systems", Water Science and Technology, Vol. 36 (8-9) pp. 95–100.

Von Münch, E. and Barr, K. (2001) "Controlled struvite crystallisation for removing phosphorus from anaerobic digester sidestreams", Water Research, Vol. 35 (1) pp. 151 – 159.

Von Munch E., Benesovsky-Scott A. J. Josey J. and Barr K. (2001) "Making a business from struvite crystallisation for wastewater treatment: turning waste into gold", Proceedings from the 2nd International Conference on Phosphate Recovery, Noordwijkerhout, The Netherlands.

Web b K.M. and Ho G.E. (1992) "Struvite solubility and its application to a piggery effluent problem" Water Science and Technology", Vol. 26 pp. 2229 –2232.

Webb K.M. and Ho G.E. (2001) "Characterisation of waste solutions to determine optimised P recovery", Environmental Technology, Vol. 22 (11) pp. 1303-1312.

Wevers J.M.A.R., Kars H. and Schuiling R.D. (1981) "A struvite occurrence in Amsterdam" Bulletin of Mineralogy, Vol. 104 pp. 686 – 689.

Wijst, M. van der, Groot-Marcus, A.P. (1998) "Huishoudelijk afvalwater: Berekening van de zuurstofvraag", Huishoud en Consumentenstudies onderzoeksrapport 1, Landbouwuniversiteit Wageningen (STOWA report 98 - 40).

Williams S. (1999) "Struvite prcipitation in the sludge stream at Slough Wastewater Treatment Plant and opportunities for phosphate recovery", Environmentl Technology, Vol 20 (7), pp. 743 – 748.

Witters H.E., Vangenechten C. and Berckmans P. (2001) "Detection of estrogenic activity in Flemish surface waters using an in vitro recombinant assay with yeast cells", Water Science and Technology, Vol. 43 (2) pp. 117–123.

Wolgast M. (1993) "Clean Water – thoughts about recycling", Creanom HB, Uppsala.

Woods N.C., Sock S.M. and Daigger G.T. (1999) "Phosphorus recovery technology modeling and feasibility evaluation for municipal wastewater treatment plants", Environmental Technology, Vol. 20 (7) pp. 663 – 679.

Zoltek J. (1974) "Phosphorus removal by orthophosphate nucleation" Journal of Water pollutionControl Vol. 46 pp. 2498 – 2520.

List of internet references

i.01   http://www.cbs.nl/nl/cijfers/kerncijfers/index.htm
i.02   http://www.computersrepair.com/aquariumsource/seawater.htm
i.03   http/www.novaquatis.eawag.ch
i.04   http://www.kraftwerk1.ch/
i.05   http://www.novaquatis.eawag.ch/nomix12slim/nomix.html
 or,   http://www.novaquatis.eawag.ch/   (click on NoMix-Tool)
i.06   http://www.otterwasser.de
i.07   http://www.tu-harburg.de/aww/   (Publikationen zum downloaden)
i.08   http://www.otterwasser.de/homee.htm
i.09   http://www.flintenbreite.de/de/wasser2.html#was4
i.10   http://www.swedenviro.com/wrs_en.html
i.11   http://www.swedenviro.com/ver_en.html
i.12   http://www.compostingtoilet.org/6OWNBLD/dblv.htm
i.13   http://www.alascanofmn.com/testing.html
i.14   http://www.wost-man-ecology.se/index.html
i.15   http://www.dubbletten.nu/english-presentation/WCdubbletteneng.htm
i.16   http://www.roevac.com
i.17   http://www.ae.iastate.edu/Ae573_ast475/Slides/msu_fecal-urine_separator.htm
i.18   http://www.waterless.com/
i.19   http://www.americanwaterless.com/index.html
i.20   http://www.waterless.com/Ecotrap.htm
i.21   http://www.restrooms.org/urinals.html
i.22   http://www.universeum.se/index.asp?LangID=2
i.23   http://www.nhm.ac.uk/mineralogy/phos/
i.24   http://www.hydro.com/
i.25   http://www.dsm.com/dag/nl/
i.26   http://www.rotem-group.com/
i.27   http://www.kemira.com/
i.28   http://www.efma.org/
i.29   http://www.books.smenet.org/imar-ch67-sc00-pt00-bod.htm#refer
(PHOSPHATE ROCK Bartels J.J. and Gurr T.M. Online Books main page, Industrial minerals
and rocks, The Society for Mining, Metallurgy and Exploration Inc.)
i.30   http://www.thermphos.com
i.31   http://www.nhm.ac.uk/mineralogy/phos/p&k217/steen.htm
i.32   http://www.usgs.gov/
i.33   http://www.brandtconsolidated.com/infocenter/news_natgas.htm
i.34   http://www.hhdelfland.nl
i.35   http://www.rivm.nl/
i.36   http://www.kunstmest.com/statistiek.html
i.37   http:/www.cbs.nl/nl/cijfers/kerncijfers/lmi0031a.htm
i.38   http:/www.agris.be/nl/
i.39   http://www.agric.wa.gov.au/agency/Pubns/farmnote/1993/f01293.htm
        (Phosphoric acid price per kg P - Australia)

List of figures

List of tables

Appendix 1

Mineral concentrations in human urine

Urine parameter (per person)		Assumed value	Average value	Reference 1 Lehnartz, 1943	Reference 2 Beck, 1971	Reference 3 Strand, 1978	Reference 4 Ciba Geigy, 1977	Reference 5 Berne and Levy, 1998	Reference 6 Guyton, 1991	Reference 7 Wright, 1965	Reference 8 Houssay, 1955
Volume	l/d	1,25	1,28	1,35	1,2	1,2	1,25	1,5	1,5	1	1,25
pH	-		6,09	6	5,5		6,17	6			
P	mg/d	1000	878	1338	489	816	1400	775	420	1200	655
	mg/l		720	991	408	680	1120	517	280	1200	524
Urea	mg/d		30000	30000	25000	30000	30000			30000	30000
	mg/l		25750	22222	20833	25000	24000			30000	24000
NH$_4$ - N	mg/d		813		700			1050			576
	mg/l		581					700			461
N- total	mg/d	12000	11500				11500				
	mg/l	9600	9200				9200				
K	mg/d		2968	2700	2000	3300	2700	4095	3510	2200	2000
	mg/l		2297	2000	1667	2750	2160	2730	2340	2200	1600
Na	mg/d		4824	5900	4000	5897	5150	4483	4414	5000	4000
	mg/l		3861	4370	3333	4915	4120	2989	2943	5000	3200
Ca	mg/d		234	500	150	300	233	360	114	200	200
	mg/l		185	370	125	250	186	240	76	200	160
Mg	mg/d		196	400	150	100	130	328	273		150
	mg/l		142	296	125	83	104	219	182		120
total COD	mg/d	12000	12000				12000				
	mg/l		9600				9600				
TOC	mg/d		5000				5000				
	mg/l		4000				4000				
Cl	mg/d		6947	8900			4750	6913	7125	9000	
	mg/l		5540	6593			3800	4609	4750	9000	
S	mg/d		1194	800	667	833	442			2700	800
	mg/l		1097	593	556	694	353			2700	640

Appendix 2

Short term scientific mission to EAWAG, Switzerland.

Urine separation technology

- Introduction
- Sustainable sanitation alternatives
- Pilot installations at EAWAG
- Evaluation of Consumer attitude
- Life Cycle Analysis and Eco-efficiency
- Anthropogenic Nutrient Solution (human urine) treatment
- Urine Separation and Combined Sewer Overflows
- Evaluation of Anthropogenic Nutrient Technology
- Aspects of mutual interest to Novaquatis and TU Delft
- Communication and future co-operation

1.   Introduction

STOWA (Foundation for applied water research) appointed the Delft University of Technology to investigate various possibilities for recovery of minerals by means of urine separation. The investigation was started on August 01, 2000 and will be completed by the end of July 2001. The primary objective of this investigation is compilation of inventories of existing knowledge and technology.

Jac Wilsenach visited EAWAG (Swiss Federal Institute for Environmental Science and Technology) from 25/09/2000 to 29/09/2000 in Duebendorf (Zurich). Within EAWAG, the Novaquatis group is doing basic research for implementation of urine separation technology in Switzerland. Mr Wilsenach gave a presentation on wastewater treatment in the Netherlands in general, after which he highlighted his own objectives for further investigation. From this presentation various discussions followed, with people in the Novaquatis group, as well as with people working on other aspects of wastewater treatment. The most important of these are discussed in this report.

Differences between the situation in Switzerland and the Netherlands had to be identified. Three important differences are:
1. Switzerland has an agreement with the European community to discharge a certain maximum limit of phosphorus and nitrogen into receiving waters. Biological phosphorus and nitrogen removal is not mandatory for wastewater treatment plants in Switzerland, other than in the Netherlands.
2. Switzerland has hills and mountains, while the Netherlands is flat. This is manifested in significantly different sewer reticulations.
3. EAWAG and TU Delft is at this stage involved with different levels of research regarding urine separation:
   - Within EAWAG, Novaquatis does detailed research into specific aspects of urine separation, funded by a government institution, aimed at reducing Switzerland's impact on surface waters
   - TU Delft is compiling inventories of possibilities for a body of non-governmental organisations (represented by STOWA) aimed at reclaiming (useful) minerals from wastewater.

- The concept of water treatment boards ("zuiveringschappen") is foreign to Switzerland, where water treatment is regulated by the government. Water treatment boards are responsible for wastewater treatment in the Netherlands.

2.   Sustainable sanitation alternatives

Sustainability has many different faces. Tove Larsen (Novaquatis project leader) approaches this issue practically and with fresh insights to consider complete cycles of mass flow. In order to develop sustainable urban water management, we must first define the services to provide, before fixing our minds on improving existing technology. The prime objective of sanitation still is urban hygiene and health. Other objectives are drainage (prevention of floods) and integration of agriculture into urban water management. New technologies must aim to make use of available resources, of which existing infrastructure might be one, without exporting the problems of pollution into time and space.

Sanitation consumes energy. According to the United Nations Environment Programme, the total global energy consumption rate is close to 2 kW/capita. The 2000 W Gesellschaft (Dieter Imboden) at ETH (Zurich) sets this as a limit for sustainable energy use. The consumption rate in Switzerland is (similar to the rest of Western Europe) just under 6 kW/capita.

However, wastewater reticulation and treatment in Switzerland consumes energy at a rate of only 5.7 W/capita (or 0,1% of the total). In comparison to this figure, the rate of energy consumption for heating of water is approximately 140 W/capita. It seems that more energy efficient treatment alternatives would make no real impact on global energy use. Research into alternative sanitation and wastewater treatment technology should focus on improving the quality of receiving surface waters and on dealing with hazardous wastes.

Separate collection and treatment of urine, is one way in which the food-nutrient-water-treatment dynamic can be changed to improve the sustainability of the whole system. The separated urine could be used to replace industrial fertiliser, as has been done in Swedish pilot projects. There are, however, still a number of unanswered questions.
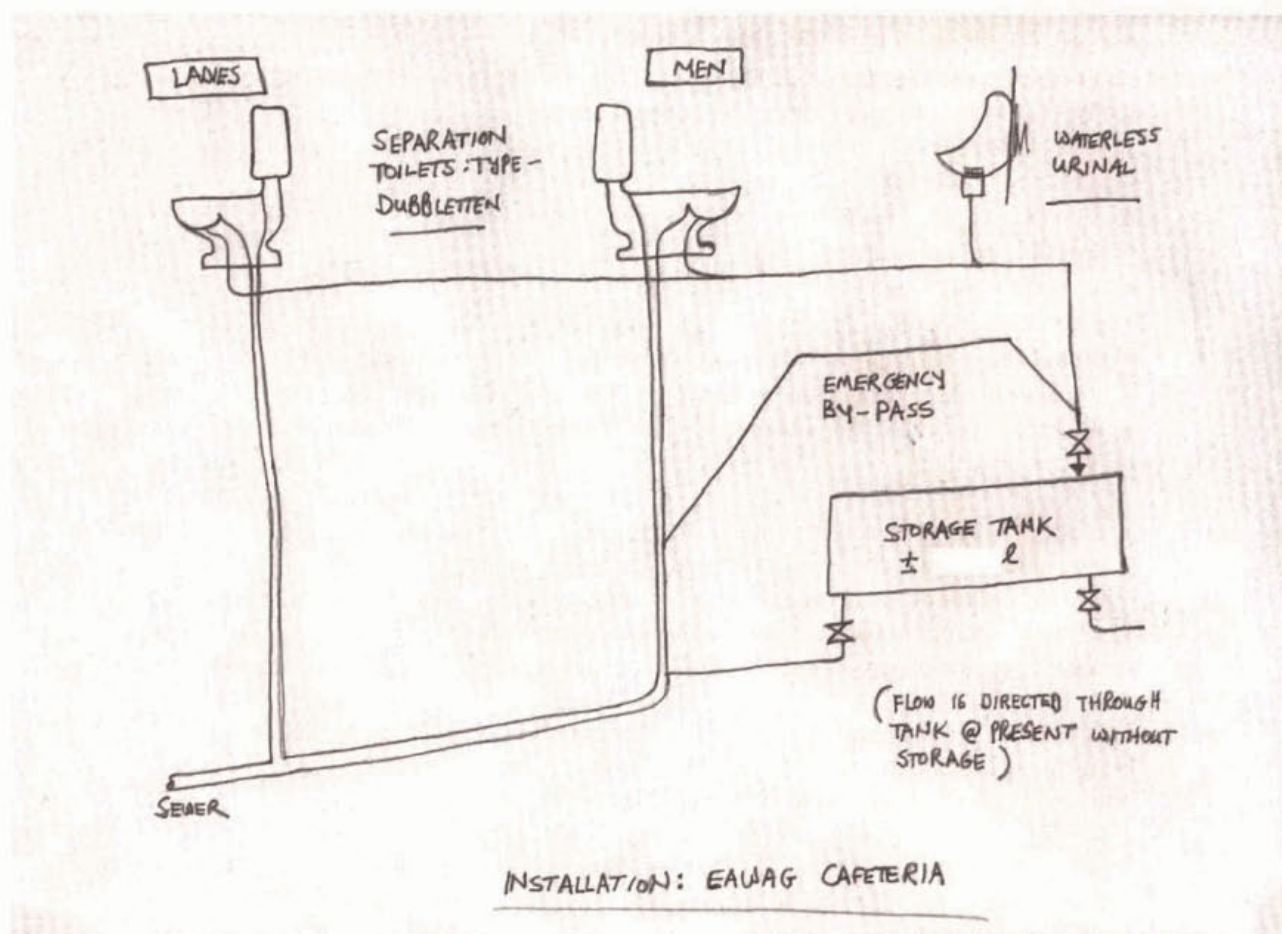
3.   Pilot installations at EAWAG

Two kinds of separation toilets have been installed at EAWAG. Kai Udert installed the first toilet in 1997 to collect urine for experiments on urine treatment. This installation is discussed in more detail in chapter 6 of this report.

A new advanced version of the separation toilet, DUBBLETTEN (refer attachment and http://www.kvarken.org/projekt/ratan/gh/) was installed in August 2000 at the central EAWAG cafeteria. A waterless urinal was also installed in the men's room. Refer attached layout diagram.

Both the separation toilet and the waterless urinal were used during the visit. The following comments relate:
- no smells, other than expected, was detected
- very little water is used for the flushing of urine in the separation toilet
- men need to sit down to enable separation and avoid splatter
- "aiming" is not necessary, as the toilet design appreciates the design of the human body
- a colour brochure is attached to the back of the toilet door, giving users a concise introduction to the philosophy and principals of urine separation.
- counters were installed for flushing, and the back and front compartments of the separation toilet was used equally frequent (± 180 each at the time).

LADIES · **MEN**

SEPARATION
TOILETS : TYPE —
DUBBLETTEN

WATERLESS
URINAL

EMERGENCY
BY-PASS

STORAGE TANK
± ____ ℓ

(FLOW IS DIRECTED THROUGH
TANK @ PRESENT WITHOUT
STORAGE)

SEWER

INSTALLATION: EAWAG CAFETERIA

The separately collected urine flows into a storage tank (300 litre) to be directly discharged into the existing sewer, with normal sewerage. It is, however, installed in such a way that removal of stored urine by other means (e.g. trucks, or release during no-flow times), would be easy.

These separation toilets and waterless urinal were mainly installed to:
- introduce the new technology to the broad public,
- and at the same time evaluate the consumer attitude towards the new technology.

The installation could also serve as source of technical data. Volumes of water used for flushing, volume of urine collected, amount of dilution, ratio of people visiting the cafeteria to people using the toilets etc. could be measured, but at present there is nobody who could do this extra work.

4.   Evaluation of Consumer attitude

Focus groups of volunteers (±100 in total) will meet from November 2000 to January 2001 at EAWAG. These groups, from all age groups and professions, although comprising non-specialists, will focus on specific aspects of urine separation technology. The subjects to be discussed will include

- People, and their need to excrete (with the focus on urine)
- Possibilities of urine collection
- Current wastewater treatment practices
- Current condition of receiving surface waters
- Possibilities for transport and treatment (into fertiliser) of urine
- Agriculture recycling urine as fertiliser, and possible side effects
- Evaluation of the system of urine separation, and how to change from the current systems to new technology.

The information gathered from these discussions will be used to
Analyse behavioural routines and social norms surrounding toilet use
Highlight preferences and prejudices regarding the no-mix toilet's characteristics
Analyse the perceptions regarding risks of micro-pollutants
Evaluate consumer perception on risks and benefits of urine separation technology.

Information is made available through the Internet to invite public participation.
http://www.novaquatis.eawag.ch/NoMixINFO.pdf and
http://www.novaquatis.eawag.ch/nomix_10/nomix.html

Similar research has been done in Sweden, there are unique features of the Swiss focus groups, and these are expected to be influential to the results of focus groups:

- "Eco-sanitation" is much more politically driven in Sweden than in Switzerland
- The no-mix toilet technology is still improving and technical imperfections must be pointed out by users

Although the final results are due for the end of 2003, intermediate results might be available earlier, which might be used to identify central problems.

5.   Life Cycle Analysis and Eco-efficiency

Two different methodologies are proposed in general for conducting life cycle analyses. The first propose focussing on the global environmental impacts (e.g. such as the greenhouse effect) of any action or process. The second proposes assessment of local effects, because these differ in different regions/countries (e.g. such as acid rain in Scandinavia, where the effect is much different than Slovenia due to soil conditions). This is however difficult to asses with "classical" life cycle assessment techniques.

One of the drawbacks of life cycle analysis, is the vast amount of information/data that must be captured and processed. Data compiled in a "cradle-to-grave" inventory can often comprise up to 500 substances. Apart from the inventory, the system boundaries must be defined and the environmental impact must be assessed, often by (ecological) specialists.

One way out of this debate is through the concept of eco-efficiency. By making total mass balances explicit, we can get an indication of the ecological footprint. New environmental problems might commence with introduction of higher effluent standards in a wastewater treatment plant for example. Eco-efficiency should involve scrutiny of all emissions involved in changing a system (comparing the status quo system with the proposed replacement)

Katrin Hügel was at the time finishing a PhD on Life cycle analyses and eco-efficiency of wastewater treatment works (ETH DISS Nr. 13913, Oekobilanzen in der Siedlungswasserwirtschaft).

6.   Anthropogenic Nutrient Solution (human urine) treatment

Three problems occur with the use of urine as fertiliser.
- Smells (resulting from ammonia emissions and organic compounds)
- Ammonia emissions leads to loss in efficiency
- Concerns about pharmaceuticals and hormones in urine

Smell, hygiene and toxicity are the themes relevant for process engineering. The research on new techniques is only starting, and most projects are of an explorative nature.

6.1   Storage, treatment and transport of urine

Kai Udert has been working on three aspects of urine handling for almost four years.

- Prevention of ammonia stripping by oxidation of nitrogen
- Precipitation of minerals from urine compounds
- Urine flows in sewers

Prevention of ammonia stripping is done in a two step biological process similar to the SHARON process. The process is illustrated in the attached schematic diagram.
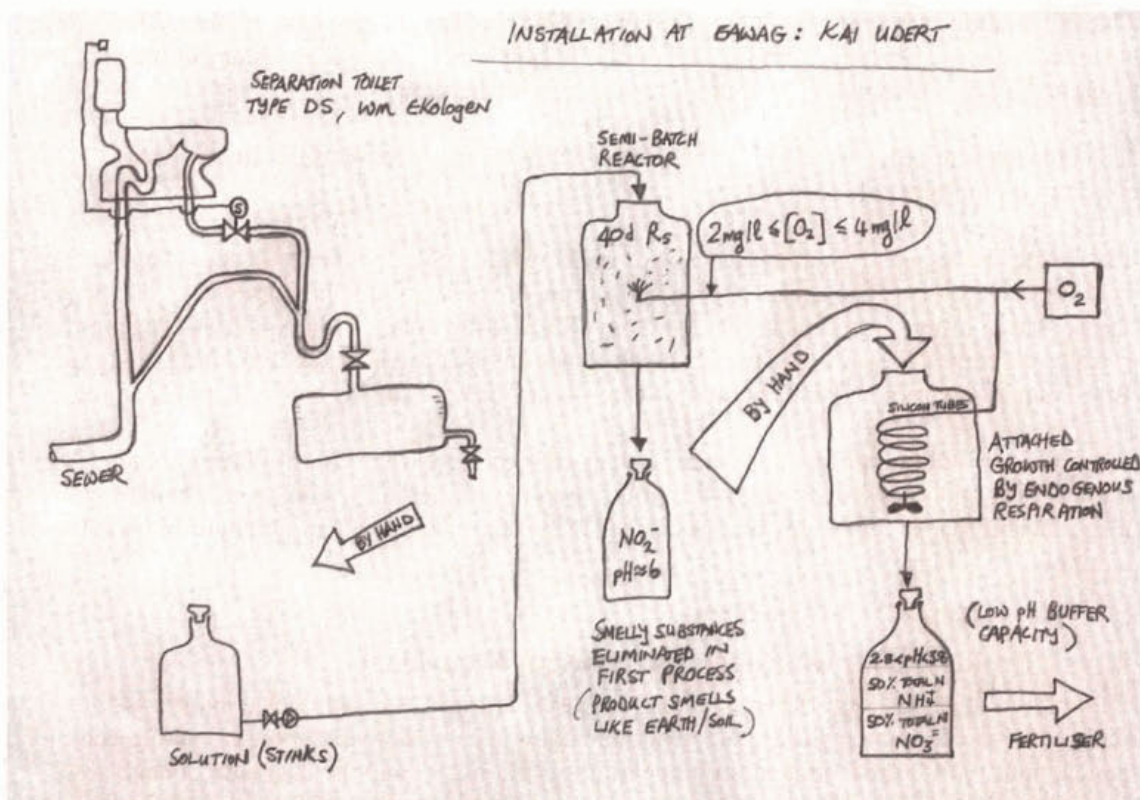
The advantage of the process is that no acid is added to lower the pH (that prevents ammonia stripping). Adding acid presents the problem of neutralising the solution again before use. Although the pH in the end product drops to between 2.8 and 3.8, the buffer capacity is very

low. This implies a rise in pH shortly after contact with soil, if used as fertiliser. The end products are 50% NH4+ and 50%NO3-.

6.2 Removal of phosphorus from solution

A few projects are planned to gain knowledge the biochemistry of phosphorus. This will include a study of precipitation and a study on avoiding high pH.

As yet, the framework of what "sustainable alternatives" would be is not defined. The question will be approached from two sides
- evaluate the current technology, to determine what is "permitted" in terms of material investment and energy consumption.
- search techniques fitting within this "permitted framework" and give feedback where it's necessary to redefine such goals.

Basic research on the kinetics of chemical reactions with phosphorus is also planned. More ideas include biological micro-pollutant removal and membrane technology.

Max Maurer was recently appointed in Novaquatis to head this research.

7. Urine Separation and Combined Sewer Overflows

Swiss surface waters mostly run rapidly. In the turbulence, high volumes of oxygen can be mixed into the water, so that the oxygen level in water is normally close to the saturation level. The general public relates "dirt" to pollution and the water with high oxygen levels seems to be "clean". Sediment in sewers is most often present as slowly biodegradable matter. When sewer sediments are re-suspended and washed out during high flow conditions (after rainstorms), the chemical oxygen demand of the water is not increased dramatically.

The biggest problem with combined sewer overflows in many Swiss cities/towns is the associated high level of erosion.

Fish deaths have been linked to presence of ammonia in water under certain conditions and at certain temperatures. The cause-effect relationship, however, of combined sewer overflow and the depletion of the receiving surface water quality is not known. Municipalities use empirical rules to determine the total volume of retention-settling basins. This is based on retaining the first 2mm of any rain event. Retention-settling basins provide $20m^3$/ha of retention volume.

According to Wolfgang Rauch, standards on combined sewer overflows in Denmark are very high. This might be more similar to the situation in the Netherlands. The Danish landscape is also more similar to the Netherlands and CSO occurrences and loads might therefor be more similar.

8. Evaluation of Anthropogenic Nutrient Technology

Economists make use of different techniques to evaluate and assign values to entities with indirect or no economic value. Cost-benefit analysis is used more and more often to reach conclusions with far reaching environmental implications. Economic cost benefit analyses is sometimes used to ascribe absolute values to abstract subjects, such as the value of fresh air, clear water, healthy soil, etc. These values change with time and are highly subjective, dependant on personal opinion and often biased.

Irene Peters discards "blind use" of this methodology for the evaluation of anthropogenic nutrient technology (urine separation technology). The technique of scenario analyses is

proposed instead. Two model scenarios are planned for the introduction of separation technology:

1. Winterthur, representing an urban environment, also housing one of the largest wastewater treatment works in Switzerland
2. Ergolz I (in Kanton Basel Landschaft), representing a rural, or semi-urban, environment

For the two different environments, the present state of affairs will be compared to a changed status quo after introduction; as well a changes brought about due to implementation, of separation technology. Information of base conditions (datum scenario) will be collected and compiled in an inventory by Christian Fux.

The ultimate question is does urine separation technology constitute an overall improvement on current technology in the two model scenarios?

Three main themes along which evaluation of this question will be guided are:
1. Nutrient economy, with overall environmental and economic impacts: Recycling of P and N from urine to agriculture against industrial fertiliser and future problems regarding heavy metals in P sources.
2. Urine separation technology: Introduction of new technology and changes in existing technology. Changes from large centralised wastewater treatment works to smaller urine treatment plants. Aspects of transport and changes in mass flows. To do justice to this question, a dynamic simulation is necessary, but as this is too complex (as it would ultimately have to include issues such as replacement of existing infrastructure), comparison of steady state scenarios must suffice.
3. Impacts on receiving waters: Reduction of daily ammonia peaks and CSO ammonia emissions. Elimination of majority of pharmaceuticals and hormones in surface water.

The information will strongly depend on findings of different NOVA work packages.

9. Aspects of mutual interest to Novaquatis and TU Delft

9.1 Novaquatis's interest in TU Delft's work

- Novaquatis is not looking at possibilities of urine recycling other than agricultural fertiliser. The TU Delft's project's title reflects the importance of reclaiming minerals. This implies more than fertiliser only, such as the possible recycling of phosphorus at Thermphos does.
- Novaquatis is not considering urine separation as cheap alternative to conventional wastewater treatment per se. The dynamic modelling of effects and costs on wastewater treatment works due to reduced influent loads of P and N (planned by TU Delft) might present interesting insights.
- Theory on combined sewer overflows in the Netherlands doesn't consider ammonia emissions. Reasoning behind this and possible effects of urine separation on combined sewer overflows will be investigated into more detail by TU Delft.
- The use of land and land shortage is a serious problem in the Netherlands. The influence of urine separation on land requirements (and possible reduced sizes of treatment works) in the Netherlands could be extrapolated to Swiss situations, where land is also scarce. It has to be kept in mind that Dutch wastewater treatment works are significantly larger than Swiss counterparts, due to amongst other tings, the long sludge ages in the Netherlands.

9.2 TU Delft's interest in Novaquatis's work

- Results from focus groups and the evaluation of consumer attitude will not fit into the time frame of TU Delft's investigation. These results are unique for Swiss situations, but central problems could be identified, which might also apply to the Netherlands
- Irene Peters guides comprehensive evaluation of the anthropogenic nutrient technology at Novaquatis. Again, such a detailed study will not fit into the time frame of TU Delft's investigation. The results would be unique for the two Swiss model regions. Aspects such as types of treatment works, urine transport (which could in the Netherlands not be done via pipelines due to the flatness of landscape) would have t be taken into consideration when drawing conclusions for the Netherlands from the Swiss results. There could, however, be a lot of useful information and insights in both the methodology and findings.
- Research on urine treatment to prevent ammonia stripping, performed by Kai Udert, is similar to the SHARON process, developed in Delft. Further discussion between the two groups is expected and the final PhD thesis is eagerly awaited.
- Max Maurer is initiating research on phosphorus removal or treatment at the moment. The scope of research projects under his guidance is still vague, but should take form shortly. Contact between the two groups must be maintained to relay information and results on aspects such as urea hydrolysis, kinetics of P-precipitation, avoiding high pH values, sterile filtration, micro-pollutants etc
- Rik Eggen's research on micro-pollutants wasn't discussed during the visit, but his results will certainly have implications for the Netherlands as well. Rik Eggen and Beate Escher, together with PhD Students Erika Vye and Zacharias Schreiber do research on micropollutants.
-

10. Communication and future co-operation

The difference in research focus (mentioned in the introduction) could stimulate thought around questions such as:

- what is the economic value of reclaiming minerals?
- what are the environmental benefits of urine separation?

Although Novaquatis has a head start of more than one year on TU Delft and more research personnel, some of their programmes are also just starting. Communication must be continued to encourage co-operation and prevent duplication of work.

To maintain a central body of knowledge, all further communication between TU Delft and Novaquatis should pass through Jane Muncke and Jac Wilsenach. Where direct questions/answers of a specialist technical nature is involved, these could be addressed to persons involved, with a copy circulated to Jane and Jac.

Appendix 3

Changing wastewater concentration with increasing urine separation efficiency.

Raw wastewater				Urine separation percentage							
				10	25	50	75	85	90	95	100
Population		45000									
WW production	l/capita	300		296	291	282	273	269	267	266	264
Flow rate	(ML/d)	13.500		13.337	13.092	12.684	12.277	12.113	12.032	11.950	11.869
	(m³/h)	563		556	546	529	512	505	501	498	495
total P influent	mg/l	8		7.8	7.4	6.7	6.0	5.8	5.6	5.5	5.3
	kg/d	108		104	97	86	74	70	68	65	63
total N influent	mg/l	50		46.6	41.2	31.9	22.0	17.8	15.7	13.6	11.4
	kg/d	675		621	540	405	270	216	189	162	135
ammonia N in	mg/l	40		36	31	21	11.0	6.7	4.5	2.3	0
	kg/d	540		486	405	270	135	81	54	27	0
COD influent	mg/l	537		539.5	543.4	550.2	557.5	560.6	562.1	563.7	565.3
	kg/d	7250		7196	7115	6980	6845	6791	6764	6737	6710
COD distribution											
SUM	1.00	537	1.00	539.5	543.4	550.2	557.5	560.6	562.1	563.7	565.3
Sa Fermentation prod	0.12	64		65	66	69	71	72	72	73	73
Sf Fermentable, readi	0.13	70	0.88	67	63	56	48	44	43	41	39
Si Soluble inert organ	0.06	32	0.12	32	32	32	31	31	31	31	31
Xi Particulate inert org	0.25	134		136	138	143	148	150	151	152	153
Xs Slowly biodegrada	0.44	236	0.00	239	244	251	260	263	265	267	269