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ANAEROBIC TREATMENT OF CONCENTRATED WASTEWATER IN DESAR CONCEPTS

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PREFACE

Het innovatieve sanitatieconcept DESAH (Decentrale Sanitatie en Hergebruik) trekt recentelijk veel aandacht als revolutionaire aanpak in relatie tot het huidige sanitatie status-quo. De belangrijkste reden hiervoor is de duurzame logica achter de keuzes van het sanitatieconcept. In zijn simpelste uitvoeringsvorm kan het resulteren in een verbetering van de situatie in minder welvarende landen waar geen enkele vorm van sanitatie aanwezig is. In meer ontwikkelde landen kan het nieuwe sanitatieconcept worden overwogen wanneer er noodzaak bestaat om de oude infrastructuur te vervangen of wanneer bijvoorbeeld nieuwe wijken of kantoren worden gebouwd.

Het DESAH concept kent vele variatiemogelijkheden en daarmee is er geen universeel DESAH concept dat past in elke situatie. Er zijn echter elementen die identiek zijn voor alle scenario's. Eén element is dat afvalwaterstromen gescheiden worden ingezameld op basis van bijvoorbeeld de vervuilingsgraad, het type vervuiling en het hergebruikpotentieel van grondstoffen. Drie soorten grondstoffen zijn te onderscheiden: bio-energie (via de omzetting van organisch materiaal), nutrienten (stikstof, fosfaat en kalium) en water (via een geavanceerde nabehandeling van minder vervuilde afvalwaterstromen). De behandeling wordt zo geselecteerd dat het hergebruikpotentieel wordt behouden.

There is no universal DESAR concept, fitting each situation. Several variations may exist within different scenarios. There are however a few elements that are the same for all these scenario's. One of these elements is that wastewater streams are separated according to their degree and type of pollution and reuse potential of resources. Three main resources are considered: bio-energy (originated from transformation of organic material), nutrients (nitrogen, phosphorus as main nutrients but also potassium and sulphur) and water (produced after advanced treatment of cleaner wastewater streams). Treatment of wastewater streams is selected in such a way that reuse potential is preserved.

Author of this report is Katarzyna Kujawa – Roelveld (WUR). The project steering group consisted of: ir. Harm Baten (Hoogheemraadschap van Rijnland), ir. Elbert Majoor (Waterschap Veld en Vecht), ir. Ruud Schemen (Hoogheemraadschap van het Hollands Noorderkwartier) en Bert Palsma (STOWA).

De technieken zijn beschikbaar, de toepassingen zijn veelbelovend; wij zijn hard op zoek naar mogelijkheden om praktijkervaring op te doen.

Utrecht, oktober 2005
De directeur van de STOWA
Ir. J.M.J. Leenen

SAMENVATTING

Het innovatieve sanitatieconcept DESAH (Decentrale Sanitatie en Hergebruik) trekt recentelijk veel aandacht als revolutionaire aanpak in relatie tot het huidige sanitatie status-quo. De belangrijkste reden hiervoor is de duurzame logica achter de keuzes van het sanitatieconcept. In zijn simpelste uitvoeringsvorm kan het resulteren in een verbetering van de situatie in minder welvarende landen waar geen enkele vorm van sanitatie aanwezig is. In meer ontwikkelde landen kan het nieuwe sanitatieconcept worden overwogen wanneer er noodzaak bestaat om de oude infrastructuur te vervangen of wanneer bijvoorbeeld nieuwe wijken of kantoren worden gebouwd.

Het DESAH concept kent vele variatiemogelijkheden en daarmee is er geen universeel DESAH concept dat past in elke situatie. Er zijn echter elementen die identiek zijn voor alle scenario's. Eén element is dat afvalwaterstromen gescheiden worden ingezameld op basis van bijvorbeeld de vervuilingsgraad,

het type vervuiling en het hergebruikpotentieel van grondstoffen. Drie soorten grondstoffen zijn te onderscheiden: bio-energie (via de omzetting van organisch materiaal), nutrienten (stikstof, fosfaat en kalium) en water (via een geavanceerde nabehandeling van minder vervuilde afvalwaterstromen). De behandeling wordt zo geselecteerd dat het hergebruikpotentieel wordt behouden.

Separaat inzamelen van verschillende afvalwaterstromen op huishoudelijk niveau is een duidelijk uitgangspunt van het DESAH concept. Meestal worden twee afvalwaterstromen onderscheiden: zwartwater en grijswater. Zwartwater betreft doorgaans het mengsel van feces en urine die beide met een zekere hoeveelheid schoon water worden weggespoeld voor transport. Grijswater is een verzamelnaam voor de overige afvalwaterstromen die in een huishouden worden geproduceerd zoals van bad, douche, wasmachine en afwasmachine. Feces en urine representeren een zeer geconcentreerd mengsel van organisch materiaal en nutriënten (waardevolle componenten). Ze bevatten echter ook pathogenen, pharmaceutische componenten en hun metabolieten en endocrien verstorende stoffen (schadelijke componenten). Het is raadzaam het zwartwater geconcentreerd te houden om duurzame behandelingsopties te kunnen toepassen en het voordeel te nemen van zijn hergebruikpotentieel. Door het concentreren van alle potentiële risico's in een zeer gering volume kan een betere controle en een beperking (eliminatie) van de negatieve effecten op het oppervlaktewater worden verkregen.

Voor een *maximale terugwinning en het hergebruik van grondstoffen* wordt in het gepresenteerde concept gesuggereerd dat ook de vaste fractie van het keukenafval wordt ingezameld, getransporteerd en behandeld met het zwartwater. De vaste fractie van het keukenafval bevat een organische vracht (kgCOD.persoon^{-1}dag^{-1}) vergelijkbaar met het zwartwater en is gemakkelijk afbreekbaar. Ofschoon de samenstelling niet hetzelfde is in vergelijking met keukenafval in huishoudens, is in het onderzoek om praktische redenen keukenafval van een ziekenhuis gebruikt. Op dit moment wordt in Nederland het keukenafval nog apart ingezameld en samen met het tuinafval voor het grootste deel gecomposteerd.

Omdat zwartwater als meest problematische stroom wordt gezien, heeft hier in het onderzoek de nadruk op gelegen. Afhankelijk van het toilet als inzamelings- en transportsysteem, komen twee verschillende technologieën aan de orde, anaërobe vergisting of compostering. Belangrijk argument voor de keuze is het resulterende vochtgehalte dat dus wordt bepaald door het geselecteerde toiletsysteem. Anaërobe vergisting van geconcentreerd afvalwater wordt beschouwd als een duurzame optie omdat energie kan worden geproduceerd terwijl de nutriënten behouden blijven voor hergebruik. Daarom is deze technologie in het onderzoek intensief onderzocht. Twee types geconcentreerd afvalwater zijn anaëroob vergist: zwartwater met en zonder keukenafval. In het zwartwater is verder onderscheid gemaakt tussen ‘klassiek’ zwartwater bestaande uit hoeveelheden feces en urine die dagelijks door een individu worden geproduceerd en bruin water dat voornamelijk bestaat uit feces.

De haalbaarheid van anaërobe vergisting voor de behandeling van geconcentreerd zwartwater en keukenafval en zijn effect op het totale DESAH concept was de hoofdvraag van dit onderzoek. Deze haalbaarheid is uitgedrukt in:
- het rendement dat kan worden verkregen in verschillende vergistingconfiguraties bedreven onder verschillende procesomstandigheden;
- de kwaliteit en het hergebruikpotentieel van de verkregen producten.

Het rendement werd beschouwd als voldoende wanneer de bereikte omzettingsgraad in de buurt kwam van de maximale afbreekbaarheid zoals bepaald in afzonderlijke batch experimenten die werden uitgevoerd onder gunstige procesomstandigheden. Stabiliteit was een belangrijke parameter voor het kwalificeren van het slib voor hergebruik. Het anaërobe effluent werd gedetailleerd gekarakteriseerd in functie van verschillende procesconfiguraties en operationele parameters. De vorm en concentratie van de essentiële nutriënten voor vegetatie speelden hierbij een belangrijke rol.

Voor het toiletsysteem werd een vacuüm toilet geïmplementeerd om geconcentreerd zwartwater te verkrijgen. Het spoelwatervolume was 1 l per spoelbeurt in plaats van de conventionele 6 tot 9 l. Hoewel dit geen primair onderzoeksdoel is geweest, maakt toepassing van vacuüm technologie het mogelijk om het schoonwaterverbruik te reduceren. Door de toepassing van vacuüm technologie kan meer dan 20% op het waterverbruik in huishoudens worden bespaard.

Keukenafval werd in de reactoren gebracht via de vacuümtoiletten. Hiermee werd een waterbesparend systeem gesimuleerd voor de inzameling van deze stroom (waterbesparend voedselrestenvermaler). Tijdens de bedrijfsvoering van drie jaar zijn nauwelijks technische problemen voorgekomen en de 1 l water was meestal voldoende om de toiletbak schoon te houden en aanzienlijke hoeveelheden keukenafval weg te spoelen (tot 1 kg nat keukenafval met 1 l water).
Hoewel het geen onderzoeksvraag was, is een juiste keuze van schoonmaakchemicaliën essentieel om er zeker van te zijn dat geen toxische componenten het biologische behandelingsysteem bereiken. Milieuvriendelijke, afbreekbare schoonmaakmiddelen werden gebruikt om het toilet schoon te maken. Tijdens het transport van afval(water) hebben zich geen verstoppingen voorgedaan maar de afstanden tussen het toilet en de reactor waren ook kort.

Het effect van het mengen van de verschillende afvalwaterstromen op de influentsamenstelling is weergegeven in Tabel 1.

TABEL 1 **SAMENSTELLING VAN HET ZWARTWATER, ZWARTWATER + KEUKENAFVAL EN BRUINWATER + KEUKENAFVAL**

Parameter	Eenheid	Zwartwater	Zwartwater + keukenafval	Bruinwater + keukenafval
Totale CZV	$g.l^{-1}$	9,5-12,3[*]	13,3-22,9	31,8-66,0
Gesuspendeerde CZV	$g.l^{-1}$	7,0-9,6	10,3-17,1	23,9-62,0
Totale N	$gN.l^{-1}$	1,0-1,4	1,2-1,7	1,7-1,9
Totale P	$gP.l^{-1}$	0.091-0,144	0.11-0,21	0,23-0,35
Afbreekbaarheid	%	60	60-80	70-85

[*] het bereik afkomstig van de verschillende onderzoeksrondes wordt aangegeven; verschillen zijn veroorzaakt doordat niet altijd de gewenste verhouding tussen zwartwater en keukenafval aanwezig was (theoretisch feces:urine:keukenafval = 1:5:1) door een moeizame bemonstering en door het verschil in spoelwatervolume voor de verschillende toiletten (bijv. 0.9 vs. 1,1 l)

Twee configuraties van anaërobe vergisters werden extensief getest: de accumulatie vergister en de UASB septic tank. Een accumulatie systeem werd gekozen omdat technisch gezien dit een eenvoudig systeem is en geen mechanische onderdelen binnen de reactor bevat. De UASB septic tank werd gekozen omdat het een interessante en goedkope verbetering is van de conventionele septic tank die wereldwijd wordt gebruikt in de niet gerioleerde gebieden.

Een accumulatiesysteem (AC) is een continue gevoede reactor waarin het effectieve (vergisting) volume toeneemt in de tijd. Vergisting en opslag zijn dus gerealiseerd in één volume. Nadat het maximale volume of de vereiste opslagtijd is bereikt, wordt de reactor in één keer leeg gemaakt. Eventueel wordt extra opslag zonder voeding toegepast om het mengsel verder te laten stabiliseren. Het hoofddoel is hergebruik van het vergiste, nutriëntenrijke medium in de landbouw. Daarmee wordt de opslagperiode bepaald door de vereiste frequentie van bemesting. In de Nederlandse situatie is het niet toegestaan om de meststoffen toe te passen in de wintermaanden, zodat 6 maanden opslag is vereist. Naast de eenvoudige reactor zijn verder voorzieningen nodig voor het inzamelen en de verdere behandeling van het geproduceerde biogas. Een meer efficiënte behandeling wordt verkregen bij hogere temperaturen. In een dergelijke situatie zijn ook verwarmingsfaciliteiten en tankisolatie nodig. Na het legen van de reactor dient een minimale hoeveelheid vergist medium als entmateriaal in de reactor achter te blijven voor de volgende run.

De UASB-septic tank ($UASB_{ST}$) is een hybride reactor met een continue voeding van influent en waarin accumulatie van slib optreedt. De hybride reactor combineert eigenschappen van de UASB reactor en de conventionele septic tank. De wijze van opwaartse doorstroming verbetert de verwijdering van het gesuspendeerde materiaal en resulteert in een betere biologische omzetting van opgeloste componenten, zoals in een UASB reactor. Accumulatie van slib is een kenmerkende eigenschap van een septic tank. Door de lange slibverblijftijd wordt een goede stabilisatie van het slib bereikt. Drie producten worden verkregen: een effluent rijk aan nutriënten, gestabiliseerd slib en biogas.

De karakteristieken van de reactoren zoals gebruikt in dit onderzoek zijn weergegeven in Tabel 2. Alleen het accumulatiesysteem is getest voor twee types afvalwater: (1) zwartwater met het keukenafval en (2) bruinwater met het keukenafval (Tabel 1,2). De verkregen resultaten zijn weergegeven in Tabel 3.

TABEL 2 **HET BEREIK VAN DE OPERATIONELE PARAMETERS TOEGEPAST TIJDENS HET TESTEN VAN HET ACCUMULATIESYSTEEM EN DE UASB SEPTIC TANK TIJDENS VERSCHILLENDE ONDERZOEKSFASEN**

Parameter (gemiddeld)	Eenheid	AC Zwartwater + keukenafval	AC Bruinwater +keukenafval	$UASB_{ST}$ Zwartwater
Reactor belasting	Aantal personen	1	1-2	1
Debiet, Q	$l.d^{-1}$	5.3-7.5*	1.8-3.4	7
Spoelwaterverbruik	$l.spoeling^{-1}$	~ 1	~ 1	~ 1
Temperatuur	°C	20	15,20	15-25
Reactor volume	l	1000		200
Hydraulisch verblijftijd/accumulatie tijd	d	115-150	115-280	27-29
Organisch belasting	$kgCOD.m^{-3}.d^{-1}$	0.3	0.1-0.3	0.33-0.42

TABEL 3 **VOORBEELDEN VAN EFFLUENTKWALITEIT VERKREGEN IN EEN ACCUMULATIESYSTEEM VOOR ZWARTWATER + KEUKENAFVAL EN BRUINWATER + KEUKENAFVAL EN VOOR DE UASB SEPTIC TANK VOOR HET ZWARTWATER**

Parameter	Unit	AC zwartwater + keukenafval, 20°C	AC bruinwater + keukenafval, 20°C	UASB septic tank, zwartwater, 25°C
Totale COD	$mg.l^{-1}$	4065*	21900*	2733
Opgeloste COD	$mg.l^{-1}$	820	1200	1376
Vluchtige vetzuren	$mg.l^{-1}$	40	<100	500
Ammonium-N	$mgN.l^{-1}$	1450	1800	1068
Fosfaat	$mgP.l^{-1}$	93	104	59

* gemiddelde concentratie nadat de inhoud van de reactor werd gemengd; het bevat dus vloeistof en het slib.

Anaërobe vergisting van geconcentreerde afvalwaterstromen in een accumulatiesysteem kan worden gezien als een efficiënte (voor-)behandelingsstap. Circa 58% omzetting van het organische materiaal in biogas is verkregen tijdens de vergisting van bruinwater en het keukenafval in de AC-reactor bij 20°C en na 150 dagen. Lange verblijftijden en de juiste keuze van het entmateriaal (type en hoeveelheid) resulteren in een hoge stabilisatiegraad van het vergiste medium. Een extra opslag zonder voeding kan worden overwogen (bijvoorbeeld door twee reactoren te plaatsen) en dit leidt dan tot een volledige stabilisatie van het slib en tot het aanzienlijk inactiveren van pathogenen.

Tijdens het voeden van de AC-reactor vindt een stratificatie plaats. Omdat tijdens het legen van de reactor het mengen van alle lagen niet is te vermijden, resulteert dit in hogere CZV-waarden in het eindproduct (tabel 3). Hoge concentraties ammonium werden gemeten in de reactor en was het hoogst bij de behandeling van bruinwater (exclusief urine) en keukenafval. Neerslag van fosfaat werd geconstateerd in alle reactoren.

Hoewel niet grondig onderzocht, is er een indicatie dat reductie van pathogenen (hier uitgedrukt door *E.coli*) afhankelijk is van de verblijftijd en operationele parameters. In een accumulatiesysteem met een extra navergistingsperiode (extra 140 dagen) kan een goede verwijdering van *E.coli* worden verkregen. De WHO standaard werd hierbij gebruikt als referentie.

Wanneer zwartwater wordt behandeld in een accumulatiesysteem dienen grotere influentvolumes te worden behandeld, resulterend in relatief grote reactorvolumes. Voor 6 maanden opslag is ongeveer 1.4-1.6 m^3 reactorvolume per persoon nodig voor de behandeling van zwartwater en keukenafval. Wanneer urine apart wordt ingezameld kan het volume worden teruggebracht tot slechts 0.3-0.5 m^3/persoon. Op basis daarvan wordt een accumulatiesysteem aanbevolen voor kleine volumes van zeer geconcentreerde media, zoals bruinwater en keukenafval die beide worden ingezameld met een minimale hoeveelheid water.

Een UASB septic tank biedt voordelen boven een accumulatiesysteem vanwege de efficiënte scheiding van slib en vloeistof. Continu wordt effluent geproduceerd met nutriënten die voornamelijk in opgeloste vorm (ammonium en ortho-fosfaat) aanwezig zijn. Afhankelijk van de reactortemperatuur en het slib-watercontact kan een hoge verwijdering van opgelost organisch materiaal worden verkregen. Zware metalen in het effluent voldoen aan de irrigatie-eisen. Het gehalte aan pathogenen na behandeling is door de relatief korte verblijftijd nog aan de hoge kant in het effluent. Nabehandeling van het effluent is noodzakelijk voor een betere effluentkwaliteit als nutriëntenterugwinning als doelstelling wordt nagestreefd. In Tabel 3 is de effluentkwaliteit weergegeven zoals verkregen met een UASB septic tank voor de behandeling van geconcentreerd zwartwater bij een temperatuur van 25°C.
In het onderzoek is één set aan operationele parameters toegepast en dit induceert de mogelijkheid van procesoptimalisaties.

Gesuspendeerd materiaal cumuleert in het langzaam groeiende slibbed. De frequentie van slibverwijdering hangt af van de procesomstandigheden, het verwijderingsrendement en de vereiste graad van slibstabilisatie. In ieder geval dient maximaal eens per jaar een deel van het slibbed te worden verwijderd en dit kan eventueel worden hergebruikt in de landbouw.

Berekeningen in dit rapport geven aan dat ook onder suboptimale omstandigheden ruim voldoende biogas wordt geproduceerd voor de energiebehoefte van de reactor (verwarming) en het vacuüm systeem. Het surplus aan energie kan voor andere doeleinden worden gebruikt. Gebruik van biogas via energieopwekking is vooral afhankelijk van de schaalgrootte waarin vergisting wordt toegepast.

In deze studie is veel informatie verzameld over de samenstelling en anaërobe behandeling van geconcentreerd zwartwater en keukenafval. Aanbeveling is om de verdere mogelijkheden van anaërobe vergisting van geconcentreerde huishoudelijke afvalwaterstromen nader te verkennen. Onderzoeksaspecten kunnen zijn hogere temperaturen, andere reactorconfiguraties en hogere influentconcentraties.

Een andere aanbeveling is om een holistische aanpak te formuleren om tot een optimaal DESAH systeem te komen. Twee essentiële aspecten verdienen hier meer aandacht. Ten eerste de nabehandeling van anaëroob effluent en ten tweede de mogelijkheid van nutriëntenterugwinning en hergebruik. Ten aanzien van de nabehandeling dient niet alleen de verwijdering van het restant aan organisch materiaal en pathogenen te worden bekeken. Ook het gedrag van microverontreinigingen (geneesmiddelen, hormonen en cosmetica) moet aanvullend worden onderzocht.

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. Dat zijn alle waterschappen, hoogheemraadschappen en zuiveringsschappen en de provincies.

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 inventarisaties van de behoefte 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 zes miljoen euro.

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SUMMARY

The innovative sanitation concept DESAR (Decentralised Sanitation and Reuse), being a revolutionary approach in relation to the current status-quo in sanitation, attracts recently more and more attention. The basic reasons are that the whole concept is logic and fits well in sustainable development. In a simple form it may constitute an answer to solve the problem of a lack of any sanitation in poor, less developed parts of the world. For richer countries having its complex sanitation infrastructure it may be an answer when necessary to replace the existing, old infrastructure or when building new housing estates.

There is no universal DESAR concept, fitting each situation. Several variations may exist within different scenarios. There are however a few elements that are the same for all these scenario's. One of these elements is that wastewater streams are separated according to their degree and type of pollution and reuse potential of resources. Three main resources are considered: bio-energy (originated from transformation of organic material), nutrients (nitrogen, phosphorus as main nutrients but also potassium and sulphur) and water (produced after advanced treatment of cleaner wastewater streams). Treatment of wastewater streams is selected in such a way that reuse potential is preserved.

Separation of different wastewater streams on household level constitutes thus the starting point of the DESAR concept. Two groups of streams are usually distinguished: black water consisting of a mixture of faeces and urine diluted with clean water used for their transport and grey water, the remaining wastewater produced in the household. Faeces and urine represent a very concentrated mixture in terms of organics and nutrients (both valuable) but also includes the main fraction of pathogens, pharmaceutical residues and their metabolites and endocrine disrupting components (all harmful).
It is recommended to keep black water concentrated to facilitate sustainable treatment options and take advantage of its reuse potential. Also concentrating all potential risks in a very small volume enables a better control and limits the negative effect on receiving water bodies.

In the concept presented here, also solid kitchen refuse, produced during meals' preparation and food leftovers, is suggested to be collected and transported with the black water stream for maximal recovery and reuse of resources. Solid kitchen refuse carries an organic load (kgCOD.person^{-1}day^{-1}) comparable with black water and is easily biodegradable. In this study, kitchen refuse from a hospital was used. Since majority of its components were processed (cooked) its characteristics may vary from normal kitchen refuse with a higher raw fraction. At this moment solid kitchen refuse is in the Netherlands separately collected and together with garden waste transported for (mainly) co-composting.

Since black water is considered as a problematic waste stream, main attention is put on its handling. Depending on the collection and transport system (toilet) and the resulting moisture of the influent two different technologies can be applied, anaerobic digestion and composting.
Anaerobic digestion of concentrated wastewater can be considered as a sustainable option as recovery of energy is applied while nutrients are preserved for reuse. Therefore this technology was here extensively investigated. Two types of concentrated streams were subjected to anaerobic digestion: black water with and without kitchen refuse. Within black water a

further distinction was made into ‘classic’ black water consisting of daily produced amounts of faeces and urine by one individual (1 times faeces and 5 times urine) and brown water mainly consisting of faeces. The brown water was obviously more concentrated than black water (Table 1).

The feasibility of anaerobic digestion for (pre)treatment of concentrated black water and kitchen refuse and its effect on the overall DESAR scheme was the main research question to be answered.
Feasibility was defined by:

- the efficiency that can be achieved in different types of anaerobic digesters operating under various process conditions;
- the quality and reuse potential of the attained product(s);

Efficiency was considered to be satisfactory when approaching maximum anaerobic biodegradability assessed in batch tests under optimal environmental conditions. Stability was an important measure when characterising sludge for reuse. Anaerobic effluents were characterised in a detailed way in relation to different process configurations and operational parameters. Form and concentration of the essential plant nutrients played here an important role.

To obtain concentrated black water, a vacuum system was implemented consisting of vacuum toilets and transport. The volume of flush water used was 1l per time instead of the conventional 6-9 l/time. Implementation of vacuum technology enables to collect black water in a very concentrated form when comparing with the current situation. It allows also saving clean water used just for transport purposes. Although this is not a primary issue in the Dutch situation, saving pure drinking water used nowadays just for transport purposes should be a logic approach. Application of vacuum technology saves more than 20% of the total water use in the household.

Also kitchen refuse was transported to the reactors via the toilets, simulating a low water use system to incorporate this waste into the wastewater stream (garbage grinder). Through almost three years of operation, the vacuum toilet and transport system did not face any problems and 1 l water was in most of the cases sufficient to clean the toilet bowl and even to flush significant amounts of kitchen refuse (up to 1 kg wet weight with 1 l water). Although not investigated, a proper choice of cleaning chemicals is essential when operating this kind of systems making sure that hazardous components do not reach the biological treatment. Environmental friendly, biodegradable detergents were used to clean the toilet bowl. There were no nuisances faced during the vacuum transport of waste but the distances between toilets and reactor were very short.

The effect of combining different concentrated wastewater streams collected with a vacuum system on the wastewater composition is reflected in Table 1.

TABLE 1 **CHARACTERISATION OF BLACK WATER, BLACK WATER + KITCHEN REFUSE AND BROWN WATER AND KITCHEN REFUSE**

Parameter	Unit	Black water	Black water + kitchen refuse	Brown water + kitchen refuse
Total COD	$g.l^{-1}$	9,5-12,3*	13,3-22,9	31,8-66,0
Particulate COD	$g.l^{-1}$	7,0-9,6	10,3-17,1	23,9-62,0
Total N	$gN.l^{-1}$	1,0-1,4	1,2-1,7	1,7-1,9
Total P	$gP.l^{-1}$	0.091-0,144	0.11-0,21	0,23-0,35
Biodegradability	%	60	60-80	70-85

* ranges from different research runs are given; the differences were caused by not always ideal ratio between black water components and kitchen refuse (theoretical ratio faeces:urine:kitchen refuse=1:5:1), difficult sampling, different weights of kitchen waste per 1l of flush water and perhaps little differences in flush volume by different toilets (e.g. 0.9 vs. 1,1 l)

Two types of anaerobic digesters were extensively tested: the accumulation digester and the UASB septic tank. An accumulation system was chosen because it is a technically simple system with no mechanical parts within the reactor. The UASB septic tank was chosen since it is an attractive, low-cost improvement of a septic tank being implemented world-wide, especially in distinct areas not connected to the sewerage.

An accumulation system is a continuously fed reactor, by which its effective (digestion) volume is increasing in time. It combines digestion and storage in one volume. After the maximum volume or the demanded storage time is reached, the reactor is emptied in once (or left without feeding for further stabilisation - additional storage). Since the objective is to reuse the digested, nutrient-rich mixture in agriculture, the storage period is determined by the required frequency of fertilisation. In Dutch conditions it is forbidden to spread fertilisers in the winter period; 6 months of storage are required.

Except of the simple tank, facilities for collection and further management of formed biogas are needed. More efficient treatment requires higher temperatures, thus heating facilities and isolation of the tank. When emptying the reactor after the required storage a minimum volume of digested medium, serving as well adapted inoculum for the next run should be left.

The UASB-septic tank is a hybrid reactor: continuous regarding the liquid, but accumulating the solids. It combines features of the UASB reactor and the conventional septic tank. The up-flow mode allows for an improved removal of suspended solids (better entrapment) and better biological conversion of dissolved components, like in a UASB reactor. Accumulation of solids characterises operation of a septic tank. The long sludge retention time (SRT) enables high stabilisation of solids being property of both types of systems. Three products are obtained: a nutrient rich mixture, a stabilised sludge and biogas.

The characteristics of the two pilot-scale reactors used in this research are given in Table 2.

TABLE 2 RANGES OF OPERATIONAL PARAMETERS APPLIED WHEN TESTING PILOT-SCALE AC AND UASB$_{ST}$ DURING DIFFERENT RESEARCH PHASES

Parameter (average)	unit	AC Black water + kitchen refuse	AC Brown water +kitchen refuse	UASB$_{ST}$ Black water
Loading	Nº individuals	1	1-2	1
Flow, Q	$l.d^{-1}$	5.3-7.5*	1.8-3.4	7
Flush water used	$l.flush^{-1}$	~ 1	~ 1	~ 1
Temperature	ºC	20	15,20	15-25
Reactor volume	l	1000		200
HRT/accumulation time	d	115-150	115-280	27-29
OLR	$kgCOD.m^{-3}.d^{-1}$	0.3	0.1-0.3	0.33-0.42

Accumulations systems were tested for two types of wastewater: (1) black water with kitchen refuse and (2) brown water with kitchen refuse (Table 1,2). The results obtained are listed in Table 3.

Anaerobic digestion of concentrated wastewater streams in an accumulation system is certainly an excellent (pre-)treatment step. About 58% conversion (transformation of influent organics to biogas) is achieved for digestion of brown water and kitchen refuse operating under 20ºC for a period of 150 days. Long retention times and proper choice of inoculum (type and quantity) allow for a high stabilisation degree of the (digested) medium. Additional storage without feeding can still be applied (if there is a possibility to use two tanks) leading even to the complete stabilisation of the influent and significant inactivation of pathogens.

TABLE 3 EXAMPLES OF EFFLUENT QUALITY ATTAINED WITH AC-SYSTEM TREATING BLACK WATER AND KITCHEN REFUSE AND BROWN WATER WITH KITCHEN REFUSE AND UASB SEPTIC TANK TREATING BLACK WATER

Parameter	Unit	AC black water + kitchen refuse, 20ºC	AC brown water + kitchen refuse, 20ºC	UASB septic tank, black water, 25ºC
Total COD	$mg.l^{-1}$	4065*	21900*	2733
Soluble COD	$mg.l^{-1}$	820	1200	1376
VFA	$mg.l^{-1}$	40	<100	500
Ammonium	$mgN.l^{-1}$	1450	1800	1068
Phosphate	$mgP.l^{-1}$	93	104	59

* average concentration after the reactor content is mixed; it includes thus both: liquid and solids.

Digested medium consists of liquid and sludge. Stratification occurs in the reactor during the feeding but during its emptying, mixing can not be avoided. That is why high COD concentrations are reported in Table 3. An AC system is not designed for separation between solids and liquid. High concentrations of ammonium are present in the reactor content, even higher for a mixture of brown water and kitchen refuse. Precipitation of phosphate was observed in all digesters but co-digestion of kitchen refuse and brown water enhanced this process.

Although not examined thoroughly there is indication that reduction of pathogens (in this case expressed as *E.coli*) is a function of retention time and operational conditions. In an accumulation system with an additional post-digestion period (in this research additional 140 days) satisfactory removal of *E.coli* can be reached. Due to the lack of any other reference WHO guidelines (irrigation water) were taken for comparison (below 1000 $cfu.100ml^{-1}$).

When treating black water in an accumulation system still significant influent volumes are provided resulting finally in relatively large reactor volumes. To give an idea of required volume for 6 months storage approximately 1.4-1.6 m^3 reactor volume per person is needed to treat black water with kitchen refuse while when urine is separately collected only 0.3-0.5 $m^3.person^{-1}$ is required. From this point of view *an accumulation system is only recommended for concentrated media, like brown water* and kitchen refuse (collected with a limited amount of flushing water).

A UASB septic tank offers an advantage above an accumulation system: (efficient) separation of the solid and liquid phases. Continuously produced effluent contains nutrients, mainly in the soluble form of ammonium and phosphate. Depending on operational temperature and sludge-wastewater contact high removal of dissolved organic matter can be achieved although not complete. Heavy metals in the effluent do not constitute any problem for reuse in light of existing standards (e.g. irrigation). Concentration of pathogens will be still high. Further concentration or treatment of the effluent will be necessary. In some cases direct reuse for close locations can be considered (e.g. urban agriculture).

Solids are concentrated in the sludge bed that slowly develops. The frequency to remove part of the sludge will depend on process conditions, efficiency and required rate of sludge stabilisation. Maximal once per year part of the sludge bed will be discharged and can be reused in agriculture.

Table 3 presents the effluent quality, achieved in this study when applying an UASB septic tank for treatment of concentrated black water under higher temperature conditions of 25°C. Since only a restricted set of operational parameters was used in this study, further process optimisation is possible.

If the total organic load from one person is 122 $gCOD.d^{-1}$ (faeces, urine, kitchen refuse); at methanisation rate of 58% as attained in this research one may expect that 27 l methane will be daily produced per person, which is almost 10 m^3 yearly; expressed in the energy term 98 $kWh.y^{-1}.p^{-1}$. Since batch tests indicate higher anaerobic biodegradability of a mixture of black water and kitchen refuse (even up to 70-80%) optimising digestion process may lead to a higher biogas yield (32 and 37 $l.p^{-1}.d^{-1}$ respectively) and further mineralisation of treated medium.

Calculations presented in this report indicate that sufficient biogas is produced even under sub-optimal conditions to cover the energy requirements of the reactor (heating) and the vacuum installation, and leaving surplus energy. Reuse of biogas and its transformation into energy will be determined by scale of application.

A lot of information was gained in this study on the characterisation and anaerobic digestion of concentrated black water and kitchen refuse. The recommendations are to invest still more time in recognizing all potentials of anaerobic digestion of concentrated wastewater streams (higher temperatures, any reactor configurations, higher influent concentrations, etc.) and to build a holistic approach leading to an optimal DESAR scenario for given conditions. Two essential aspects deserve still more attention: (1) polishing of digested effluent and (2) nutrient recovery and reuse. Within polishing not only removal of remaining organics and pathogens but also fate of micropollutants (pharmaceuticals, endocrine disrupting compounds, personal care products) will probably attract a lot of attention in the nearest future.

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, 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 six million euro.

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ANAEROBIC TREATMENT OF CONCENTRATED WASTEWATER IN DESAR CONCEPTS

CONTENTS

SUMMARY
STOWA IN BRIEF
SAMENVATTING
STOWA IN HET KORT

1 INTRODUCTION 1
1.1 DESAR 1
1.2 Characterisation of domestic wastewater streams 3
  1.2.1 Waste(water) quantities 3
  1.2.2 Wastewater composition per stream 4
1.3 Low flush collection and transport of black water 7
  1.3.1 Low flush toilets 7
  1.3.2 Vacuum sewer 9
1.4 Anaerobic digestion of concentrated wastewaters 9
1.5 Technical configurations: AC, UASB septic tank, CSTR 11
  1.5.1 Accumulation system (AC) 11
  1.5.2 UASB septic tank (UASB$_{ST}$) 11
  1.5.3 CSTR systems 12
1.6 Research objectives 13

2 MATERIAL AND METHODS 14
2.1 Vacuum collection and transport system 14
2.2 Pilot-plant and research phases 15
2.2.1 General 15
2.2.2 Accumulation reactors 17
2.2.3 UASB septic tanks 17
2.3 Start-up of the pilot-plant 18
2.3.1 Phase 1 (AC1 black water and kitchen refuse and AC2 brown water and kitchen refuse, 20ºC) 18
2.3.2 Phase 2 (AC2 brown water and kitchen refuse, $UASB_{ST}$ 1 and 2 black water 15ºC and 25ºC) 18
2.4 Reactors' feeding 19
2.5 Operational conditions and parameters 21
2.6 Batch tests 21
2.6.1 Maximum anaerobic biodegradability 21
2.6.2 Sludge stability 21
2.6.3 Maximum specific methanogenic activity 21
2.6.4 Hydrolysis constant 21
2.7 Sampling and analysis 22

3 RESULTS INFLUENT CHARACTERISATION 23
3.1 Influent characterisation, separated components (faeces, urine, kitchen refuse) 23
3.2 Influent characterisation, mixtures 24
3.2.1 Black water 24
3.2.2 Black water with kitchen refuse 25
3.2.3 Brown water with kitchen refuse 25
3.3 Bacteriological characterization of faeces, urine and kitchen refuse 37
3.4 Heavy metals in concentrated mixtures 28
3.5 Hydrolysis constant (k_H) 29
3.6 Anaerobic biodegradability of influent 29
3.7 Final remarks 30

4 ACCUMULATION SYSTEM (AC UNDER 15 AND 20ºC) 32
4.1 Results 32
4.1.1 Operational problems 32
4.1.2 Reactor content composition (20ºC, 150 days filling time) 32
4.1.3 Stratification 36
4.1.4 Conversion to biogas 41
4.1.5 Mass balance 42
4.1.6 Effluent stability (digestibility) 43
4.1.7 Methanogenic activity 44
4.1.8 Heavy metals 44
4.1.9 Fate of pathogens 46
4.2 Reactor design 48
4.3 Discussion 51
4.3.1 Organic fractions removal 51
4.3.2 Biogas and potential energy production 51
4.3.3 Nutrients 52
4.3.4 Constraints in agricultural reuse 53

5 UASB SEPTIC TANK UNDER 15 AND 25^{0}C (UASB$_{ST}$ 1 AND 2) 54
5.1 Results 54
5.1.1 Process Conditions and Operational Problems 54
5.1.2 Effluent characteristics at operation of UASB septic tanks with black water 55
5.1.3 Conversion of organic compounds 59
5.1.4 Sludge Bed Development 60
5.1.5 Sludge Stability 61
5.1.6 Heavy metals 62
5.1.7 Fate of pathogens 64
5.1.8 Mass balance 64
5.2 Design 65
5.3 Discussion 67
5.3.1 Organics removal 67
5.3.2 Biogas production 67
5.3.3 Nutrients and organics in the effluent and sludge 68
5.3.4 Constraints in agricultural reuse 68

6 GENERAL DISCUSSION 70

7 CONCLUSIONS 74

LITERATURE 76

Appendix 1 List of commonly found pathogens in excreted urine 86
Appendix 2 List of commonly found pathogens in excreted faeces 87
Appendix 3 Effect of chosen metal exposure on/to humans 88
Appendix 4 Heavy metal concentrations of different wastewater components, guidelines found in literature and examples of standards for different disposal/management practices 90
Appendix 5 Batch test: Maximum substrate biodegradability 91
Appendix 6 Batch test: sludge stability 92
Appendix 7 Batch test: maximum specific methanogenic activity 93
Appendix 8 Batch test: Determination of hydrolysis constant 94
Appendix 9 Analytical methods 96

1

INTRODUCTION

1.1 DESAR

Current advanced central collection, transport and treatment of domestic waste(water) mainly implemented in richer countries, do not always comply with sustainability issues, such as prevention, reduction and reuse. Sanitation systems as developed in richer countries are usually complex and thus expensive to be implemented worldwide. In contrast, in poorer countries there is often no sanitation at all resulting in serious hygenic and environmental impacts.

The sanitation in the Netherlands is characterised by extensive sewerage collecting all types of (domestic) wastewater streams (concentrated and diluted, often including storm water), transporting it over long distances to 393 (in year 2000) central wastewater treatment plants (WWTP). The sewerage with its total length of 86 452 km serves 85% of the municipalities (Rioned, 2002). Translating it into number of inhabitants connected to the sewerage – only 1.6% is not connected to the sewer (2001). Wastewater is treated through several purification steps where a biological aerobic activated sludge process is a main technology. A side product produced during this multi-step process – sludge has to be further managed, what usually represents the most problematic economic and environmental issue.

The centralised sanitation systems in many ways can be blamed for not being logic. To start with, pure, high quality drinking water is wasted to transport the highest load of pollution originally produced in very small quantities such as faeces and urine. As result of mixing of different streams produced in households, pollution components originally mainly present in one stream (e.g. pathogens, pharmaceuticals present in black water) diffuse to other streams (grey water, storm water). On the other hand concentrated streams rich in potential resources (nutrients, organics) get diluted making potential recovery of resources economically not feasible and technically difficult.

On a WWTP energy is used (24 kWh per p.e. per year, CBS, 2000) for several purposes where aeration of wastewater mixture to oxidise organics and nitrogen constitutes an important cost post. Produced sludge (24 kg TS per p.e. per year, CBS, 2000) after its stabilisation and significant reduction of volume, although having high reuse potential, can not be reused in agriculture due to its contamination with heavy metals (from rain water, grey water, industrial discharges, infiltrations). Because landfilling is not an option, incineration is generally applied. About 85% of municipal sewage sludge is incinerated or thermally dried, 5-10% is composted while 5% undergoes wet oxidation (Infomil, 2002). Wastewater sludge management costs amount to 40% of total costs of wastewater treatment.

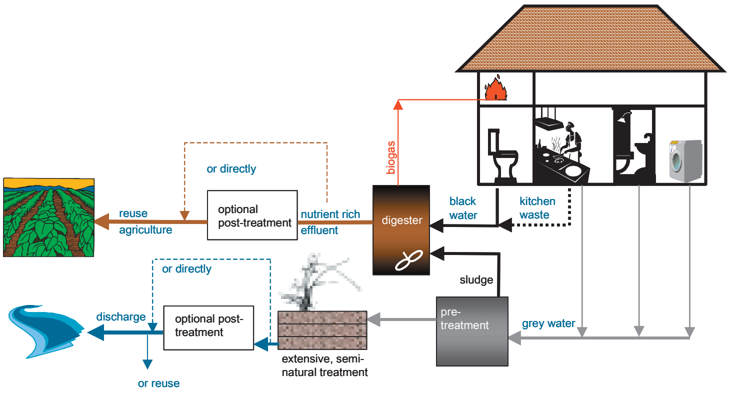
The DESAR – decentralised sanitation and reuse - concept is meant to be an alternative for current wastewater collection, transport and treatment oriented to reduce drinking water and energy consumption, to produce energy (biogas) and conserve, recover and finally reuse potential resources. Any types of applications can be considered: small communities, large buildings or residential areas.

The collection, transport and treatment of concentrated wastewater streams (black water) can be realised within a considered area or object, while relatively diluted streams (grey water, rain water) can be transported to a semi-centralised location in the neighbourhood for their purification. The recovery of resources from both streams (nutrients, water) can be also located semi-centrally depending on the application scale.

This concept forms an alternative not only to current wastewater management but also to management of kitchen refuse (organic waste) being an inconvenient practice in large, multi storey buildings. In DESAR concepts the solid fraction of kitchen refuse can be co-treated with concentrated toilet wastewater as both have similar characteristics. This can solve problems of smell, and lower the transport and treatment costs, though garden waste still has to be transported. The collection, transport and treatment of storm water are managed separately but this aspect is beyond the scope of this report.

Since black water and kitchen refuse carry the highest organic load and their volume can be far minimised by using small amounts of water for their collection and transport, anaerobic digestion of both substrates is considered as a core technology. Conversion of organics leads to production of biogas that depending on process scale can be reused for reactor operation and/or in the household. The effluent of the anaerobic digester will still contain residual organics and high nutrients concentrations. The stability of the effluent will depend on process configuration and operational conditions. When considering effluent reuse, fate of pathogens, heavy metals and possibly for future also micropollutants (pharmaceutical metabolites, endocrine disrupting compounds, personal care products) have to be considered. Post-treatment may involve an additional step to remove remaining organics, recovery and concentration of nutrients and hygenisation. The grey water considered as a relatively clean stream can be treated using ‘simpler’ process, like sand filtration. Pre-treatment to eliminate the particulate fraction will be usually required. When a high quality product is required an additional step might be needed (Figure 1.1).

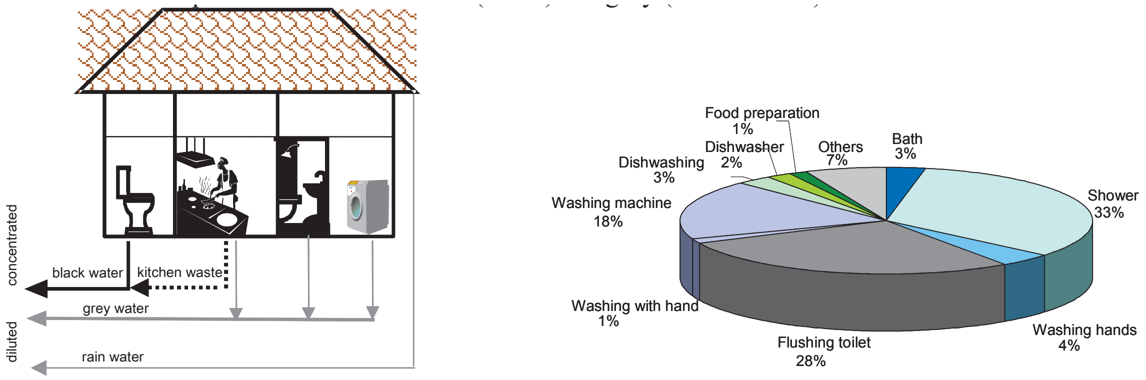
FIGURE 1.1 **PRINCIPLE OF A DESAR CONCEPT**



1.2 CHARACTERISATION OF DOMESTIC WASTEWATER STREAMS

A number of wastewater streams of different characteristics and composition is produced in a household as a result of various human activities. They can be generally divided into diluted and concentrated streams. The most concentrated streams originate from the toilet (faeces and urine) and from the kitchen (solid kitchen refuse generated during meals preparation and food left-overs). Relatively diluted streams are produced in the bathroom (shower, bath, washing up), during textiles washing, and partially from the kitchen (food preparation and washing the dishes) (Figure 1.2). The wastewater stream from the kitchen tends to be voluminously small but quite concentrated in terms of organics (part of cooking waste wasted to the sink), nutrients and detergents (dish-washers). One may consider its incorporation to the first waste(water) category (concentrated).

FIGURE 1.2 GENERAL AND SPECIFIC TYPES OF WASTEWATER PRODUCED IN HOUSEHOLD AND WATER USAGE PER HOUSEHOLD ACTIVITY IN THE NETHERLANDS (NIPO/VEWIN, 2002)



1.2.1 WASTE(WATER) QUANTITIES

The grey water is the largest wastewater fraction and in the Netherlands it constitutes 72% of the total domestic wastewater volume (91.4 l.person^{-1}.day^{-1}) (NIPO/VEWIN, 2002). Its volume can be hardly reduced since people have to use its certain amount for personal care, washing clothes and dishes and cooking. The water consumption depends on the characteristics of household equipment available on the market. The wastewater produced from flushing the toilet in the Netherlands was in year 2001 (Table 1.1) 34.8 l.person^{-1}.day^{-1} (being 27% of total drinking water usage). The fraction, which is flushed, urine and faeces account for a very small volume when comparing with the water used. The collected urine volume is 365-550 l.p^{-1}.y^{-1} and the faeces with toilet paper around 40–55 kg wet weight.p^{-1}.y^{-1} (Vinnerås, 2002, van der Wijst and Groot-Marcus, 1998). Contrary to the grey water, the volume of black water can be drastically reduced by implementation of low flush- or water free (compost) toilets.

TABLE 1.1 DRINKING WATER CONSUMPTION IN THE NETHERLANDS IN THE PERIOD 1980-2001, IN *L*.PERSON^{-1}.DAY^{-1} FOR DIFFERENT HOUSEHOLD ACTIVITIES

Year **Activity**	**1980**	**1985**	**1990**	**1995**	**1998**	**2001**
Bath	12,6	13,1	8,6	9,0	6,7	3,7
Shower	19,2	21,2	28,2	38,3	39,7	42,0
Washing hands	7,4	6,9	4,5	4,2	5,1	5,2
Washing clothes (by hand)	2,4	2,4	2,4	2,3	2,1	1,8
Washing machine	22,0	21,6	22,7	25,5	23,1	22,8
Dish washing (hand)	9,3	10,5	9,4	4,9	3,8	3,6
Dishwasher	1,1	1,1	0,8	0,9	1,9	2,4
Food preparation	4,0	4,0	4,0	2,0	1,7	1,6
Flushing toilet	29,0	32,7	34,0	39,0	36,2	34,8
Others	2,0	3,0	8,0	8,2	7,7	8,2
Total	109,0	116,5	122,6	134,3	128,0	126,2
m^3.year^{-1}	39,8	42,5	44,7	49,0	46,7	46,0
Grey water m^3.year^{-1}	29,2	30,6	32,3	34,8	33,5	33,3

The weight of total biowaste produced in Dutch household, as measured in 2001, was 240g. p^{-1}.d^{-1} (IPA, 2001). This amount includes also garden waste, which is not subjected to digestion in the proposed sanitation concept. In one Dutch province - Drenthe the total amount of collected biowaste including yard waste was much higher in comparison to the average from the whole country and amounted to almost 390 and 370 g.p^{-1}.d^{-1} in 2000 and 2001 respectively (IPA, 2003). Study conducted on kitchen refuse (swill) characterisation in Dutch hospitals (Duynhoven, 1994) indicated a much higher production of 450 g.patient^{-1}.day^{-1} but in these types of institutions usually excessive amounts and volumes of meals are produced. De Koning (2003) in his theoretical study on the effect of kitchen refuse on the operation of WWTP assumes 150 g.p^{-1}.d^{-1} of wet kitchen waste. American data indicate *ca.* 200 gram per person per day of solid kitchen refuse (Cooley *et al*. 1996).

1.2.2 WASTEWATER COMPOSITION PER STREAM

Urine contains material, metabolised in the body extracted from the bloodstream by the kidneys (Guyton, 1992). The components of urine are water and nutrients in a water-soluble form. The main fraction of nitrogen, phosphorus and potassium (N, P, K) present in domestic wastewater originates from urine, commonly used numbers are 80% N, 55% P and 60% K. On the other hand volumetrically it is just 1% of the total wastewater stream produced in the household. The volume of produced urine depends on sex and fluid intake. The minimum and maximum limits reported in literature are 0.6 to 2.5 l.p^{-1}.d^{-1} (STOWA, 2001). An average volume of 1.25 l.p^{-1}.d^{-1} is commonly assumed. The nutrient content will vary according to the food intake, for instance protein intake.

Nitrogen is mainly present as urea (80%), ammonia (7%), and creatine (6%), and further small concentrations of free amino acids or shorter peptides (Johnston & McMillan, 1952; Lentner et al., 1981; Guyton, 1992; Kirchmann & Pettersson, 1995; NV, 1995; Fittschen & Hermann, 1998; Almeida et al., 1999). The enzyme urease (Alef & Nannipieri, 1995) transforms the urea into ammonia and carbon dioxide (Vinnerås et al., 1999), which rises pH. Complete hydrolysis of urine takes 5 to 8 days in completely closed systems (Helström, 1998). The total load of excreted total nitrogen (N) is 12 gN.p^{-1}.d^{-1} (Table 1.2). More than 95% of phosphorus (P) in urine is inorganic. Its content is directly related to phoshorus intake. Phosphorus is a na-

tural component in food and originates also from various food additives and preservatives. The average amount of phosphorus excreted is 1 $g.p^{-1}.d^{-1}$. Potassium (K) is present in relatively high concentrations as free ions and ranges from 2 to 4 $g.p^{-1}.d^{-1}$. Calcium (Ca) and magnesium (Mg) are both present in small quantities. A value of 0.2 $g.p^{-1}.d^{-1}$ is reported average for both minerals. Sodium (Na) and chlorine (Cl) are present in high concentrations in urine. Average values of 4 $g.p^{-1}.d^{-1}$ and 7 $g.p^{-1}.d^{-1}$ respectively can be assumed (STOWA, 2001). Concentrations of heavy metals in urine are very low. The pH of fresh urine is slightly acidic, pH=6 (Berne and Levy, 1998) but increases rapidly to more than 8.5 as a result of degradation/hydrolysis of urea into ammonia. The COD content in urine is about 12 $g.p^{-1}.d^{-1}$ of which about 85-90% is aerobically degradable (STOWA, 2001).
By a healthy person the urine in the bladder is sterile. However when excreted, different types of dermal bacteria may enter this stream. Normally in fresh urine less than 10 000 bacteria per 1ml can be measured (Tortora et al., 1992). By urinary tract infections caused by *E.coli* (Murray et al., 1990) these concentrations will be higher. No transmissions to other individuals throughout the environment have been reported, when urine was disposed or reused in agriculture. Pathogens causing venereal diseases can be occasionally found in urine but there is no evidence of associated health risk due to survival of these organisms outside the body. The list of commonly found pathogens in excreted urine is included in appendix 1. A concluding remark of this review is that pathogens, which may be transmitted by urine do not constitute a significant public health risk. Inactivation of urinary excreted pathogens during e.g. urine storage (Höglund, 2001) and later in the environment minimises the problem even further.

Faeces contain both material extracted from the bloodstream and undigested material. One third of the dry weight originates from food, one third of intestinal bacteria and one third of the intestine itself. As a consequence the composition of faeces after a diet change will vary less significantly than of the food itself. Fibres (bread, other grain products, vegetables, fruit, potatoes) determine mainly the faecal weight (Wijst and Groot Marcus, 1998). High consumption of non-biodegradable fibres contributes to high faecal weight also because of higher fraction of bound water. In the faecal fraction of wastewater in the Western world, there is also a quantity of toilet paper. The actual produced quantity of faeces depends on age, body mass and amount of food consumed (STOWA, 2001). Based on a dry weight, volume of faeces is in the range of 10-50 $g.p^{-1}.d^{-1}$. Based on a wet weight values in the range of 70-170 $g.p^{-1}.d^{-1}$ are reported (Wijst and Groot Marcus, 1998).
Faeces contain significant amounts of nutrients in a relatively small volume. The nutrient content will vary according to the food intake, for instance protein intake.
Around 50% of faecal nitrogen is water soluble, 20% of it is ammonium, biochemically formed from urea, peptides and amino-acids. A 7% of N is found in (intestinal) bacteria and the rest is organic N incorporated in molecules like uric acid and enzymes. It is assumed based on literature values that one person excretes 1.5 $gN.d^{-1}$ in faeces. The main fraction of phosphorus is undigested calcium phosphate. A total amount of 0.5 $gP.p^{-1}.d^{-1}$ is excreted via faeces. Potassium (up to 1 $g.p^{-1}.d^{-1}$) is present in ionic form (Vinnerås, 2002). Quantities of potassium, calcium, magnesium and sodium are related to the content of these minerals in the undigested food rests (Table 1.2).
Faeces contain pathogens although not always (Höglund, 2001). However, from a risk perspective their presence should always be considered since many different types of enteric infections prevail and the prevalence is unknown for several of them. In appendix 2 prevailing pathogenic groups in this waste stream are given. To ensure a reduction in pathogens, faeces need to be treated or stored under controlled conditions.

Experts attending the First Project Meeting on the Safe Use of Human Wastes in Agriculture and Aquaculture, in Engelberg, Switzerland, in 1985, reviewed epidemiological evidence concerning the agricultural use of human wastes and formulated the Engelberg Guidelines for the microbiological quality of treated wastewater intended for crop irrigation. Those guidelines recommend that treated wastewater should contain:

- < 1 viable intestinal nematode egg per litre (on an arithmetic mean basis) for restricted or unrestricted irrigation; and
- < 1000 faecal coliform bacteria per 100 millilitres (on a geometric mean basis) for unrestricted irrigation.

Unrestricted irrigation refers to irrigation of trees, fodder and industrial crops, fruit trees and pasture, and *restricted* irrigation to irrigation of edible crops, sports fields and public parks.

The guidelines are also applicable to agricultural use if the excreta, in the form of liquid night-soil for example, is applied to the field while crops are growing. The intestinal nematode egg guideline value is designed to protect the health of both field workers and crop consumers and represents a high degree of egg removal from the wastewater (> 99%). The faecal coliform guideline value is less stringent than earlier recommendations, but is in accordance with modern standards for bathing waters and more than adequate to protect the health of consumers. Guidelines for the microbiological quality of treated excreta and wastewater for aquacultural use were developed at the Second Project Meeting held in Adelboden, Switzerland, in June 1987. These recommend zero viable trematode eggs per litre or per kilogram (on an arithmetic mean basis), and less than 10000 faecal coliform bacteria per 100 millilitres or 100 grams (on a geometric mean basis). Such a stringent trematode guideline is necessary as these pathogens multiply very rapidly in their first intermediate aquatic host. The value for faecal coliforms assumes a 90% reduction of these bacteria in the pond, so that fish and aquatic vegetables are not exposed to more than 1000 faecal coliforms per 100 ml (WHO, 1989).

Due to a low uptake in the body heavy metals pass through and can be found in excreta. However, the total amount of (majority of) heavy metals originating from faeces and urine is considerable lower than from grey water and (certainly) storm water (Höglund, 2001). Some health concerns related to the exposure to heavy metals are described in appendix 3. The concentrations of heavy metals in different wastewater components and mixed streams, guidelines found in literature and examples of standards for different disposal/management practices are listed in appendix 4.

Grey water is usually defined as a household wastewater stream without any input from the toilets, consisting of wastewater produced during bathing/showering, washing hands, laundry and from kitchen sink, *each having its own characteristics and variations*. Grey water contains a major fraction of heavy metals but a minor proportion of nutrients. The nutrients in grey water are mainly inorganic. Potassium and phosphorus are used in detergents and their concentrations will mainly reflect the usage rate of these products (Vinerås, 2002). The other sources of nutrients are (solid) kitchen residues (food leftovers) ending up in a kitchen refuse water. The heavy metals in grey water originate from dust (wiped out during house cleaning activities) and chemicals used in household (Cd in P-containing detergents and Zn in anti-fungal shampoo). Although considered as a relatively clean, simple wastewater, polluted with mainly COD, concentration and composition of COD can vary considerably from one to another location (Jefferson, 2001). Eriksson et al (2002) published a review on grey water characteristics. They report that based on the information available in the declaration of contents of the different types of common Danish household products at least 900 different organic chemical substances and compound groups components can be found in grey water. The variation in concentration and composition is due to personal and cultural habits with respect to water use and waste handling and quality and quantity of products used for kitchen, laundry and personal care.

Handling of food rests in the kitchen may have an important impact on composition of kitchen refuse water. Food rests can either end in the sink and will subsequently make part of the grey water wastewater, or as solid can be collected as (green) household waste. Henze (1997) shows that the application of 'clean tech cooking', where large part of the cooking waste is transferred from the sink to the solid waste, can reduce the COD load of grey water from 55g COD.person^{-1}.day^{-1} to 32 gCOD.person^{-1}.day^{-1}. In DESAR concepts where toilet waste (water) is separately collected using a little amount of flush water (e.g. vacuum toilets), the produced mixture of faeces and urine can be treated together with the collected kitchen refuse (due to their similar organic strength) in an anaerobic digester (Zeeman et al, 2000).

TABLE 1.2 **COMPOSITION AND VOLUME OF DOMESTIC WASTEWATER STREAMS AND WASTEWATER COMPONENTS IN G/P/D (AFTER VINNERÅS, 2002, HELLSTRÖM AND KÄRRMAN, 1996; JÖNSSON ET AL., 1997, ERIKKSON ET AL., 2002)**

Parameter	Unit	Urine	Faeces	Greywater	Kitchen refuse
Volume	kg.p^{-1}.d^{-1}	1.25-1.5	0.07-0.17	91.3	0.2
Nitrogen		7-11	1.5-2	1.0-1.4	1.5
Phosphorus	g.p^{-1}.d^{-1}	0.6-1.0	0.3-0.7	0.3-0.5	0.28
Potassium		2.2-3.3	0.8-1.0	0.5-1	0.22
Calcium		0.2	0.53		
Magnesium		0.2	0.18		
BOD		5 -6	14-33.5	26-28	
COD		10-12	45.7-54.5	52	
Total solids, TS	g.p^{-1}.d^{-1}	20-60	30	54.8	75
Cu		4	400	2900	549
Cr		3.7	7.3	365	137
Ni		2.6	27	450	82.3
Zn	mg.p^{-1}.y^{-1}	16.4	3900	3650	700
Pb		0.73	7.3	365	275
Cd		0.25	3.7	15	2.7
Hg		0.30	3.3	1.5	0.25
bacteria	N°.ml^{-1}	10000			
E.coli	Cfu.g^{-1}		10^5-10^8		1.3.10^5-2.5.10^9
Fecal Streptococci (enterococci)			10^5-10^7		5150-5.5.10^8

1.3 LOW FLUSH COLLECTION AND TRANSPORT OF BLACK WATER

1.3.1 LOW FLUSH TOILETS

As shown in the review on wastewater characterisation it is obvious that faeces, urine and kitchen solid waste are responsible for the highest pollution load in total domestic wastewater. Since it is originally present in a very small volume, it is worth to prevent its dilution. There are several reasons to keep it concentrated: on one side high reuse potential on the other hand presence of risks (pathogens, pharmaceuticals, endocrine disrupting compounds) and prevention of (direct) surface water/environment pollution). For water based collection of faeces and urine several systems are available on the market allowing for their minimal

dilution. Another benefit of water saving waste collection is obviously lowering consumption of high quality drinking water.
For the purpose of this study vacuum technology is proposed to collect and transport concentrated waste to the treatment unit. Other possible systems are (extremely) low flush toilets connected to gravity sewers, urine diverting toilets and (water-free) urinals.

Urine diverting toilets have two bowls, and/or two outlets, and/or two different flush water volumes so called small and large flush. Some models have water free urine collection. Total amount of water used in urine diverting toilets varies between 4 and 14 $l.p^{-1}.d^{-1}$ depending on the installed type of toilet (WRS, 2001).
A new generation of urinals on the market do not consume any water at all. The traditional water trap was replaced by oil-based liquids or float systems.

Vacuum toilets were originally developed for use in planes, trains and ships but are becoming more common in households. Vacuum toilets operate with a pump generated vacuum using small volumes of water to flush. Water is used only for rinsing the toilet bowl. The vacuum system operates pneumatically (vacuum pumps, ejectors or compressors) and needs a water connection to the toilet and electricity for the vacuum units. Some types of vacuum toilets are equipped with a (24-volt) battery making it suitable in places where no (constant) electricity is provided. The transport can be vertical for limited distances and horizontal for long distances. The water consumption is in the range of 0 (no flush for urine) to 2.0 $l.flush^{-1}$. A total daily water consumption varies from 0 (usually 4) to 12 (average 7) $l.p^{-1}.day^{-1}$.

A vacuum system requires airtight pipes. The pipes have much smaller diameter than in conventional gravity systems. There are no requirements regarding the slopes (or the length of the pipes). The wastewater can be transported for longer distances and even to a higher level. Vacuum systems depend on electricity. Many systems have been installed already in a larger scale: schools, hotels, whole villages or residential estates (WRS, 2001). In small-scale systems (single household) a vacuum generator is activated only when the toilet is in use, while on a larger scale is running constantly leading to a higher energy consumption. Many manufacturers of vacuum toilets are already on the market (e.g. Evac Oy, Finland, Jets, Standard, A/S, Norway; LVS Low vacuum system, KTH arkitektur, Sweden; Microphor, Microflush, USA; Roediger Vakuum + Haustechnik, Roevac, Germany; Sealand Vacuflush, Sealand Technology Inc, USA; Clever-Vac Wost Man Ecology, Sweden).

Not all types of vacuum toilet available on the market can be installed in individual houses. Some are only appropriate to be installed for large-scale black water systems. Low-vacuum systems involve a moderate vacuum pressure to secure silent operation, and still low water consumption. In some types of vacuum toilets the amount of flush water can be adjusted during use by pressing a flush button/foot pedal for longer or shorter time.
The amount of electricity depends on the size of the system. Toilets installed in single family house use around (2) 10 – 12 $kWh.year^{-1}$ (Jets, Roevac) for an average-sized family.
Like all technical systems vacuum systems require regular service. The service should be done by professionals. Usually most components (pumps, valves, filters, etc.) need to be checked and cleaned annually and replaced about every five years in a private house. In public utilities being exploited more intensively replacements should be done more frequently.

1.3.2 VACUUM SEWER

The construction of vacuum sewers is considered to be easier than of conventional sewers. Construction is much faster and environmental impact is lower. It has been already operated for decades proving to be a reliable system. Vacuum sewer can be a solution in situations where conventional sewers are difficult to be installed. The most serious reasons to consider vacuum technology are: (1) too low or too high terrain slopes (uphill transport is possible), (2) difficult ground conditions (swamps, high water level, rocks, unstable ground), (3) necessity to renovate municipal (or industrial) sewer, (4) varying and low wastewater flow. Also low (and very high?) housing density, presence of water protection areas and septicity of sewage can be listed as reasons to consider vacuum sewage (Roediger, 2000).

Implementation of plastic (PE-HD or PVC) pipes much smaller in diameter than conventional sewer pipes is the biggest advantage of vacuum transport. Pipes can be installed in shallow and narrow trenches minimising costs of installation, maintenance and future modifications. Construction is thus faster having little impact on residents and traffic. Also removal of groundwater is usually not necessary since the pipes can cross over or under streams or trenches. Aspects as water conservation and optimization of reuse potential were already mentioned as important DESAR requirements. Thanks to vacuum, streams characterized by high solid-to-liquid ratios can be (easily) transported. The system tightness totally excludes infiltration and ex-filtration (Airvac, 2004). If system is not tight it will not operate.

A potential disadvantage with vacuum sewer include the possibility of disruption in service due to mechanical breakdowns and power outages. Also, systems may be poorly designed or installed if engineers or contractors have little experience with the technology.

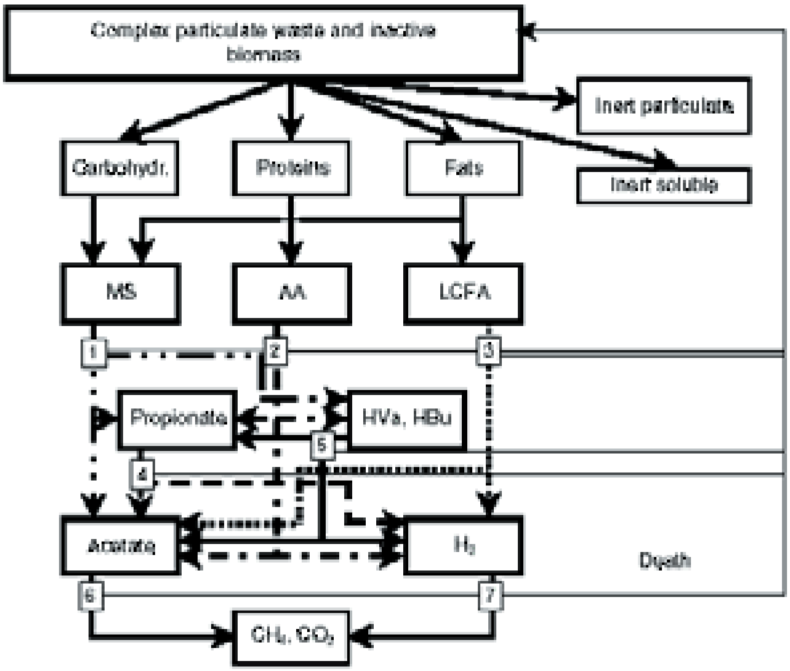
1.4 ANAEROBIC DIGESTION OF CONCENTRATED WASTEWATERS

Anaerobic digestion is a well established technology to convert organic material from a wide range of wastewaters, solid waste and biomass. It is a technologically simple process that requires little energy. One of the end product - biogas consisting principally of methane (55-75% volume) and carbon dioxide (25-25%) is a useful, renewable energy source. Digested medium rich in organics and nutrients can be considered for reuse in agriculture.

This multi-step process consists of four major stages (Figure 1.3):

- *Hydrolysis* where particulate organic matter is converted by extracellular enzymes to monomer or dimeric components (amino acids, single sugars, long chain fatty acids). These compounds can penetrate through bacterial membrane.
- *Acidogenesis* where hydrolysis products are fermented or anaerobically oxidised to short chain fatty acids (SCFA), alcohol and ammonia.
- *Acetogenesis* where SCFA other than acetate are converted to acetic acid or hydrogen and carbon dioxide.
- *Methanogenesis* where acetic acid, CO_2 and H_2 are converted to methane and CO_2.

FIGURE 1.3 THE ANAEROBIC DIGESTION SCHEME OF COMPLEX ORGANIC WASTE MODEL INCLUDING BIOCHEMICAL PROCESSES: (1) ACIDOGENESIS FROM SUGARS (MS), (2) ACIDOGENESIS FROM AMINO ACIDS (AA), (3) ACETOGENESIS FROM LONG CHAIN FATTY ACIDS (LCFA), (4) ACETOGENESIS FROM PROPIONATE, (5) ACETOGENESIS FROM BUTYRATE (HBU) AND VALERATE (HVA), (6) ACETICLASTIC METHANOGENESIS, AND (7) HYDROGENOTROPHIC METHANOGENESIS (BATSTONE ET AL., 2002)



For a stable digestion process it is important that various conversions remain sufficiently coupled, balanced during the process to prevent against the accumulation of intermediate products causing process inhibition. For instance accumulation of volatile fatty acids (VFA) may result in pH decrease up to the level where methanogens do not occur anymore causing even further pH decrease.
Hydrolysis is considered to be the rate limiting step during digestion of complex wastewaters (e.g. Sanders, 2001, Zeeman, 1991, Van Velsen, 1981).

Several environmental factors influence anaerobic digestion. Primary factors are temperature, pH, alkalinity and toxicity (Mes et al., 2003). Three ranges of temperature classify three types of 'controlled' digestion: psychrophilic (10-20°C), mesophilic (20-40°C) and thermophilic (50-60°C). Due to a lower bacterial growth rate under lower temperatures, longer retention of wastewater is required resulting in larger reactor volumes. This contrary to mesophilic digestion where all processes proceed much faster and smaller reactor volumes are required. Thermophilic digestion can be considered when generated wastewater is of high temperature (industrial wastewaters) or pathogens elimination is an objective. The relation between energy requirement and attainable biogas yield determines which type of digestion is to be implemented.

The first steps of anaerobic digestion can occur in a wide range of pH values while methanogenesis proceeds when pH is around neutral (Lettinga and van Haandel, 1993). For pH values outside the range 6.5 – 7.5, the rate of methane production is lower.
Several compounds, such as VFA's, ammonia and cations as Na^+, K^+ and Ca^{++}, heavy metals and xenobiotics at excessive concentrations cause toxicity
High concentrations of ammonia may cause a serious disturbance in the process performance and may inhibit all the sub-processes of anaerobic digestion. There is not much reported

on the effect of ammonia on the hydrolysis, while it seems to occur at high concentrations (Zeeman, 1991, Van Velsen, 1981). Regarding methanogenesis, high ammonia concentration can either affect acetate (directly and indirectly) or hydrogen consuming bacteria. When using adapted biomass, anaerobic digestion can proceed at very high NH_4^+-N concentration, though SRT should be increased.

1.5 TECHNICAL CONFIGURATIONS: AC, UASB SEPTIC TANK, CSTR

One of the DESAR objectives is implementation of a simple, effective and robust treatment system. Considering anaerobic digestion of concentrated domestic waste(water) three relatively simple technical configurations are taken into account, viz. an accumulation system (AC), an up-flow anaerobic sludge bed (UASB) septic tank and a completely stirred tank reactor (CSTR). Below a short description of each system and some examples of practical implementations are given.

1.5.1 ACCUMULATION SYSTEM (AC)

The accumulation system is continuously fed, by which its effective (digestion) volume is increasing in time. After the maximum volume or the demanded reaction/storage time is reached, the reactor is emptied at once (or left without feeding for further stabilisation-additional storage). It is the most simple configuration for anaerobic digestion of waste(water). It combines biological conversion of treated medium and its storage. Except of the tank, facilities for collection and further management of formed biogas are needed. More efficient treatment requires higher temperature, thus heating facilities and isolation of the tank. When emptying the reactor after required storage, attention should be paid to leave a volume of digested medium, serving as inoculum to the next run. Accumulation system has been used for the digestion of liquid animal manure (Wellinger and Kaufmann, 1982, Zeeman, 1991, El-Mashad, 2003).

1.5.2 UASB SEPTIC TANK ($UASB_{ST}$)

The UASB-septic tank system can be a promising alternative for the conventional septic tank (Bogte et al., 1993; Lettinga et al., 1993). The major difference in relation to the conventional UASB system is that the UASB-septic tank also accumulates and stabilises the sludge. In relation to the conventional septic tank system it differs by the up-flow mode implemented, achieving an improved suspended solids removal and better biological conversion. Therefore, the UASB-septic tank is a continuous system regarding the liquid (like UASB reactor), but accumulating the solids (like AC). Accumulation of solids leads to stabilisation of sludge.

Bogte et al., (1993) and Lettinga et al., (1993) investigated the use of a UASB septic tank for on-site treatment of black and total domestic wastewater under low and high temperature conditions. At higher temperatures very high removal efficiencies could be achieved. Under low temperatures (12°C) the conversion of produced VFA to methane was too slow. In such situation implementation of a two-step UASB reactor can be considered (Zeeman and Lettinga, 1999). The first reactor would serve for accumulation of solids and provide limited hydrolysis, acidification and methanogenesis in colder months. In the second reactor mainly methanogenesis would occur. In the warmer months the subsequent hydrolysis/acidification/methanogenesis will proceed faster in the first reactor while the second UASB septic tank may serve as a polishing step for removing and converting of the remaining soluble and suspended COD, washed from the first reactor because of increased biogas production. Expected removal efficiencies in a UASB septic tank (first step) are shown in Table 1.4.

TABLE 1.4 REMOVAL EFFICIENCIES OF COD FRACTIONS IN THE FIRST STEP OF TWO-STEP UASB REACTOR TREATING TWO TYPES OF WASTEWATER UNDER DIFFERENT TEMPERATURE CONDITIONS (BASED ON BOGTE ET AL., 1993, LETTINGA ET AL., 1993, LUOSTARINEN ET AL., 2004)

Removal efficiencies (%)	Temperature (°C)	Black water	Grey+ black water
COD_t	5-20	* 52-69	* 58
COD_t	> 20	** 90-93	** 67-77
COD_{part}	5-20	* 71-86	* 62
TS	> 20	** 93-97	** 74- 81

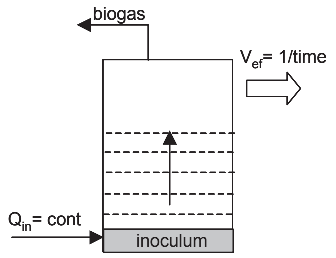
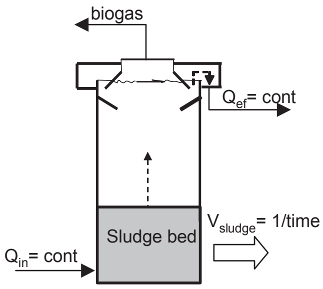
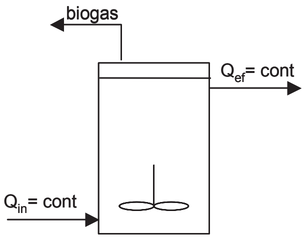
* normal water use, ** low water use

1.5.3 CSTR SYSTEMS

The content of a CSTR tank consists of a mixture of bacteria and treated medium. It is a continuously fed system. Both bacteria and waste(water) have the same retention times, so hydraulic retention time (HRT) equals solid retention time (SRT). The term steady state can be applied to the CSTR reactor under condition that the system is ‘well’ adapted and loading is constant. Steady state conditions will ensure stable biogas production and effluent quality. An additional tank will be usually needed for storage of digested medium. CSTR systems are often applied for digestion of sludge and slurry (Van Velsen, 1981, Zeeman, 1991, Angelidaki and Ahring, 1994).

The CSTR system operates usually well between 15 and 20 days (Zeeman and Lettinga, 1999). In general it can be stated that mesophilic CSTR systems (30-35°C) can be applied when the influent is so much concentrated that it will at least provide enough biogas to produce the energy for heating of the system. The higher the concentration the more surplus energy is produced for other applications. Moreover concentration of the medium to be treated (e.g. vacuum collection of black water) will result in a smaller reactor volume to be installed, providing that the same SRT can be applied for a diluted and a concentrated influent (considering that no inhibiting compounds are present). The digested effluent can be easier applied in agriculture (if possible) as transport cost will be limited when small volumes are produced. Brief characteristics of all described reactors in summarised in Table 1.5.

TABLE 1.5 SCHEMATICALLY SHOWN DIFFERENCES BETWEEN THREE POSSIBLE REACTOR CONFIGURATIONS FOR ANAEROBIC TREATMENT OF CONCENTRATED WASTEWATER IN DESAR CONCEPT (REACTORS ARE SCHEMATICALLY PRESENTED SO THE RATIOS BETWEEN DIFFERENT VOLUMES ARE NOT REALISTIC)

AC	**UASB $_{ST}$**	**CSTR**
		
	Some features	
Long retention time ensures far stabilisation and reduction of pathogens Can be used for high solid streams Treatment and storage combined in one reactor Simple design and operation	Optimised contact with sludge ensures better conversion of soluble compounds than in a conventional septic tank Long retention times for stabilisation of solids Separation of solid and liquid phase	Good contact between waste and biomass, Can be used for high solid streams Homogenous temperature throughout the tank
Larger volumes needed Due to stratification during the feeding period sludge-wastewater contact not optimal During emptying both, treated substrate and biomass are present in the effluent;	Scum formation may appear Continuously produced effluent has to be further managed (disposed off, discharged, polished) Periodically discharged sludge has to be managed as well.	Both, treated substrate and biomass are present in the effluent; additional separation technique may be needed depending on effluent concentration and destiny, Energy needed for mixing Undigested substrate is leaving the reactor Separate storage needed Separation between solid and liquid phase may be required.

1.6 RESEARCH OBJECTIVES

The main objective of this research was to evaluate whether anaerobic digestion is a feasible technology to treat concentrated domestic wastewater streams within a DESAR concept. Concentrated domestic wastewater consisted in this study of toilet wastewater collected in vacuum toilets and solid kitchen refuse. Technically simple configurations, an accumulation system and UASB septic tank, were tested using pilot-tests. The evaluation of applicability of a given configuration was based on effluent characteristics, biogas yield and occurrence of health related risks for reuse, disposal or discharge of treated medium and its residues. Detailed characterisation of concentrated wastewater streams important for reactors' design was obtained. Based on the obtained results of the continuous experiments and batch tests the design procedure for simple digesters was proposed.

2

MATERIAL AND METHODS

2.1 VACUUM COLLECTION AND TRANSPORT SYSTEM

The vacuum collection and transport system used in this research was purchased from a German company Roediger Vakuum + Haustechnik, Germany. It consisted of four essential elements: vacuum toilet, pressure/equalisation tank, vacuum pump and wastewater pump, described below.

FIGURE 2.1 VACUUM COLLECTION AND TRANSPORT SYSTEM FOR BLACK WATER

VACUUM TOILET

The Roevac vacuum toilet is made of porcelain and does not differ usually from a conventional toilet. After use of the toilet, flushing is activated with the push button opening the intermediate valve for approximately 4 seconds. Bowl is flushed under the influence of under-pressure (vacuum), and it refills again. In this research 1 l of water per flush was used. This flush water volume can be adjusted.

EQUALISATION TANK

After flushing, the wastewater was transported to the equalisation tank (long black column in Figure 2.1), where it was stored before sent to the treatment reactor. The equalisation tank functions primarily as the vacuum unit, where a constant under-pressure of 0.5 bar is secured. There was a float in the tank. When the wastewater reaches a height resulting in a volume of 10l the sensor activates the pump/shredder transporting the stored wastewater volume to the reactor.

VACUUM PUMP

The vacuum pump of the manufacturer Roediger (type; KC 0016 D120) had a power of 0.55 kW and the maximum capacity to create an under-pressure of 20 mbar. The pump was installed on top of the equalisation tank and established the under-pressure in the tank. For the set-up as used in this research, the pump had an overcapacity.

PUMP/SHREDDER

The pump/shredder of the manufacturer Ebara (type; DW rox 150) had a power of 1.1 kW. The pump was connected at the bottom of the equalisation tank. Controlled by sensors the pump transported the wastewater from equalisation tank to the reactor. The pump had the property to shred the wastewater in order to increase the contact surface of the substrate with biomass in the biological treatment reactor.

2.2 PILOT-PLANT AND RESEARCH PHASES

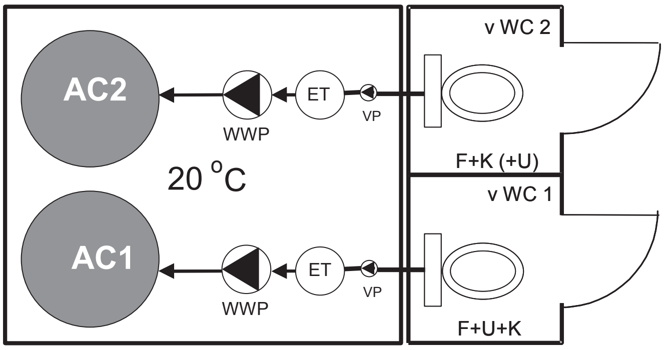
2.2.1 GENERAL

The pilot-plant was built in the Experimental Hall of the Sub-department of Environmental Technology, Wageningen University, the Netherlands. The vacuum collection and transport system and the reactors were located in a specially constructed for this purpose, separated room of the hall (Figure 2.2). Regarding the arrangement of the experimental set-up, two research phases involving different sets of reactors, process conditions and influent compositions can be distinguished.

FIGURE 2.2 IMAGE OF THE SEPARATE ROOM IN THE EXPERIMENTAL HALL OF WAGENINGEN UNIVERSITY, SUB-DEPARTMENT OF ENVIRONMENTAL TECHNOLOGY, WHERE THE VACUUM COLLECTION AND TRANSPORT SYSTEMS AND TREATMENT UNITS FOR CONCENTRATED WASTEWATER STREAMS WERE LOCATED.

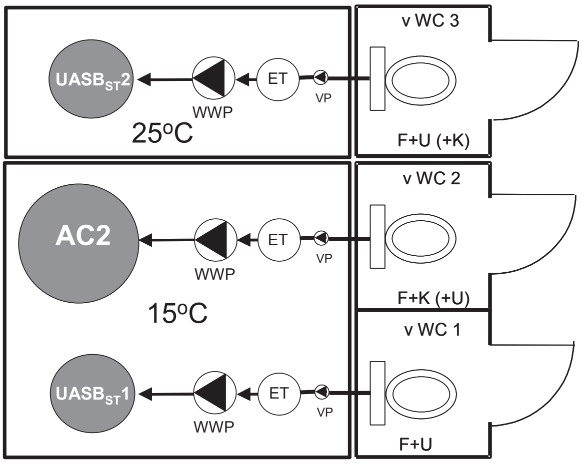
In *phase* 1 two accumulation reactors (AC) were implemented for co-digestion and storage of two types of influent: AC1 for a mixture of faeces, urine and kitchen refuse and AC2 for a mixture of faeces and kitchen refuse, both at 20°C (Figure 2.3).

FIGURE 2.3 THE LAYOUT OF THE PILOT-PLANT USED IN THE FIRST RESEARCH PHASE (PHASE 1)



After two research runs with the AC-systems (main and verification run, 1st and 2nd run respectively) within the first phase, the third research run started within the second phase involving different types of reactors. Two UASB septic tanks were tested fed with mixtures of faeces and urine under two different temperature conditions (15 and 25°C). In addition one accumulation system (AC2) fed with faeces and kitchen refuse was operated at 15°C (Figure 2.4).

FIGURE 2.4 **THE LAYOUT OF THE PILOT-PLANT USED IN THE RESEARCH PHASE 2**



The summary of both research phases and research runs is given in Table 2.1.

TABLE 2.1 **CHARACTERISTIC OF DIFFERENT RESEARCH PHASES AND RUNS**

Phase	**Phase 1**		**Phase 2**
Run	**1st run**	**2nd run**	**3rd run**
Duration, d	143 (AC1) 157 (AC2)	115	> 365 ($UASB_{ST}$ 1,2) 280 (AC2)
Reactors' set	AC1, AC2	AC1, AC2	$UASB_{ST}$ 1, $UASB_{ST}$ 2, AC2
Temperatures, °C	20	20	15 ($UASB_{ST}$ 1, AC2) 25 ($UASB_{ST}$ 2)
Influent	F+U+K (AC1) F+K (AC2)	F+U+K (AC1) F+K (AC2)	F+U ($UASB_{ST}$ 1,2) F+K (AC2)
Loading (=No of individuals per reactor)*	1 (AC1) 2 (AC2)	1 (AC1, AC2)	1 ($UASB_{ST}$ 1, 2, AC2)
Average influent flow, $l.d^{-1}$	7.5 (AC1) 3.4 (AC2)	5.3 (AC1) 2.6 (AC2)	7.0 ($UASB_{ST}$ 1) 6.8 ($UASB_{ST}$ 2) 1.8 (AC2)

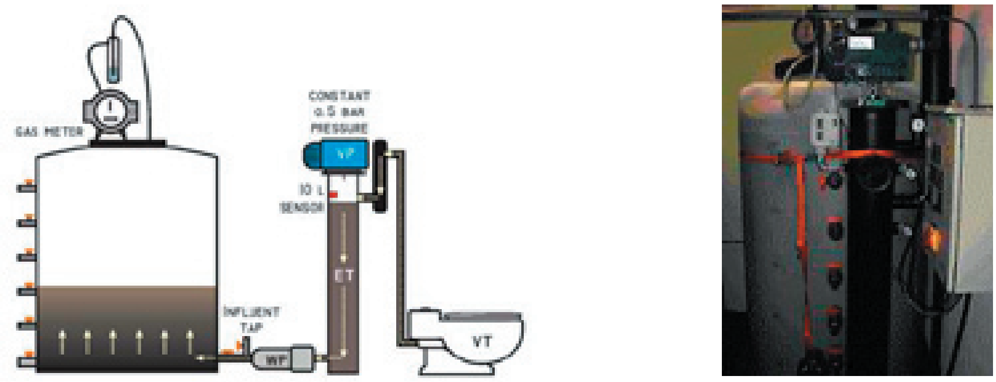
F – faeces, U – urine, K – kitchen refuse, * one (average) individual uses toilet 6 times a day: 1 time for defaecating and 5 times for urinating.

2.2.2 ACCUMULATION REACTORS

In the first phase two identical accumulation reactors (AC1 and AC2) were tested twice, while in the phase 2 only AC2. Each reactor was a closed, semi-transparent polyethylene tank with height, diameter and maximum volume of 1.6 m, 1.0 m and 1.2 m^3 respectively (Figure 2.5). The headspace of the accumulation system was connected to a gas meter situated on top of the reactor. Six sampling taps were mounted on the reactor wall, through its height approximately every 20 cm from each other to monitor the process performance by controlling the quality of the reactor content in time. The accumulation system does not produce any effluent, since all the influent is accumulated inside the reactor until it reaches its maximum capacity.

Accumulation reactors were fed with a mixture of faeces and urine flushed via the vacuum toilet and kitchen refuse, for simplification also transported via the toilet. The vacuum toilet content was first stored in equalisation tank (ET). After the ET was filled up to 10l its content was pumped to the AC. The content of the AC was not mixed mechanically. Short and rapid mixing (partial) was ensured only during emptying of the ET. Biogas formation contributed also (somehow) to reactor's mixing.

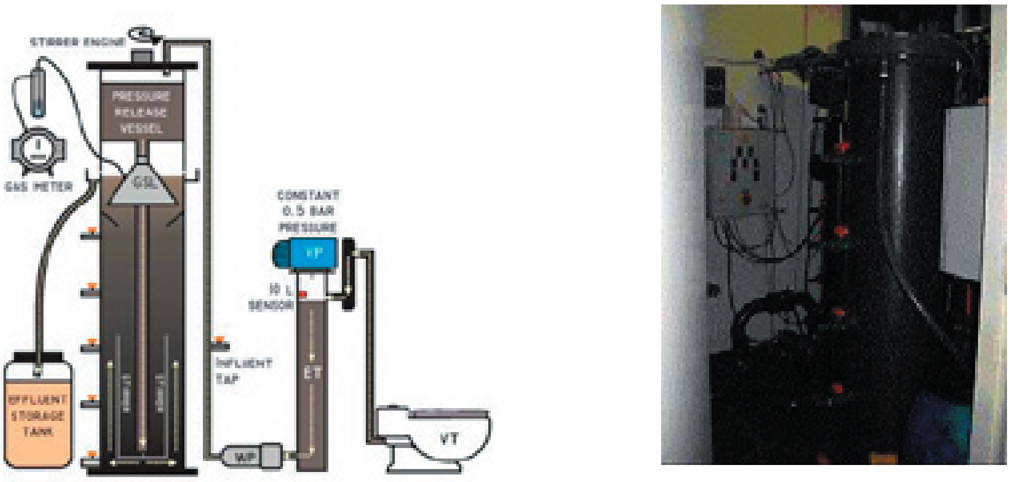
FIGURE 2.5 EXPERIMENTAL SET-UP CONSISTING OF ACCUMULATION REACTOR AND VACUUM COLLECTION AND TRANSPORT SYSTEM



2.2.3 UASB SEPTIC TANKS

The UASB septic tanks tested in phase 2, were built from a PVC cylinder with a height, diameter and volume of 1.65 m, 0.4 m of 0.2 m^3 respectively. Five sampling taps were mounted on the reactor wall through its height every 30 cm (0, 30, 60, 90, 120 cm). The gas, solid, liquid (GSL) separator of the reactors was connected to a gas meter (Schlumberger). The effluent from the reactor was collected in a 60 l storage tank, which was emptied manually every few days. To avoid a hydraulic shock during a rapid emptying of the ET, a so called pressure release vessel was mounted on the top of the UASB$_{ST}$ column (Figure 2.6). After it got filled the wastewater was slowly transported by gravity through the opening in the bottom of the vessel to the bottom of the UASB$_{ST}$ column (5 cm above the reactor bottom) and subsequently passed the sludge bed in an up-flow direction.

FIGURE 2.6 UASB SEPTIC TANK INTEGRATED WITH VACUUM COLLECTION AND TRANSPORT SYSTEM



2.3 START-UP OF THE PILOT-PLANT

2.3.1 PHASE 1 (AC1 BLACK WATER AND KITCHEN REFUSE AND AC2 BROWN WATER AND KITCHEN REFUSE, 20°C)

To ensure a direct start-up of the process and avoid process inhibition by accumulation of volatile fatty acids (VFA) and an undesired drop in pH, inoculum was added. Its amount was calculated based on measured sludge characteristics, waste(water) composition, process temperature and filling time. The mesophilically digested primary sludge from a municipal wastewater treatment plant (WWTP) was used as inoculum. Respectively 100 l and 150 l of inoculum was added to AC1 and AC2 in the beginning of the 1st run (Table 2.2).

After 150 days of operation and 70 days of additional storage (post-digestion) AC1 and AC2 were emptied and restarted (2nd run). Respective volumes of 137 and 167 litres of bottom sludge were used as inoculum (Table 2.2).

2.3.2 PHASE 2 (AC2 BROWN WATER AND KITCHEN REFUSE, $UASB_{ST}$ 1 AND 2 BLACK WATER 15°C AND 25°C)

When the AC2 reactor was emptied after two runs in phase 1, 150 litre of sludge was left as inoculum for the next operation at 15°C. The $UASB_{ST}1$ (15°C) was filled with 87l inoculum sludge from another UASB-septic tank operated at ambient temperatures fed with the effluent of conventional toilets from the Experimental Hall. No inoculum sludge was added to the $UASB_{ST}2$ (25°C). Both $UASB_{ST}$s were subsequently completely filled (above the inoculum where applied) with the effluent of a UASB reactor operated with sewage from village Bennekom, The Netherlands at ambient temperatures. Feeding was subsequently introduced at the set loading rate. The characteristics of all types of inoculum sludge are given in Table 2.2.

TABLE 2.2 **CHARACTERISTICS OF INOCULUM USED IN ALL RESEARCH RUNS**

	phase 1, 1st run AC1, AC2	phase 1, 2nd run AC1, AC2		phase 2, AC2	phase 2, UASB$_{ST}$1
Origin	**WWTP Ede, digested primary sludge (35°C)**	**Adapted bottom sludge from the 1st run**		**Adapted bottom sludge from the 2nd run but not fed for 4 months**	**Sludge from another UASB septic tank treating black water at ambient temperatures**
V, l	100 and 150	137	167	150	87
TS, g.l^{-1}	29,12 (± 2,58)	24.1		18.3	10.4
VS, g.l^{-1}	17.84 (± 1.57)	18.7	23.73	13.1	8.3
Actual methanogenic activity g COD.g VS^{-1}.day^{-1}	0.039	0.015	0.027	0.016	

2.4 REACTORS' FEEDING

PHASE 1, 1ST RUN (AC1 AND AC2)

The AC1 was fed with a mixture of faeces (F), urine (U) (black water) and kitchen refuse (K) in an amount equivalent to the daily production of 1 individual. The AC2 was fed with a mixture of F (brown water) and K, in an amount equivalent to the daily production of these types of waste by 2 individuals. Because no urine separating toilets have been used in this research the contribution of some urine into the AC2 was unavoidable. It needs to be emphasised that no urine was provided *separately* to AC2 and it simulated satisfactorily behaviour of vacuum separation toilets (if available on the market). In no-mix toilets there is also some contribution of urine to the faeces.

The daily (designed) quantity of physiological waste was determined based on the reported frequency that people visit the toilet (Wijst and Groot-Marcus, 1998) and the average volume of F and U being daily produced. For F it was once per person per day (1.p^{-1}.d-1) while for U 5.p^{-1}.day^{-1}. One individual produces approximately 138 g of F and 1.25 l.p^{-1}.d^{-1} or 0.25 l.time^{-1} of U (Table 1.2). The quantity of the generated kitchen refuse by one individual of 0.2 l or kg.p^{-1}.d^{-1} (Table 2.3) was assumed based on statistical numbers for biowaste (vegetable-, fruit- and yard waste) produced in the Netherlands (AOO/IPA, 2000) diminished by an assumed quantity of yard waste which was not considered for digestion within the study.

The reactors were fed with black or brown water from employees of the Experimental Hall, where the experimental set-up was localised. The calculated influent quotum had to be ensured in a week time, but most of the waste was usually provided in the beginning of the week (Monday till Thursday) while hardly anything was produced during the weekend. People contributing to the reactor feeding were asked to mark each time that the toilet was used including kind of substrate provided. Toilets were closed when the week quotum was reached.

TABLE 2.3 **ASSUMPTION OF DAILY PRODUCTION OF FAECES, URINE AND KITCHEN REFUSE PER 1 INDIVIDUAL IN HOUSEHOLDS, WHEN A VACUUM TOILET IS USED FOR WASTE COLLECTION**

Substrate	Daily production	Water used for flushing (c) x frequency of use	Total water consumption by flushing	Total wastewater volume (l)
Faeces	138 g (a)	1.0 litre per flush x 1	1.0 litre	1.138
Urine	1250 ml	1.0 litre per flush x 5 times	5.0 litre	6.25
Kitchen refuse	200 g (b)	1.0 litre per flush (or less)	1.0 litre	1.2

(a) de Mes (2000); (b) e.g. Cooley, A., et al. 1996; (c) Roediger, Vakuum+ Haustechnik, GmbH, vacuum toilet

Kitchen refuse was collected from the hospital “Gelderse Vallei”, Ede, The Netherlands, first once per two weeks, later less frequent. Relatively fresh grab samples were taken from different containers containing a variety of meals. Approximately 75% of the samples was cooked the other 25% consisted of uncooked material. Approximately 75% contained always potatoes, pasta, rice, vegetables and meat. The uncooked part consisted mainly of bread, vegetables, fruit and some preparation waste (like peel etc.). To minimize the moisture content the samples were taken with a sieve. After collection the sample was shredded to a puree and stored in a refrigerator at 4 °C or frozen (depending on frequency of fresh sample collection). In the beginning of the experiment 0.4 kg of K sample and after two months up to 1 kg was flushed with 1 liter of water. Summarizing, K was added in a controlled way, while provision of F and U depended on presence and willingness of the employees present at the location.
For additional kitchen refuse characterisation there was also once a sample collected in a student flat. It consisted of the leftovers from dinner (cooked vegetables, soup, meat, bread, pasta and fruit).

2ND RUN (AC1 AND AC2)

In the second research run AC1 and AC2 were again fed with a mixture of faeces, urine and kitchen refuse and brown water and kitchen refuse respectively in the amount as normally provided by 1 and 2 individuals respectively.

PHASE 2 (UASB$_{ST}$1, UASB$_{ST}$2, AC2)

In phase 2 all used reactors were fed with the appropriate waste types in amounts as produced by one individual. The UASB septic tanks were fed with black water while accumulation reactor received a mixture of faeces and kitchen refuse. For a short period at the end of phase 2, kitchen refuse was also added to UASB$_{ST}$2.

2.5 OPERATIONAL CONDITIONS AND PARAMETERS

The combination of used reactors in all research phases and runs, conditions and parameters characterising their operation are listed in Table 2.4.

TABLE 2.4 CHARACTERISTICS OF ALL RESEARCH RUNS

Parameters	phase 1, 1st run AC1	phase 1, 1st run AC2	phase 1, 2nd run AC1	phase 1, 2nd run AC2	phase 2, AC2	phase 2, $UASB_{ST}1$	phase 2, $UASB_{ST}2$
T ,ºC	20	20		20	15	15	25
Accumulation /operational period, d	157	157	115	115	280	330	330
Total reactor volume (after feeding terminated –AC), l	1167	696	704	431	550	200	200
Average flow, $l.d^{-1}$	7.5	3.4	5.3	2.6	1.8	7.0	6.8
HRT, d						29	29
OLR, d	0.08*	0.32*	0.14*	0.32*	0.10*	0.33	0.42

* OLR in an AC reactor calculated based on the total volume after the feeding terminated

2.6 BATCH TESTS

2.6.1 MAXIMUM ANAEROBIC BIODEGRADABILITY

Maximum anaerobic biodegradability is defined as the maximum percentage of substrate total COD converted to methane. It is usually used to evaluate the percentage of organic matter that is potentially mineralizable under considered redox conditions (anaerobic in this case). The higher the biodegradability of an influent, the higher conversion potential, providing, under optimal conditions, better effluent quality and a higher production of biogas. Method to determine substrate biodegradability are described in more detail in appendix 5.

2.6.2 SLUDGE STABILITY

Sludge stability test is used to get information on the biodegradable organic fraction of the sludge from anaerobic digester. The stability of the sludge (%) is an important parameter when sludge is to be applied in the agricultural field or has to be disposed off. Description of the test can be found in appendix 6.

2.6.3 MAXIMUM SPECIFIC METHANOGENIC ACTIVITY

The specific methanogenic activity (SMA, $gCH_4\text{-}COD.gVS^{-1}.d^{-1}$) is the maximum potential of the biomass to produce methane (CH_4) from a certain substrate in a certain time interval under favourable substrate level. A procedure to determine SMA of anaerobic sludge is described in appendix 7.

2.6.4 HYDROLYSIS CONSTANT

The hydrolysis constant (d^{-1}) of complex substrates is determined based on the biodegradable polymers or total COD (Sanders, 2001). It can be used as design parameter and/or for modelling purposes. A detailed procedure to determine hydrolysis constant based on COD conversions is given in appendix 8.

2.7 SAMPLING AND ANALYSIS

In the experimental set-up as used in this research, it was difficult to obtain a representative influent sample of any of the used reactors as in practice no constant feeding pattern occurs. The chance to obtain a sample with an ideal F:U:K:W = 1:5:1:7 (W=flush water) ratio was little. An equalisation tank of 10 l was installed between the vacuum toilet and the reactor, but at the time of sampling in most extreme situation it could be filled only with urine or only faeces. In the beginning of the research (phase 1, 1st run) only a few grab samples were taken. High variations between these samples were measured. In the 2nd run of phase 1, black water was analysed separately from kitchen refuse and the average concentrations were calculated. In the 3rd run many samples were analysed and an attempt to collect a representative composite sample was made. Latter was done by taking the grab samples everyday in the beginning of the week, storing them in the fridge, mixing them together and analysing the mixture at the end of the week.

Since the AC reactor does not have any effluent, the reactor content was sampled regularly. Every week a composite sample (equal volumes from each tap) and analysed for different constituents. The effluent of the $UASB_{ST}$'s was collected in frequently emptied vessel, from which samples for analysis were collected. The development of the sludge bed in the $UASB_{ST}$ reactors was followed, by taking sludge samples from different heights. The comprehensive list of analysis associated with influent, reactors' content and effluent is listed in appendix 9. Besides the biogas production was monitored frequently from all tested reactors.

3

RESULTS INFLUENT CHARACTERISATION

3.1 INFLUENT CHARACTERISATION, SEPARATED COMPONENTS (FAECES, URINE, KITCHEN REFUSE)

In the study proceeding the start-up and operation of the pilot-plant, characterisation of separate components of concentrated waste(water) stream was conducted. In Table 3.1 the results of detailed characterisation of faeces, urine and kitchen refuse are given.

TABLE 3.1 **SEPARATE CHARACTERISATION OF FAECES, URINE AND KITCHEN REFUSE PROCEEDING PILOT-PLANT RESEARCH**

	Faeces	**Urine**	**Kitchen refuse**	
Parameter	**g.kg^{-1} wet weight**	**g.l^{-1}**	**Kitchen refuse sample originating from a student flat**	**Kitchen refuse sample originating from a hospital g.kg^{-1} wet weight**
COD_{total}	567.4 ± 76.7	12.8 ± 0.2	294.3 ± 25.1	307±23
$COD_{soluble}$	89.4 ± 7.3	11.3 ± 0.5	117.2 ± 9.0	
$COD_{particulate}$	455.0 ± 84.0	1.5 ± 0.8	177.0 ± 34.1	
VFA (COD)	8.5 ± 0.4			
TS	254.6 ± 4.2		214.9 ± 2.3	233
VS as % TS	83.2 ± 0.2		93.4 ± 0.06	97
N_{Kj}	17.8 ± 2.8	5.3 ± 1.0	9.5 ± 0.6	7.2
N_{NH4}		0.41 ± 0.01	0.47 ± 0.13	
P_{total}	6.9 ± 0.1	0.33 ± 0.01	0.67 ± 0.06	0.7
P_{PO4}	2.4 ± 0.1		0.29 ± 0.03	
pH	6.9	7.1	4.1	

All values given in Table 3.1 are expressed in g constituent per volume or wet mass of substrate. To know the daily loading originating from one individual respective volumes of produced waste have to be known. Based on the obtained results one may see that faeces and kitchen refuse are very concentrated mixtures in term of total (COD_{tot}). The majority of COD in these two wastes is particulate. Kitchen refuse is characterised by a higher VS fraction than faeces, indicating that it is easier biodegradable. Faeces has much higher nutrient content, for total phosphorus even 10 times higher than kitchen refuse. Kitchen refuse is characterised by a low pH, what can have an effect on biological treatment processes when not sufficiently buffered with co-substrate (e.g. urine/faeces). Composition of kitchen refuse is entirely dependent on the food ingredients used and degree of their processing. Direct comparisons with (wide-world) literature data should therefore be done rather carefully. Detailed composition of kitchen refuse will also depend on which fraction of it was already processed and to what extent (e.g. cooking).

3.2 INFLUENT CHARACTERISATION, MIXTURES

Except of characterisation of the separate substrates, also the influent mixtures of the tested reactors were characterised. Three types of influent mixtures are characterised below:

- *black water* consisting of faeces and urine collected in vacuum toilet (1l per each flush)
- *black water with kitchen refuse* consisting of faeces and urine collected in vacuum toilet (1l per *each* flush) and kitchen refuse also transported with vacuum. The amount of water used to flush one portion of kitchen refuse (0.2 kg) varied between 0.2-1l.
- *brown water with kitchen refuse* principally consisting of faeces and kitchen refuse. The amount of water used to transport a portion of kitchen refuse (0.2 kg) to the reactor varied between 0.2-1l.

3.2.1 BLACK WATER

Black water was frequently analysed in phase 1, 2nd run of AC1 and during phase 2 when two UASB septic tanks were fed with this mixture (Table 3.2).

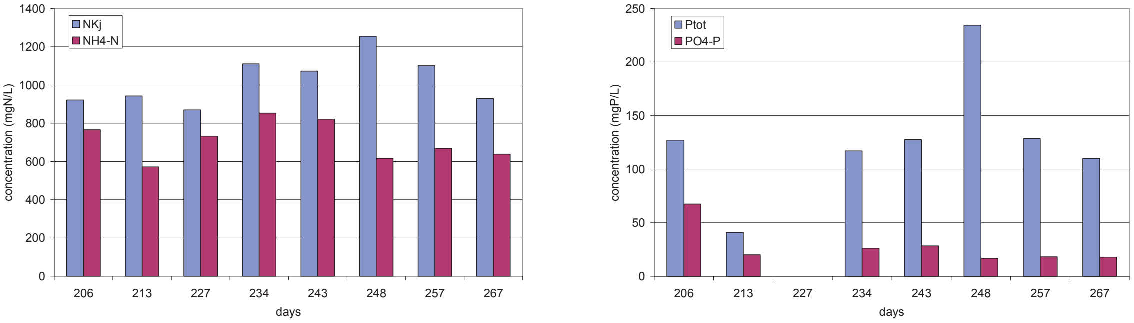
TABLE 3.2 **AVERAGE BLACK WATER CONCENTRATIONS MEASURED DURING DIFFERENT RESEARCH RUNS**

Parameter		**Influent to**		
		AC1[*]	**UASB$_{ST}$ 1**	**UASB$_{ST}$ 2**[**]
COD_{total}	$gCOD.l^{-1}$	10.0±8.9	9.5 ± 6.5	12.3 ± 7.8
$COD_{soluble}$		2.8±2.0	1.4 ± 0.48	2.0 ± 1.21
COD_{partic}		7.0±6.9	7.9 ± 6.1	9.6 ± 6.8
COD_{col}		0.16±0.2	0.2 ± 0.16	0.66 ± 0.8
VFA		1.9±1.5	0.5 ± 0.37	0.76 ± 0.64
N_{Kj}	$gN.l^{-1}$	1.0±0.63	1.0 ± 0.13	1.4 ± 0.25
N_{NH4}		0.59±0.5	0.71 ± 0.1	1.0 ± 0.96
P_{total}	$gP.l^{-1}$	0.09±0.01	0.11 ± 0.063	0.14 ± 0.06
P_{PO4}		0.06±0.03	0.03 ± 0.017	0.045 ± 0.022
TS	$g.l^{-1}$		7.41 ± 1.71	8.25 ± 3.75
VS			5.55 ± 2.05	7.53 ± 3.80
pH		8.1±0.9	8.81 ± 0.2	8.65 ± 0.4

[*] without kitchen refuse (sample taken before addition of kitchen waste); [**] generally higher concentrations were mainly caused by lower flush water use

The concentrated black water characterisation obtained in phase 1 is quite similar with that obtained for UASB$_{ST}$ 1 in phase 2. The higher concentrations of all parameters in the influent of UASB$_{ST}$ 2 were probably caused by decreased flush water consumption (<1 $l.time^{-1}$) of the used vacuum toilet. While the measured particulate COD fraction varied considerably during the research phases, nitrogen showed a more stable course. Approximately 80% of total N was in the form of ammonium. The phosphorous was mainly in the form of organically bound phosphate (Figure 3.1). Both influents to UASB$_{ST}$ were characterised by a similar, high pH.

FIGURE 3.1 EXAMPLE OF N AND P DISTRIBUTION IN BLACK WATER



3.2.2 BLACK WATER WITH KITCHEN REFUSE

The results of characterisation of mixture black water and kitchen refuse is given in Table 3.3.

TABLE 3.3 CHARACTERISATION OF A MIXTURE BLACK WATER AND KITCHEN REFUSE IN ALL RESEARCH RUNS

Parameter	Unit	Phase 1, 1st run *	Phase 1, 2nd run**	Phase 2 **
COD_{total}	$gCOD.l^{-1}$	13.30 ± 2.02	18.7	22.9
$COD_{soluble}$		3.04 ± 0.21	2.7	5.4
$COD_{particulate}$		10.26 ± 2.23	15.7	17.1
VFA		1.87 ± 0.84	1.9	
N_{Kj}	$gN.l^{-1}$		1.2	1.7
N_{NH4}		1.30 ± 0.23	0.59	
P_{total}	$gP.l^{-1}$		0.11	0.21
P_{PO4}		0.082 ± 0.01	0.059	

* measured, grab samples; ** calculated based on values measured for black water (Table 3.2, first and third column respectively) and measured concentration kitchen waste (flush volume kitchen waste as used)

3.2.3 BROWN WATER WITH KITCHEN REFUSE

In all three research runs always the same reactor – AC2 was fed with a mixture of faeces and kitchen refuse. Due to little dilution with urine and flush water this wastewater was characterised by significant organic matter content. Also nutrient concentrations were higher than in other mixtures mainly due to the fact that both nutrients sources - urine and faeces were frequently present in one water volume. As it can be seen from Table 3.5, the measured composition of this stream varied a lot in different research runs.

The first characterization was obtained from few grab samples, the second is outcome of calculation of a separate characterization of brown water (Table 3.5) and kitchen refuse. The values from the last column represent average from frequent analysis of the total mixture.

Some factors which probably influenced these high variations are:

- some differences in flush water consumption during different runs;
- flushing different amounts of kitchen residues (0.6-1 kg) with one volume of flush water
- different types of samples during different research phases (grab samples, calculated values, semi-composite samples)

The theoretical value of COD_{total} for this type of influent when 1l of water is used to flush is expected to be in the range of 45-65 $g.l^{-1}$ depending on how much water is used to transport kitchen refuse to the reactor (Table 3.11).

TABLE 3.4 CHARACTERISATION OF A MIXTURE BROWN WATER AND KITCHEN REFUSE IN ALL RESEARCH RUNS

Parameter	Unit	Phase 1, 1st run	Phase 1, 2nd run*	Phase 2
COD$_{total}$	gCOD.l^{-1}	66.00 ± 26.05	53.6	31.8±10.5
COD$_{soluble}$		4.0 ± 1.30	10.1	5.3±2.1
COD$_{part}$		62.00 ± 27.45	43.4	23.9±11.9
COD$_{col}$				3.9±3.0
VFA		3.09 ± 1.89	8.9	3.4±1.1
N$_{Kj}$	gN.l^{-1}		1.9	1.7±0.18
N$_{NH4}$		1.22 ± 2.54	0.78	0.86±0.086
P$_{total}$	gP.l^{-1}		0.35	0.23±0.034
P$_{PO4}$		0.21 ± 0.04	0.31	0.13±0.068

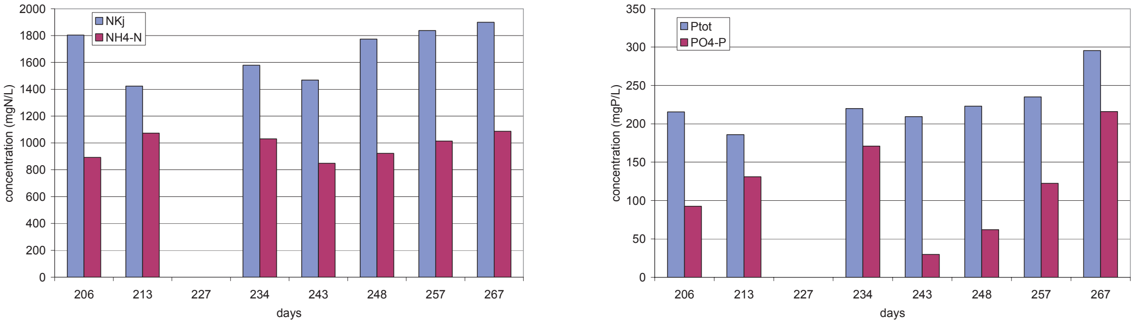
*calculated based on values measured for brown water and measured concentration kitchen waste (flush volume kitchen waste as used)

In Figure 3.2 an example of nutrient distribution as measured in the 3rd run in this concentrated influent is shown. Ammonium constituted approximately 60% of total N (lower than in black water) while phosphate 57% (higher than in black water).

TABLE 3.5 CHARACTERISATION OF BROWN WATER OBTAINED IN THE SECOND RESEARCH RUN

Parameter	Unit	Phase 1, 2nd run
COD$_{total}$	gCOD.l^{-1}	38.8 ± 20.3
COD$_{soluble}$		10.1 ± 1.5
COD$_{particulate}$		28.7 ± 21.1
VFA		8.9 ± 2.7
N$_{Kj}$	gN.l^{-1}	1.5±0.82
N$_{NH4}$		0.78 ± 0.65
P$_{total}$	gP.l^{-1}	0.32±0.25
P$_{PO4}$		0.31 ± 0.072
pH		6.3±0.7

FIGURE 3.2 NUTRIENTS DISTRIBUTION IN THE INFLUENT TO AC2 IN THE 3RD RUN



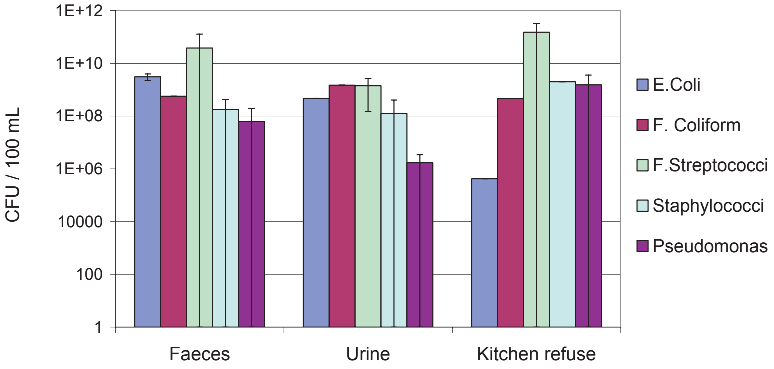
3.3 BACTERIOLOGICAL CHARACTERIZATION OF FAECES, URINE AND KITCHEN REFUSE

In separate samples of faeces, urine and kitchen refuse *E.coli*, *F.coliform (FC)*, *F.streptococci (FS)*, *Staphylococci* and *Pseudomonas* were measured (Table 3.6, Figure 3.3). These bacteria were used as indicators for the presence of pathogens.

TABLE 3.6 BACTERIOLOGICAL CHARACTERISATION OF FAECES, URINE AND KITCHEN REFUSE

Sample	E.coli	F.coliform	F.streptococci	Staphylococci	Pseudomonas CFU.100ml^{-1}
Faeces	3.08E+09± 8.74E+08	5.70E+08	3.81E+10± 8.91E+10	1.78E+08± 2.38E+08	6.18E+07± 1.34E+08
Urine	4.70E+08	1.50E+09	1.41E+09± 1.26E+09	1.26E+08± 2.80E+08	1.70E+06± 1.70E+06
Kitchen refuse	4.20E+05	4.60E+08	1.51E+11± 1.68E+11	2.00E+09	1.52E+09± 2.09E+09

FIGURE 3.3 **BACTERIAL DISTRIBUTION IN FAECES, URINE AND KITCHEN REFUSE**

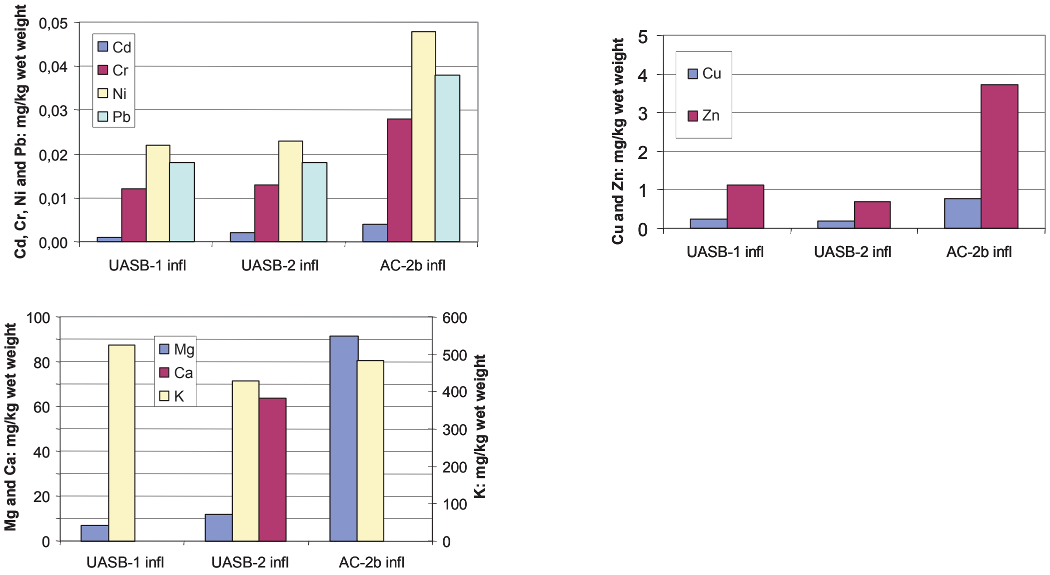


Bacterial concentrations in the different samples ranged from 10^5 CFU.100 ml^{-1} (*E.coli* in kitchen refuse) to 10^{11} CFU.100 ml^{-1} (FS in kitchen refuse) and show different distributions in faeces, urine and kitchen refuse. *Faecal coliform* had similar concentrations in all samples. Faecal Streptococci concentrations were the highest of all measured indicators in faeces and kitchen refuse. Separately collected for this purpose urine (using clean portable toilets) was not expected to contain such high concentrations as measured. When comparing with faeces and kitchen refuse samples only the *Pseudomonas* and *Faecal Streptococci* concentrations were lower in urine. The concentrations of the other indicators from urine are roughly in the same range as those from faeces and kitchen refuse. Höglund (2001) reports *E.coli* and *F.coliform* concentrations of 10^3 CFU.100 ml^{-1} and 10^5 –10^7 CFU.100 ml^{-1} respectively, in urine collections tanks in Sweden. *E.coli* and *F.coliform* concentrations in urine found in this study exceed these values with a few orders of magnitude. However, a relatively fresh urine sample (< 3 days) was used in this study, while the urine analysed by Höglund (2001) was stored in a collection tank where bacterial numbers were decreasing during the storage. Low concentrations of *E.coli*, but high of *F. streptococci*, *staphylococci* and *pseudomonas* were measured in kitchen refuse samples. The last indicators were clearly higher than in faeces or urine.

3.4 HEAVY METALS IN CONCENTRATED MIXTURES

Heavy metals concentrations were determined for two types of influent: concentrated black water and in a mixture of brown water and kitchen refuse. The results are presented in mg per kg wet weight (or l). As expected the influent of the AC2 reactor was more concentrated than that of the UASB septic tanks also in term of these elements. Faeces and especially kitchen refuse contain higher concentrations of metals, while very low metal concentrations are usually found in urine. The concentration of Cd, Cr, Ni, Pb and Zn in the influents of $UASB_{ST}$ 1 and $UASB_{ST}$ 2 were lower than in the influent of AC2. For Cr, Ni and Pb the concentration in the AC2 influent is about twice as high as the concentration in the UASB influents while for Zn it is even 4 times higher (Figure 3.4, Table 3.9).

FIGURE 3.4 CONCENTRATION OF ELEMENTS IN THE INFLUENT TO $UASB_{ST}$ (F+U) REACTORS AND AC2 (F+K)



The concentration of copper was approximately 4 times higher in the influent of AC2 than in black water. The concentration of potassium was similar for all types of measured influents, while Mg in concentrated AC2 influent was approximately 10 times higher than in black water.

TABLE 3.6 ELEMENTS MEASURED IN BLACK WATER AND A MIXTURE OF BROWN WATER AND KITCHEN REFUSE

	Element in mg.kg wet weight^{-1}								
Reactor	**Cd**	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**	**K**	**Mg**	**Ca**
$UASB_{ST1}$	0.001	0.012	0.234	0.022	0.018	1.124	523	6.7	n.d.
$UASB_{ST2}$	0.002	0.013	0.185	0.023	0.018	0.682	428	11.8	63.8
AC2	0.004	0.028	0.760	0.048	0.038	3.724	482	91.4	n.d.

When comparing with literature values (Table 1.2) recalculated to concentrations in concentrated black water as used in this study a majority of elements is of the same (Cd, Cu, Zn, K) or similar (Cr, Ni) order. Much higher concentrations were found for Pb (6 times higher). For the influent to AC2 (brown water with kitchen refuse) similar concentration of the elements when comparing with literature data were again measured for Cd, Cu, Zn and K while Ni, Cr and Pb were respectively ca. 2, 5 and 7 times higher.

3.5 HYDROLYSIS CONSTANT (K_H)

In the first run the k_H values were estimated for two types of influent wastewater (AC1 and AC2) under three different temperatures, 20, 30 and 50°C. For tests at 20 and 30°C sludge from respective accumulation systems was used as inoculum. To estimate k_H at 50°C thermophically digested manure was used (El-Mashad, 2003).

There were no significant differences in k_H estimated for 20 and 30°C for the two different substrate mixtures. Low values of 0.06 d^{-1} were obtained at 50 °C. The calculated (k_H) values for black water in combination with kitchen refuse found in this research are in the same range as reported by Mahmoud (2002) for 25°C (0.07 d^{-1}) and 35 °C (0.113 d^{-1}) for primary sludge. In the second run the k_H of the influent wastewater of AC1 and AC2 system was again assessed for 20 and 30°C (Table 3.7). The obtained values under considered T conditions are comparable with values obtained in the previous run. Somewhat higher k_H values were obtained for 30°C than for 20°C. There was not remarkable difference in k_H in relation to substrate composition.

TABLE 3.7 THE HYDROLYSIS CONSTANT K_H (D^{-1}) OF TWO TYPES OF INFLUENT IN RELATION TO TEMPERATURE

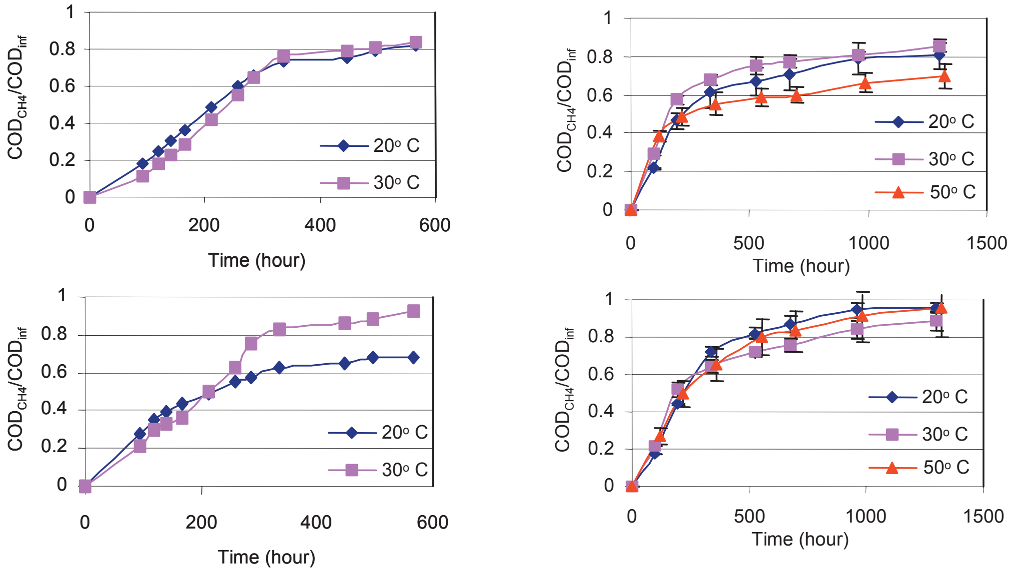
	black water + kitchen waste			brown water + kitchen waste		
	Temperature					
	20 °C	30 °C	50 °C	20 °C	30 °C	50 °C
1st run	0.093 (0.016)**	0.093 (0.003)	0.06 (0.76)	0.08	0.095 (0.018)	
2nd run	0.11	0.13		0.1	0.11	

** Standard deviation

3.6 ANAEROBIC BIODEGRADABILITY OF INFLUENT

The biodegradability of two influent mixtures, which served as influent to the tested AC reactors was assessed in the first research run in two series of batch tests. Figure 3.5 shows the cumulative biogas production at different temperatures, 20 and 30°C for the first batch and 20, 30 and 50 °C for the second batch.

FIGURE 3.5 COURSE OF METHANE PRODUCTION IN INOCULATED BATCHES CONTAINING INFLUENT WASTEWATER OF AC1 (TWO UPPER PLOTS) AND AC2 (TWO LOWER PLOTS) AT DIFFERENT TEMPERATURES



The estimated values for the anaerobic biodegradability of black water in combination with kitchen refuse (Table 3.8) were higher than reported in the literature 71% for black water, 74 for Dutch domestic sewage (Elmitwalli, 2000) and 50% for Dutch primary sludge (Mahmoud, 2002). The higher biodegradability may result from the presence of kitchen refuse, which is highly biodegradable (Veeken and Hammelers, 1999).

FIGURE 3.8 **MAXIMUM BIODEGRADABILITY (%) OF TWO TYPES OF CONCENTRATED MIXTURES OBTAINED UNDER DIFFERENT TEMPERATURES**

BLACK WATER + KITCHEN REFUSE			BROWN WATER + KITCHEN REFUSE		
TEMPERATURE, ^{0}C					
20	30	50	20	30	50
81	84	69	81	90	95

In the 3rd run a biodegradability of different substrate mixtures under two higher temperatures was estimated for a limited period of 28 days (Table 3.9). Lower values obtained at 35°C can be explained by a much shorter incubation period, so the biogas production did not cease. On the other hand at 55°C also lower biodegradability was obtained although biogas production in both batch tests stopped (black water + kitchen refuse) or almost stopped (brown water + kitchen refuse).

TABLE 3.9 **BIODEGRADABILITY (%) OF TWO TYPES OF CONCENTRATED MIXTURES AT TWO HIGHER TEMPERATURES AFTER 28 DAYS OF INCUBATION**

Black water	Brown water + kitchen refuse	Black water	Brown water + kitchen refuse
Temperature			
35°C		55°C	
54	61	54	71

3.7 FINAL REMARKS

Detailed characterisation of concentrated wastewater streams subjected to anaerobic digestion in DESAR concept was performed. First separate components of concentrated wastewater streams were characterised: faeces, urine and kitchen refuse. Afterwards composition of different types of concentrated mixtures forming influent to the tested reactors was obtained. This section ends up with tables useful for design of a system treating concentrated domestic wastewater streams. In Table 3.10 calculated, based on this research, daily loadings of concentrated wastewater produced by one individual are shown. These values can be compared with literature values from Table 1.2. Both characteristics differ significantly from each other in several points. One has to realise however that limited number of samples taken from a small group of people may have an influence on the result. Furthermore the volumes needed to calculate daily loadings are literature and not measured values.

TABLE 3.10 **DAILY LOADING OF FAECES, URINE AND KITCHEN REFUSE AS PRODUCED BY ONE INDIVIDUAL CALCULATED BASED ON THE CHARACTERISATION OF SEPARATE COMPONENTS (TABLE 3.1). DAILY VOLUMES ARE ASSUMED BASED ON OTHER INTERNAL RESEARCH (DE MES, 2001) FOR FAECES AND ON LITERATURE FOR URINE AND KITCHEN REFUSE**

Parameters	Faeces	Urine	Kitchen refuse
Volume (l or kg.p^{-1}.d^{-1})	0.138$^{1)}$	1.25$^{1)}$	0.2$^{1)}$
COD_{total}	78.3	16	58.9
COD_{sol}	12.3	14.1	23.4
$COD_{part+col}$	62.8	1.9	35.4
N_{tot}	2.45	6.6	1.9
P_{tot}	0.95	0.4	0.13
pH	6.9	7.1	4.1

$^{1)}$ literature values

In Table 3.11 theoretical compositions of wastewater mixtures as produced by one individual are calculated based on daily loadings of constituents (Table 3.10) and a volume of water used to collect and transport a given type of waste(water). Except for a brown water, a comparison between calculated characterisation and measured influent shows a lot of similarities.

TABLE 3.11 **THEORETICAL CHARACTERISATION OF VARIOUS MIXTURES USED AS THE FEED TO THE TESTED REACTORS WITHIN HERE DESCRIBED STUDY (G. L^{-1}.D^{-1})**

Parameters	Black water	Brown water	Black water + kitchen refuse (0.5 l water)	Black water + kitchen refuse (1 l water)	brown water* + kitchen refuse (0.5 l water)	brown water* + kitchen refuse (1 l water)
Volume (l.p^{-1}.d^{-1})	7.388	1.138	8.088	8.588	2.088	2.588
COD_{total}	12.8	68.8	18.9	17.8	67.2	54.2
COD_{sol}	3.6	10.8	6.16	5.8	18.5	14.9
$COD_{part+col}$	8.7	55.1	12.37	11.6	47.2	38.1
N_{tot}	1.2	2.1	1.35	1.28	2.7	2.2
P_{tot}	0.18	0.83	0.18	0.17	0.56	0.45

* assumed presence of one portion of urine (1/5th of 1.25l) in the provided faeces

4

ACCUMULATION SYSTEM (AC UNDER 15 AND 20⁰C)

4.1 RESULTS

4.1.1 OPERATIONAL PROBLEMS

Accumulation reactor alone, generally and as applied in this research, is a very simple technical system, involving no mechanical parts. Thanks to it no operational problems were faced. Few operational nuisances were related to the vacuum system. The vacuum pump of AC2 in the 3rd run (15°C) was broken twice, due to a failure when taking the influent sample.

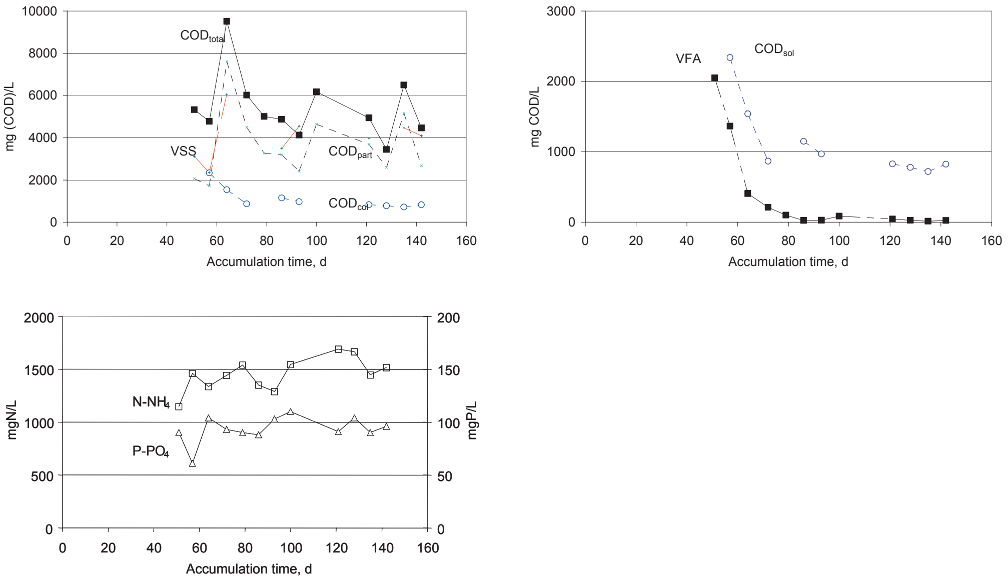
Another nuisance related rather to research objective than comfort of the user was that at a certain moment more flush water was used in the toilet serving AC1 than assumed (> 1l). The flush button could be re-adjusted. There were also some problems with gas meters resulting that in some research periods no continuous monitoring of gas production took place. As often mentioned throughout this report it was difficult to obtain a representative influent sample for wastewater characterisation.

4.1.2 REACTOR CONTENT COMPOSITION (20⁰C, 150 DAYS FILLING TIME)

Since no effluent is discharged from an AC reactor during the accumulation period, to control the reactor performance only the reactor content could be monitored. As described previously volumetrically identical samples were taken from all available (= filled with sludge-wastewater mixture) taps and mixed together. Afterwards their composition was determined. In order to follow the development of the process this was repeated frequently during the research period.

The results of the AC1 content characterisation throughout the 1st accumulation period are shown in Figure 4.1.

FIGURE 4.1 CONCENTRATION OF COD FRACTIONS (COD_{TOT}, COD_{PART}, COD_{COL}, COD_{SOL}), VFA-COD, NH_4^+-N AND P-PO_4 AS A FUNCTION OF ACCUMULATION TIME, DURING OPERATION OF AN AC1 SYSTEM WITH CONCENTRATED BLACK WATER AND A KITCHEN REFUSE AS A FEED AT 20^{0}C AND FILLING TIME OF 150 DAYS

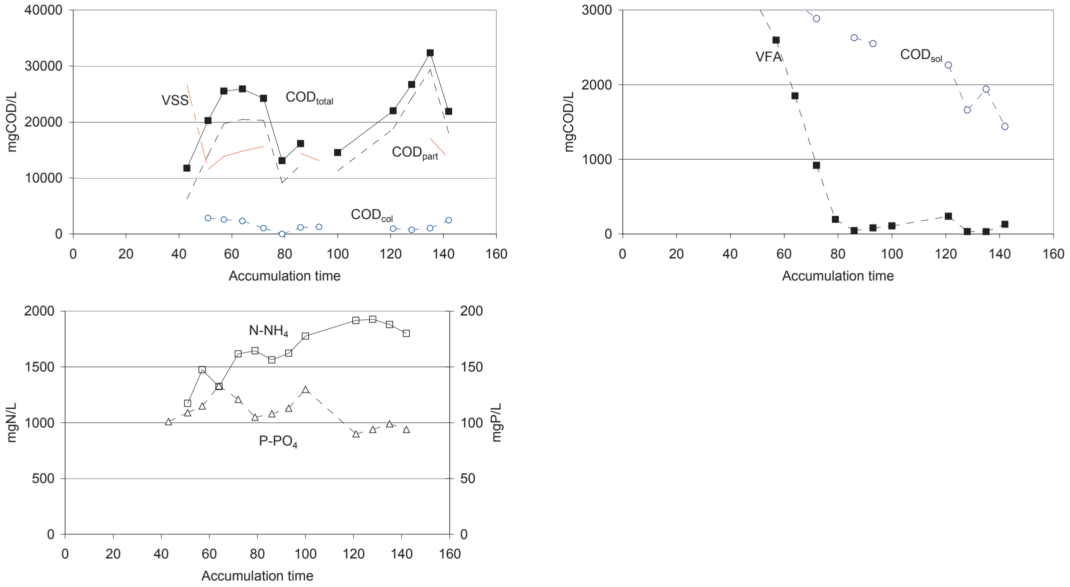


The results show a stable performance of the system. High concentration of ammonium (average 1450 mgN.l^{-1}, increasing throughout the accumulation period) seemed not to inhibit the anaerobic digestion process as the VFAs concentration decreased in time up to very low values (from more than 2050 to average of 40 mgCOD.l^{-1} after 80th day of operation).

The soluble phosphate content remained rather stable and attained (relatively low as compared to the influent total P concentration) an average concentration of 93 mgP.l^{-1}. The pH value was also stable and in a suitable to anaerobic digestion ange (pH= 7.4).

The AC2 (fed with brown water and kitchen refuse) was characterised by higher concentrated reactor content in terms of all parameters (Figure 4.2) except of soluble phosphate. Also in this reactor even higher ammonium concentration of approximately 1800 mgN.l^{-1} seemed not to inhibit the process. After 80 days low concentrations of VFA were measured (< 100 mgCOD.l^{-1}). The concentration of soluble and colloidal COD was higher than in AC1 (factor 2.4 and 2.0 respectively). The concentration soluble ortho-phosphate was similar as in AC1 (104 vs. 93 mgP.l^{-1}).

FIGURE 4.2 CONCENTRATION OF COD FRACTIONS (COD_{TOT}, COD_{PART}, COD_{COL}, COD_{SOL}), VFA-COD, NH_4^+-N AND P-PO_4 AS A FUNCTION OF ACCUMULATION TIME, DURING OPERATION OF AN AC2 SYSTEM WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE AS A FEED AT 20°C AND FILLING TIME OF 150 DAYS



In the shorter verification run (2nd run) of both accumulation systems (AC1 and AC2) the quality of the reactor content was again regularly controlled. In Figure 4.3 the course of COD fractions and VFA for AC1 and AC2 are presented. A significant difference in the strength (concentration) of reactors' content regarding total and particulate COD can be clearly seen. Smaller differences are shown for colloidal and soluble COD. The system inoculated with the adapted sludge from the 1st run started to perform stable (expressed by low VFA concentration) already after 40th days of accumulation. In general the results obtained in the 2nd run are very similar to those obtained in the 1st run. It is interesting to see that both reactors, in both runs reached a stable performance (expressed as low VFA values) at the same time, although the influent strength differed significantly in both reactors. Regarding fate of nutrients in both reactors (Figure 4.4), very similar results were obtained for N_{Kj} and soluble phosphate. The concentration of ammonium was higher in AC2. A significant difference was also observed for total phosphate, where 3 fold higher concentrations were obtained in AC2. It seemed that a significant phosphate precipitation occurred in AC2 system, as the inocolum sludge (from the first run with an accumulation period of 150 days) had a high P_{total} content (see 4.1.3) and the ortho P in the system was lower than that of the influent.

FIGURE 4.3 COURSE OF COD FRACTIONS (COD_{TOT}, COD_{PART}, COD_{COL}, COD_{SOL}) AND VFA-COD AS A FUNCTION OF ACCUMULATION TIME, DURING OPERATION OF AN (□) AC1 SYSTEM WITH CONCENTRATED BLACK WATER AND KITCHEN REFUSE AND (■) AC2 SYSTEM WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE AS A FEED AT 20⁰C AND FILLING TIME OF 105 DAYS

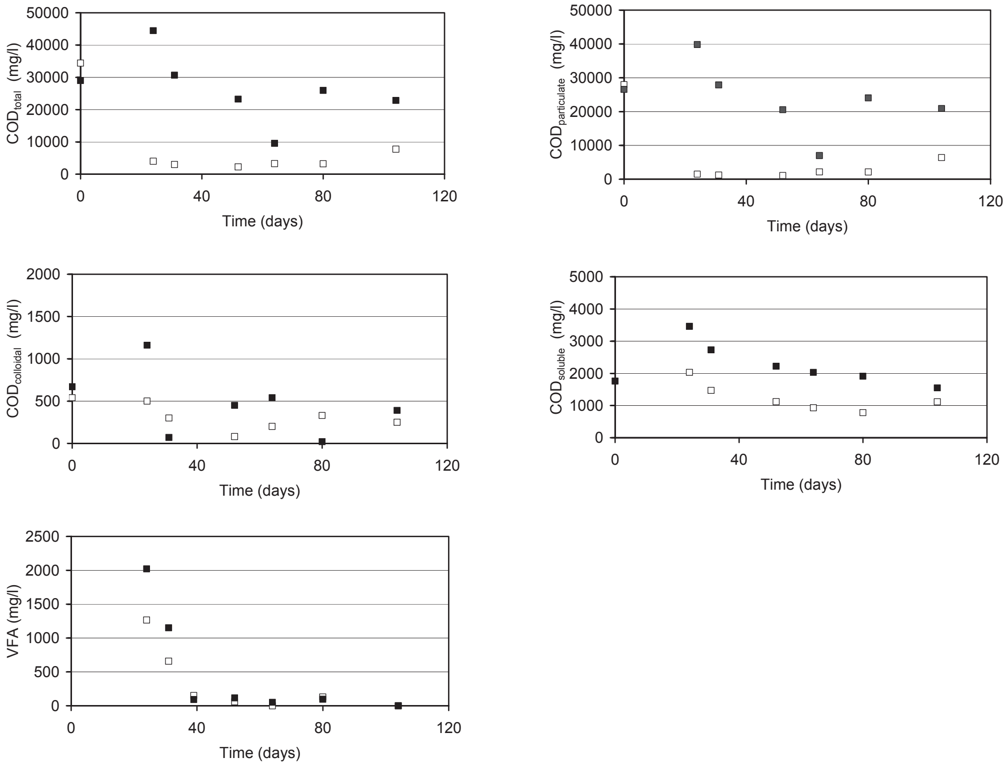
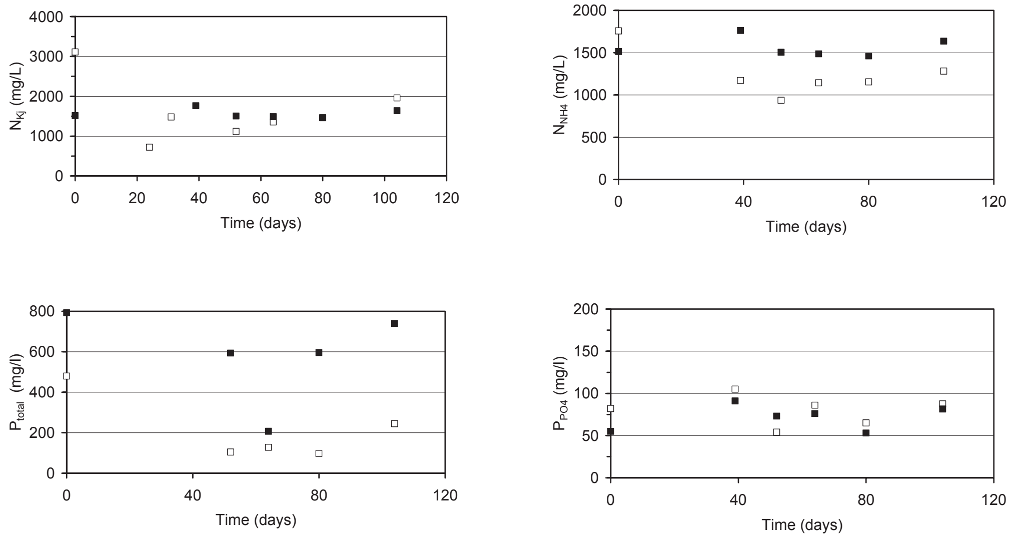
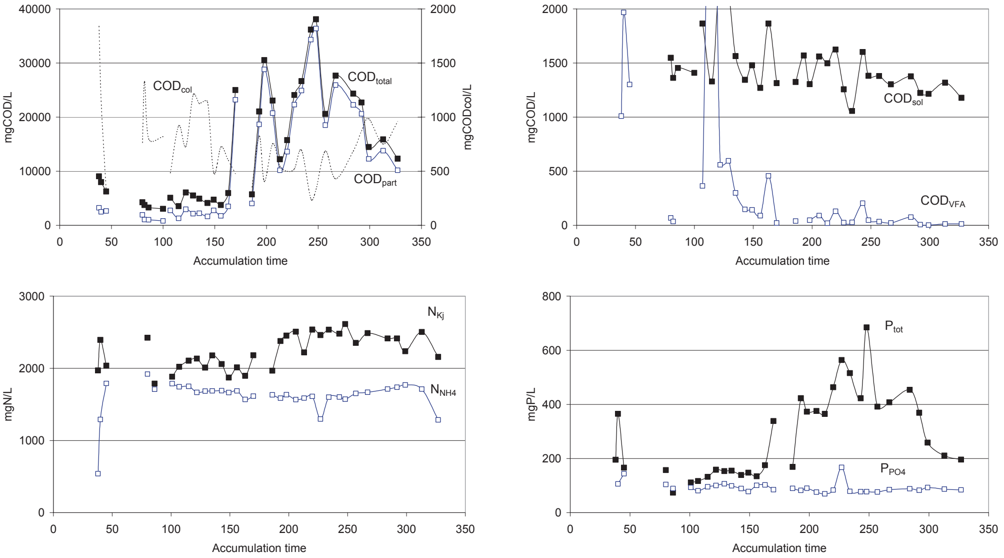


FIGURE 4.4 COURSE OF TOTAL AND SOLUBLE NUTRIENTS (N_{KJ}, P_{TOT}, NH_4^+-N AND P-PO_4) AS A FUNCTION OF ACCUMULATION TIME, DURING OPERATION OF AN (□) AC1 SYSTEM WITH CONCENTRATED BLACK WATER AND KITCHEN REFUSE AND (■) AC2 SYSTEM WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE AS A FEED AT 20⁰C AND FILLING TIME OF 105 DAYS



The average composition of the reactor content of AC2 during an accumulation period of 280 days (excluding break in operation) at 15°C (3rd run) is shown in Figure 4.5. The average COD_{total} was similar as in the previous runs - 22.4 $g.l^{-1}$. Soluble COD attained again very low values of 1350 $mg.l^{-1}$. Most VFA formed during the process was completely converted at the end of the accumulation period. The same pH was measured as in the other runs, in an optimal range of 7.2-7.3. The concentration of total N was the highest of all three runs and attained an average of 2400 $mgN.l^{-1}$ at the end of this accumulation period. This high concentration was due to the use of an excessive amount of inoculum, containing a high concentration of ammonium and particulate nitrogen. The ammonium level of 1600 $mgN.l^{-1}$ was similar in all three runs. Again, low concentration of soluble phosphate (90 $mgP.l^{-1}$) confirm the occurrence of phosphate precipitation. The average total phosphate concentration was lower than expected suggesting that the most concentrated layer of precipitate was located below the 1st sampling tap.

FIGURES 4.5 COURSE OF COD FRACTIONS (COD_{TOT}, COD_{PART}, COD_{COL}, COD_{SOL}), VFA-COD AND TOTAL AND SOLUBLE NUTRIENTS (N_{KJ}, P_{TOT}, NH_4^+-N AND P-PO_4) AS A FUNCTION OF ACCUMULATION TIME, DURING OPERATION OF AN AC2 SYSTEM WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE AS A FEED AT 15°C AND FILLING TIME OF 280 DAYS (50 DAYS IN THE BEGINNING WITHOUT THE FEEDING)



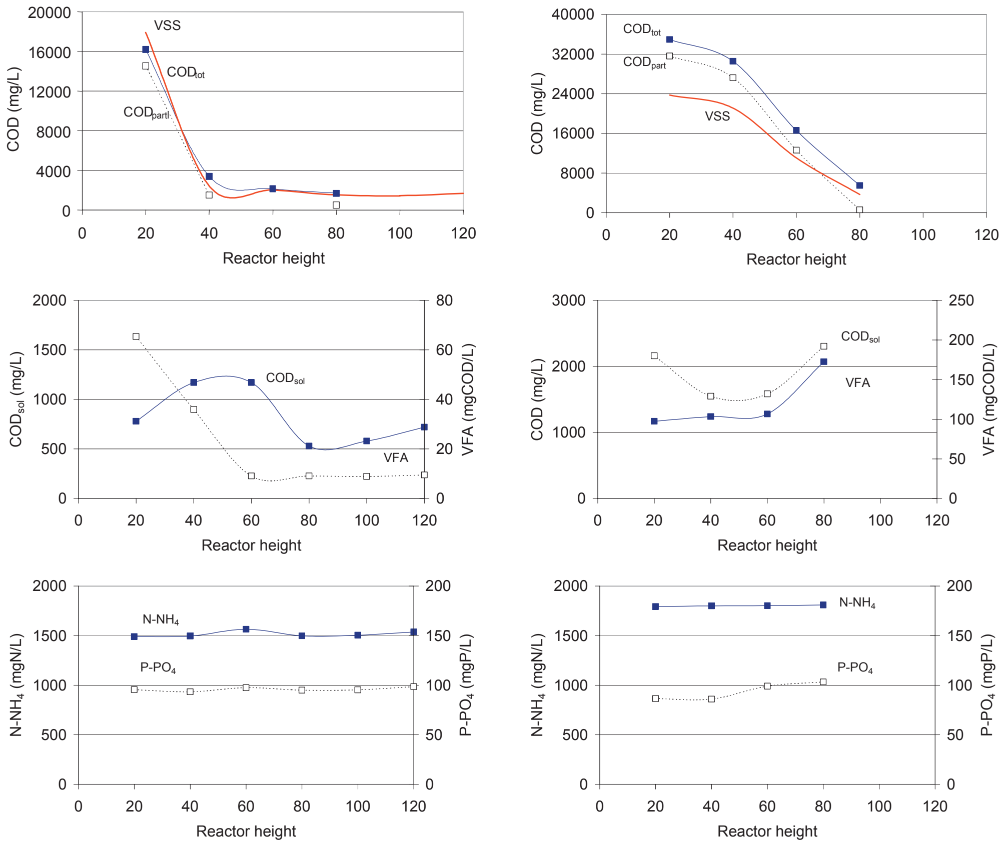
4.1.3 STRATIFICATION

Except for the average reactor content composition (mixed sample), the height profile was performed regularly during reactor operation in order to evaluate the difference between sludge and supernatant quality. This kind of information can be important when separation between sludge and water phase after emptying the reactor takes place (e.g. for further treatment) or to estimate in this way treatment efficiency. Not stirred reactors (also up-flow reactors) will have a stratified content, with a sludge layer on the bottom (the height and characteristics depending on the type of the reactor, accumulation period, process performance, influent characteristics) and supernatant in the upper zone. In some cases a scum layer can be observed on the top layer of the reactor content.

In Figure 4.6 the AC1 and AC2 profiles for various components at the end of the 1st accumulation period are shown. The concentrations of particulate components in the AC1 system show that the system was highly stratified, indicating, as expected, poor mixing of the reactor content. At the end of the 1st run the sludge occupied only the lowest part in the system (up to tap 1). The soluble compounds (N_{NH4}, P_{PO4}, pH, VFA and COD_{sol}) were generally equally distributed over the whole reactor content.

In AC2 only the upper volume of the system (4th tap) contained diluted wastewater after 143 days of operation indicating that there was better sludge distribution in AC2 system as compared to AC1 system. But on the other hand AC2 was fed with more concentrated matter in terms of particulate material (according to the analysis, 4 fold higher). Also the amount of inoculum was higher in AC2 than in AC1 and it was more concentrated. The concentrations of the dissolved components in AC2 did not vary significantly with the reactor height.

FIGURE 4.6 STRATIFICATION IN AC1 (LEFT) FED WITH CONCENTRATED BLACK WATER AND A KITCHEN REFUSE AND AC2 (RIGHT) FED WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE AT THE END OF THE 1ST RUN AT 20^{0}C AND FILLING TIME OF 150 DAYS. PROFILES OF PARTICULATE (COD_{TOT}, COD_{PART},VS) AND SOLUBLE (VFA-COD, NH_4^+-N AND P-PO_4) FRACTIONS ARE PLOTTED AGAINST REACTOR HEIGHT



Figures 4.7 and 4.8 represent the wastewater profiles in AC1 and AC2 at the end of the 2nd run (day 105). A clear stratification was observed in both reactors, stronger in AC1 due to the more diluted character of its influent. The accumulation of significant amounts of phosphate in the sludge layer can be observed in both reactors, higher in AC2 (ca. 1350 and 550 mgP. l^{-1} respectively). The soluble components were evenly distributed. The soluble phosphate was similar in both reactors. More ammonium was measured in AC2. Significant amount of particulate nitrogen accumulated in the sludge.

FIGURE 4.7 **STRATIFICATION IN AC1 FED WITH CONCENTRATED BLACK WATER AND KITCHEN REFUSE (LEFT) AND AC2 FED WITH CONCENTRATED BROWN WATER AND KITCHEN REFUSE (RIGHT) AT THE END OF THE 2ND RUN AT 20^{0}C AND FILLING TIME OF 105 DAYS. PROFILES OF PARTICULATE (COD_{TOT}, COD_{PART}, COD_{COL}) AND SOLUBLE (COD_{SOL}) FRACTIONS ARE PLOTTED AGAINST REACTOR HEIGHT**

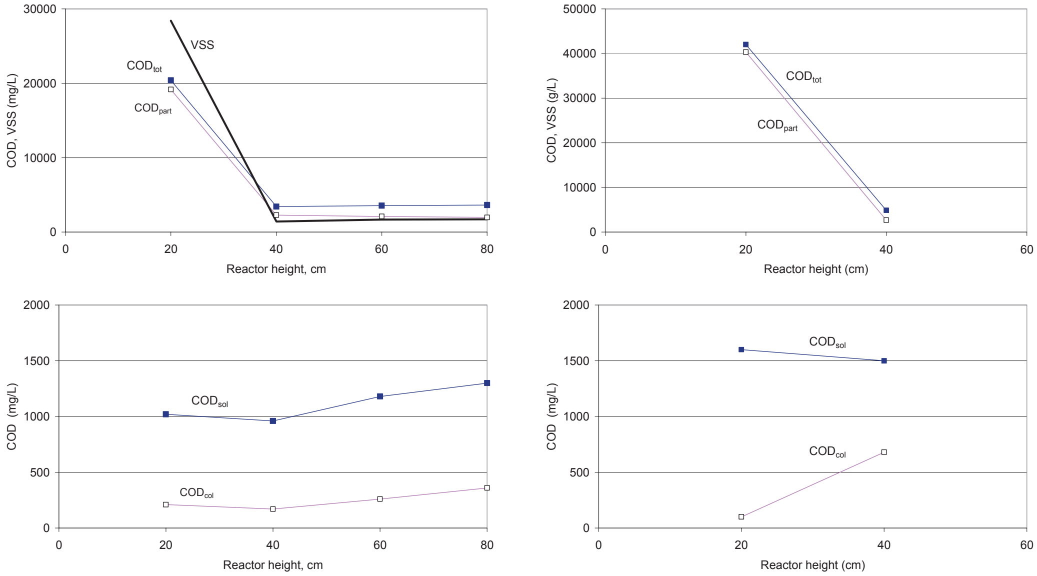
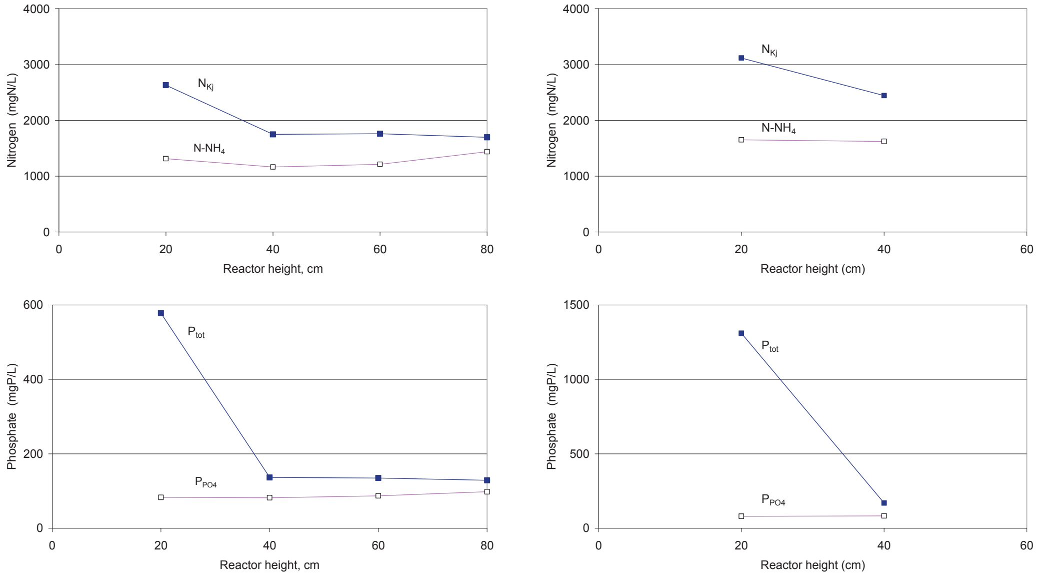
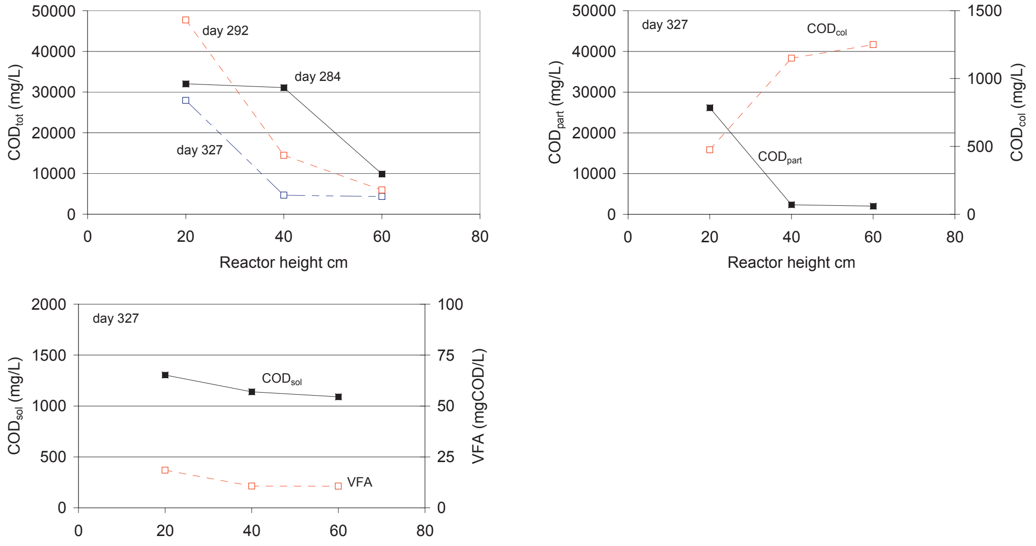


FIGURE 4.8 **STRATIFICATION OF NUTRIENTS IN AC1 FED WITH CONCENTRATED BLACK WATER AND A KITCHEN REFUSE (LEFT) AND AC2 FED WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE (RIGHT) AT THE END OF THE 2ND RUN AT 20^{0}C AND FILLING TIME OF 105 DAYS**



The profiles of the wastewater constituents measured during the 3rd run of AC2 are shown in the Figure 4.9. This last accumulation run lasted the longest of all others, reaching 8 months (exclusive more than 1 month of non-fed period in the beginning of operation). A technical problem at the end of the operational period created turbulence inside the reactor mixing up the reactor content. Hence on the 284th day measurement, the total COD values for first and second tap were quite similar and high, 32 and 31.1 g.l^{-1} respectively while for the third tap the total COD increased to 9.9 g.l^{-1}. During the off (unfed storage, post-digestion) period, the reactor content was again characterised. Within a week the accumulated influent slowly settled, increasing the COD$_{total}$ values at 292nd day from the first tap up to 47.7 g.l^{-1} and decreasing the COD of analysed supernatant at the 3rd tap to 5.9 g.l^{-1}. The settling and thickening of the sludge stopped and the COD$_{total}$ of supernatant decreased slowly further due to ongoing biological conversions.

FIGURE 4.9 STRATIFICATION IN AC2 FED WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE AT THE END OF THE 3RD RUN AT 15^{o}C AND FILLING TIME OF 280 DAYS. PROFILES OF PARTICULATE (COD$_{TOT}$, COD$_{PART}$, COD$_{COL}$) AND SOLUBLE (VFA-COD, COD$_{SOL}$) FRACTIONS ARE PLOTTED AGAINST REACTOR HEIGHT



The soluble COD was evenly distributed over the reactor height and attained a low average value of 1.3 gCOD.l^{-1}. The system accumulating the influent wastewater for a significant period of time ensured that there was enough methanogenic potential to convert the soluble fraction both from influent and produced in the reactor even though the reactor operated at low temperature. The results obtained in this last run are quite comparable with the previous runs using the same reactor and similar feeding but operating under higher temperatures of 20^{o}C. The comparison for COD fractions is given in Table 4.1.

TABLE 4.1 COMPARISON OF COD FRACTIONS INSIDE THE AC2 SYSTEM FED WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE AT THE END OF ACCUMULATION PERIOD OF THE ALL THREE RESEARCH RUNS (IN G.L^{-1})

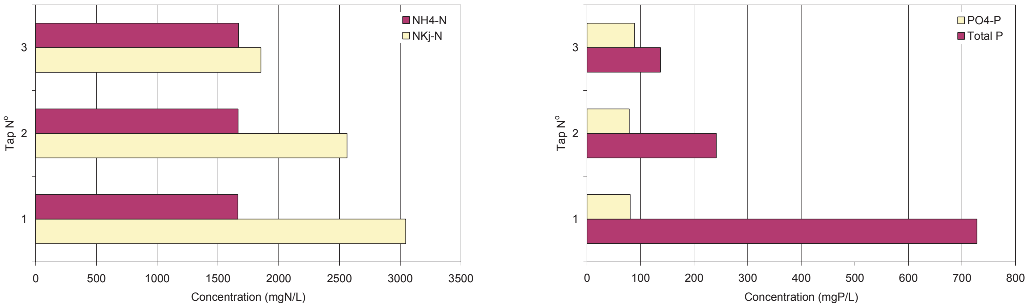
	AC2								
Tap	**1st run, 20^{o}C, 150 days OLR$^{1)}$= 0.32 kg.m^{-3}.d^{-1}**			**2nd run, 20^{o}C, 105 days OLR=0.32 kg.m^{-3}.d^{-1}**			**3rd run, 15^{o}C, 280 days OLR=0.10 kg.m^{-3}.d^{-1}**		
	COD$_{tot}$	**COD$_{part+col}$**	**COD$_{sol}$**	**COD$_{tot}$**	**COD$_{part+col}$**	**COD$_{sol}$**	**COD$_{tot}$**	**COD$_{part+col}$**	**COD$_{sol}$**
1	34.9	33.8	1.2	42.0	40.4	1.6	36.45	35.06	1.39
2	30.6	29.3	1.2	4.8	3.3	1.5	14.57	13.24	1.33
3	16.6	15.3	1.3				6.19	4.96	1.23
4	5.5	3.4							

$^{1)}$ Organic loading rate is calculated based on the filled reactor volume

Similar to the previous runs, the VFA concentrations attained low values (<50 mgCOD.l^{-1}) but later in the accumulation period (day 150th). There were two main reasons for this late reached stable operation: lower operational temperature and stop of the feeding after approximately 50 days (at that time VFA started to decline) for longer than 1 month so the system had to be restarted. During the storage period, after feeding was stopped, the average VFA concentration dropped even further to less than 10 mgCOD.l^{-1}. Based on the fact that the VFA concentration was so low, the effluent COD_{sol} fraction probably consisted mainly of inert material.

The nutrient concentrations in each tap of AC2 during the 3rd run is shown in Figure 4.10. The average concentration of N_{NH4} was approximately 1600 mgN.l^{-1}. The N_{Kj} concentration was different for each tap corresponding with the stratification of the particulate components in the reactor. The inoculum sludge retained from the previous run (phase 1 2nd run) and low influent flow contributed to a higher N_{Kj} concentration than measured in the influent. The average N_{Kj} were 2873±281, 2158±244 and 1943±53 mgN.l^{-1} for the 1st, 2nd and 3rd tap respectively. A similar distribution was observed for phosphate. The soluble P_{PO4} concentrations in every tap were similar, around 90 mgP.l^{-1}. For the total phosphate, the concentrations decreased along the reactor height with the highest concentration at the bottom of the reactor: 634±214, 263±160 mg.l^{-1} and 139±26 mg.l^{-1} at the 1st, 2nd and 3rd tap respectively.

FIGURE 4.10 **NUTRIENTS PROFILE INSIDE THE AC2 FED WITH CONCENTRATED BROWN WATER AND A KITCHEN REFUSE AT THE END OF THE 3RD RUN AT 15^{0}C AND FILLING TIME OF 280 DAYS**



Concentration of orthophosphate was 32% lower than in the influent. This should be theoretically higher due to hydrolysis of organic bound phosphate if no precipitation occurred. The removal of orthophosphate inside the reactor was most likely due to the formation of complexes such as calcium phosphate ($CaOHPO_4$) or magnesium ammonium phosphate ($MgNH_4PO_4.6H_2O$). These complexes could precipitate and settle to the bottom of the reactor.

An attempt to estimate the prevalence of any phosphate precipitate in the ashes from AC2 was done. Except of phosphate also Ca and Mg ions were determined (Table 4.2). The sludge from AC2 from this 3rd run contained higher concentration of Mg than Ca with molar ratio between P:Ca and P:Mg of 1:0.54 and 1:1.1 respectively.

TABLE 4.2 IONS CONCENTRATION IN THE ASH OF AC2 SLUDGE

P_{PO4} mgP.mg ash^{-1}	Ca mg Ca.mg ash^{-1}	Mg mg Mg.mg ash^{-1}
0.00432	0.00302	0.00376

4.1.4 CONVERSION TO BIOGAS

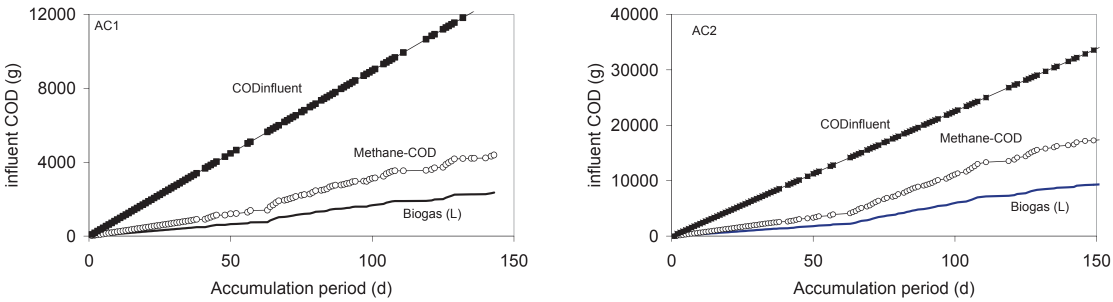
Except of monitoring the quantity of produced biogas, its composition was analysed in the 1st run. The methane fraction was, in the range as reported in literature, 55 – 75% of total biogas volume (Mes et al., 2003) (Table 4.3). Determined fraction of methane was important to calculate the mass balance for the organic matter in the system.

TABLE 4.3 COMPOSITION OF BIOGAS PRODUCED IN AC1 FED WITH BLACK WATER AND KITCHEN REFUSE AND AC2 FED WITH BROWN WATER AND KITCHEN REFUSE AT 20ºC AND 150 DAYS

	O_2	N_2	CH_4	CO_2
		%		
AC1	0.16 ±0.11	0.94 ±0.03	68.44 ±4.58	30.46 ±4.46
AC2	0.20 ±0.06	0.70 ±0.19	65.99 ±1.78	33.12 ±1.62

The biogas production in the accumulation systems was not measured in the first 36 days, due to malfunction of the gas meters. The gas production in this period was calculated based on gas production rate in the period from day 38 and 58 (12.6 and 38 l.d^{-1} from AC1 and AC2 respectively). This simplification does not encounter possible lag phase in gas production. Figure 4.11 shows the cumulative methane production of both reactors expressed as volume and in terms of methane COD compared with total influent COD load. The average biogas production of AC1 and AC2 were respectively 16.4 (1.1 person) and 62 (2.1 person) l.d^{-1}.

FIGURE 4.11 CUMULATIVE BIOGAS PRODUCTION IN AC1 AND AC2 DURING PHASE 1 OF OPERATION EXPRESSED IN (L) AND (G COD-CH_4) COMPARED TO CUMULATIVE INFLUENT TOTAL COD LOADING



From comparison between the total influent loading and biogas produced a methanisation rate can be calculated over the operation time. For AC1, approximately 30% of the influent total COD was converted into methane after 143 days. For AC2 58 % of the influent was converted into methane after 157 days. The cumulative influent COD loading was calculated based on average measured COD and flow.

In the 2nd run the monitoring of the produced biogas was also delayed due to malfunction of the gas meters and started after 31 and 49 days of operation of AC2 and AC1 respectively. Calculations indicate again higher conversion of the wastewater in AC2 (51 %) as compared to that in AC1 (28%).

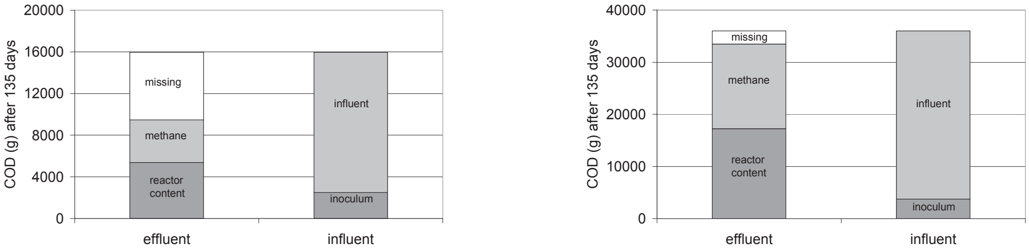
Due to several problems with gas meters and stop in the reactor operation for longer than one month no continuous readings could be obtained during the AC2 operation in the 3rd run. During the pseudo steady state of the reactor (between 250th – 290th day) the average daily biogas production was 25.7±3.1 $l.day^{-1}$ (approximately 1 person). The methanisation rate calculated based on the average total COD loading and biogas production would be 78%. This value carries a high degree of uncertainty because the influent characterisation of this run exposed much lower values than in the previous runs although the same ratio faeces and kitchen refuse was provided (the same influent compositions). Taking influent characterisation from the previous run would give methanisation rate of 47 %, which is closer to the finding of the previous run. During the storage period the biogas production continued attaining an average rate of 7.75±8.21 $l.day^{-1}$ for a period approximately 30 days.

4.1.5 MASS BALANCE

The differences in biodegradability (degree of methanisation) found in the AC1 reactor in the 1st run and in the batch experiments (see 3.6) are also reflected in the mass balance (Figure 4.12). A considerable amount of total influent COD (+inoculum) is not recovered based on analysis of all outcome values (reactor content, biogas). Several reasons may explain this difference. Firstly, the influent characterization based on the limited number of grab samples could have been erroneous (too concentrated, less probable). Secondly, there was a possibility that the amount of monitored methane was too low (whether the gas-meter worked not properly, or there was a gas leakage). In both cases the actual biodegradability would be higher. It is not unlikely to explain the 'gap' by a higher actual biodegradability, since the results of the batch experiments gave biodegradability of around 80%. If the not recovered part would be lost (unmeasured) biogas, methanisation rate as high as 66% could be even attained (inoculum included).

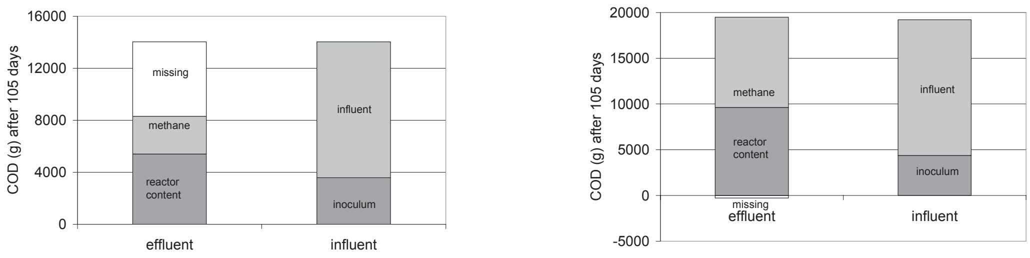
In the beginning of the pilot-plant experiment, the reactors contained oxygen, which was consumed for the degradation of organic matter. The presence of oxygen did not significantly contribute to the total COD depletion, since only 1.8% and 0.7% of the total AC1 and AC2 influent could be converted by oxygen.

FIGURE 4.12 COD BALANCE FOR AC1 (L) AND AC2 (R) REACTOR IN THE 1ST RUN CALCULATED ON DAY 135TH (AC1 $COD_{TOT,IN}$ = 13.3 $G.L^{-1}$; AC2 $COD_{TOT,IN}$ =66 $G.L^{-1}$)



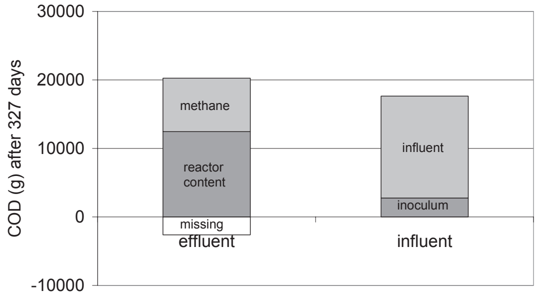
The very similar result of the mass balance for the AC reactors was obtained in the 2nd run (Figure 4.13). Again a large gap was obtained for AC1. If this gap was unmeasured methane, methanisation rate as high as 61% could be attained (inoculum included).

FIGURE 4.13 COD BALANCE FOR AC1 AND AC2 REACTOR IN THE 2ND RUN (AC1 $COD_{TOT,IN}$ = 18,7 G.L^{-1}, AC2 $COD_{TOT,IN}$=53.6 G.L^{-1})



For AC2 in the 3rd run the mass balance over a total accumulation period of 284 days was estimated (Figure 4.14). The calculations showed that 41% of accumulated influent COD_{total} was converted into methane and the rest remained in the reactor. This conversion is lower than obtained in the previous runs using the same reactor but under higher operational temperature. Again several reasons can be listed for this difference such as underestimation of biogas production and/or influent strength due to gas meter malfunction, leakage in the gas collector, etc.

FIGURE 4.14 COD BALANCE FOR AC2 IN THE 3RD RUN ($COD_{TOT,IN}$ = 22.9 G.L^{-1})



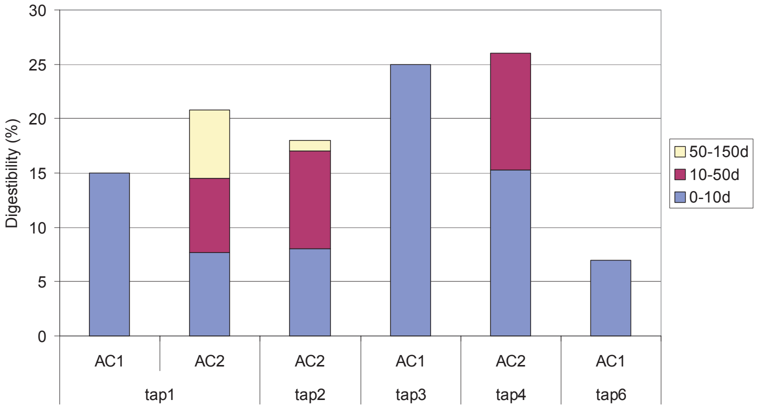
4.1.6 EFFLUENT STABILITY (DIGESTIBILITY)

After the feeding terminated in the 1st run, samples were taken from the reactors' content to determine their digestibility (=remaining biodegradable COD). To get an idea on the differences between stability of sludge and treated wastewater, three samples (in triple) were taken from respectively tap 1,3 and 6 for AC1 and tap 1,2 and 4 for AC2. The digestibility was followed over a long period of 150 days.

In the digestibility experiment for AC1 (for tap 1,3 and 6) all methane was produced in the first 10 days after it ceased. After analysis of the COD_{total}, it turned out that the sample of tap 1 AC1 was less concentrated in terms of TS than during reactor feeding. Samples were taken a week after stopping the feeding of the reactor, and most of sludge was likely concentrated below sampling tap 1.

The methane production measurement in the batches with AC2 content took 150 days. Figure 4.15 shows the digestibility of different samples from different reactors heights estimated for AC1 and AC2 after respectively 10, 50 and 150 days. Based on the duration of the experiment it seems that AC1 wastewater got well stabilized during the accumulation period of 150 days under 20°C (although low methanisation rate does not confirm it, mass balances do, see Figure 4.12). Only the first 10 days of the test, the biogas production was observed. The sludge layer could have had a higher digestibility than measured 15%. The water volume from tap 3 had still quite high digestibility of almost 25%, but probably related to easier biodegradable (soluble) matter that did not get chance to get in contact with sludge. On the other hand the highest layer of stored wastewater had a low digestibility of 7%. It could be that a layer of scum was formed on the top of this volume allowing for better contact with biomass for degradation of biodegradable matter. In AC2 reactor the lowest digestibility of ca. 17% had the middle layer of the reactor. The highest digestibility of 26% was estimated in the top layer of the reactor.

FIGURE 4.15 DIGESTIBILITY-PROFILE OF AC2 FROM THE 1ST RESEARCH RUN AFTER RESPECTIVELY 10, 50 AND 150 DAYS FROM THE MOMENT OF STOPPING THE FEEDING



From this profile one could conclude that the supernatant (tap 2 and 4) contained still the most of non-stabilised, easily biodegradable organic matter. Obviously the sludge layer contained both: easily and slowly digestible (particulate) matter.

The maximum digestibility was followed also after the 3rd run of AC2 (15°C). For tap 1,2 and 3 respectively 8, 13 and 11% digestibility were estimated in 70 days of incubation. The digestibility of the reactor content was lower than in the 1st run. A very long retention time and high inoculum/influent ratio contributed to the fact that the mixture was far stabilised.

4.1.7 METHANOGENIC ACTIVITY

Based on the average biogas production and total volume of sludge the actual, average methanogenic activity was determined. The estimated results from all three runs are depicted in Table 4.4. Because of a low organic loading of the reactor the actual specific methanogenic activity (SMA) were very low.

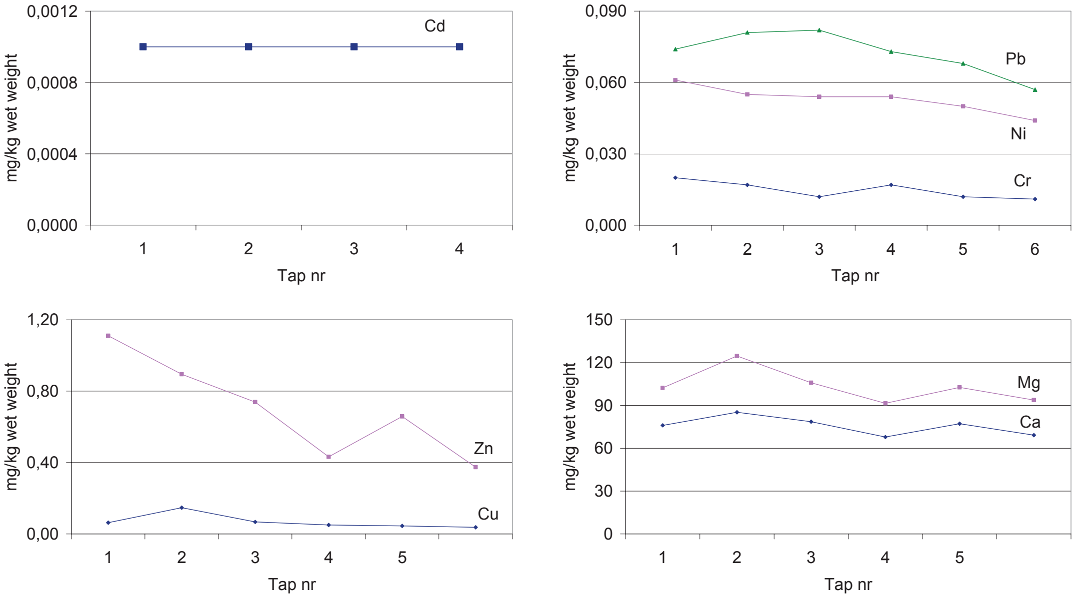
TABLE 4.4 ACTUAL SMA (GCH_4-COD.GVS^{-1}.DAY^{-1}) OF THE AC SLUDGE DURING ALL THREE RUNS

	1st run (20°C)	2nd run (20°C)	3rd run (15°C)
AC1	0.006	0.013	
AC2	0.01	0.027	0.024

4.1.8 HEAVY METALS

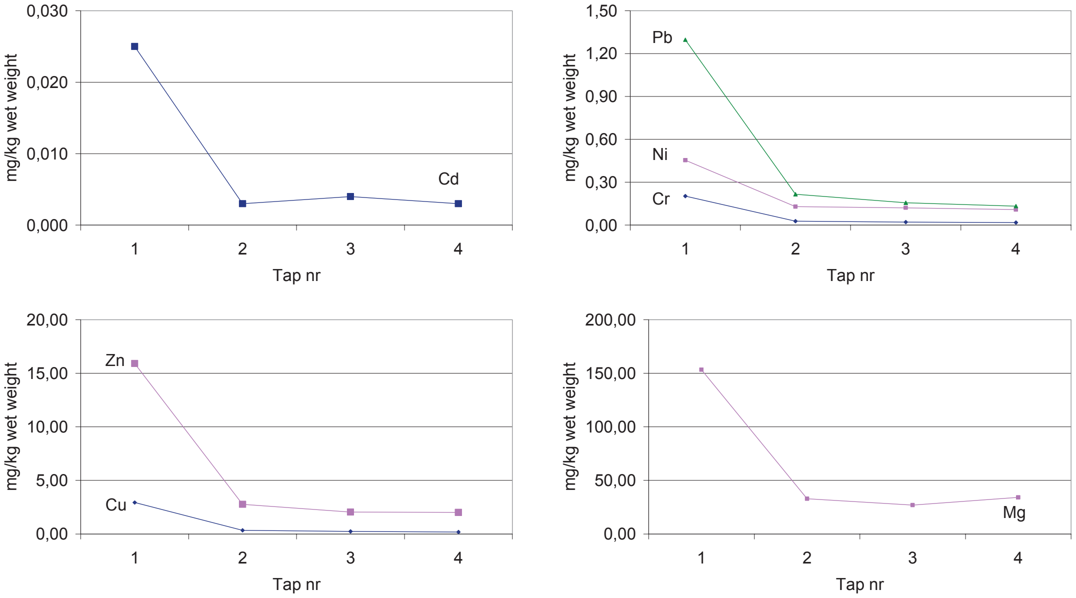
After the 2nd verification run stopped, samples from different heights of AC1 and AC2 were taken to determine the profile of chosen heavy metals and other elements (Figure 4.16 and 4.17). In AC1 these concentrations did not show a clear profile and were quite constant along the reactor height for most cases. Only Pb, Ni and Zn concentration declined clearly from the bottom to the top of the reactor. After the reactor feeding was stopped the sludge accumulated at the bottom started to thicken and finally became a compact layer, however below the first tap. It means that the 1st tap did not represent a real sludge sample.

FIGURE 4.16 **COURSE OF CHOSEN ELEMENTS IN AC1 AFTER STOPPING THE FEEDING IN THE 2ND RU**



In AC2 metals clearly accumulate at the bottom of the reactor (Figure 4.17). The Cr, Ni, and Pb concentrations were still higher at the height of the second tap than in the influent.

FIGURE 4.17 **COURSE OF CHOSEN ELEMENTS IN AC2 AFTER STOPPING THE FEEDING IN THE 2ND RUN**



In table 4.7 the sludge from AC1 and AC2 (2nd and 3rd run) is compared with Dutch standards (BOOM, 1998) for reuse of organic fertilisers in agriculture and with a commercial fertilizer (Palmquist and Jönsson, 2003).

TABLE 4.7 SLUDGE AND EFFLUENT CONCENTRATION FROM AC1 AND AC2 REACTORS REGARDING HEAVY METALS CONTENT AS COMPARED TO DUTCH STANDARDS FOR USE OF ORGANIC FERTILISERS IN THE AGRICULTURE (VROM, 1991) AND COMPOSITION OF COMMERCIAL, MINERAL FERTILISERS (PALMQUIST AND JÖNSSON, 2003)

	Cd	Cr	Cu	Ni	Pb	Zn
Dutch standard, mg.kg^{-1} dry matter	1.25	75	75	30	100	300
AC1 sludge (2nd run)	0.03	0.53	1.66	1.08	0.34	27.58
AC2 sludge (2nd run)	0.57	6.04	66.9	9.99	16.4	372
AC2 sludge (3rd run)	1.03	8.54	123.3	10.6	35.4	545
Commercial fertiliser, mg.kgN^{-1}	0.08	12	2.3	7	0.7	25
AC1 sludge	6.21E-03	0.163	0.513	0.331	0.107	9
AC2 sludge	0.21	2	25	2	7	2
Commercial fertiliser, mg.kgP^{-1}	0.24	37	6.9	22	2	76
AC1 sludge	0.04	1	3	2	0.69	55
AC2 sludge	0.79	7	95	8	27	8

For all studied metals, the concentrations in the sludge from AC1 (after 2nd run) are below the critical values, so from this point of view this sludge could be considered safe for use as agricultural fertilizer. Furthermore, the metal concentrations in AC1 were lower compared to the concentrations of the commercial fertilizer. Latter concentrations are expressed in mg metal/kg N or P. On the other hand the most compact sludge (probably containing the highest fraction of metals) was located below the sample tap 1 and could not be analysed. The definitive evaluation regarding the quality of AC1 sludge can not be thus given. Also one has to remember the purpose of an AC application, summarized as direct application (reuse) of the whole digested mixture and remaining the part of the sludge in the reactor for the following run. In that way both, sludge and supernatant are present in one volume and derivative metals concentration is lower than for the sludge only.

In samples from AC2 (3rd run) the critical concentrations for Cu and Zn were exceeded. These samples contained except of metals provided with the influent in that run also metals from the previous two runs. When comparing with composition of commercial fertilisers, three elements tended to be higher – Cd, Cu and Pb.

One has to remember also that both AC reactors were inoculated with the primary digested sludge from a municipal WWTP containing originally higher metals concentrations than described here types of systems.

4.1.9 FATE OF PATHOGENS

When operation of AC1 and AC2 was stopped in the 2nd run, samples were taken to determine *E.coli* concentration as a function of additional storage. The samples were stored in separate bottles for an additional period of 142 days. *E.Coli* concentrations and reduction in concentration after a storage period of 84, 113, 120 and 142 days are presented in Figure 4.18.

After a storage period of 142 days, the *E.Coli* concentration in AC1 at the height of tap 1 (sludge) had dropped to approximately 400 CFU.100 ml^{-1}, corresponding to a removal of 99,993 %. No *E.coli* bacteria could be detected in the AC1 content from tap 2 and 3 after 142 days of storage. At the 4th and 5th tap, however, the *E.coli* concentrations were approximately 600 and 1100 CFU.100 ml^{-1}, respectively.

In AC2 *E.coli* concentration at the height of tap 1 (sludge) had dropped after 142 days of storage to approximately 120 000 CFU.100 ml^{-1}, corresponding to a 99,86 % removal. At tap 2 and in the composite sample no *E.coli* bacteria were detectable after a storage period of 142 days, but in the sludge sample (tap 1) the concentration had increased. The results from the *E.coli* and *F.Coliform* analysis in AC2 during the 3rd run (15ºC, half way accumulation period), presented in Table 4.8 and Figure 4.19 show clearly that *E.coli* bacteria are concentrated in the sludge. The attachment of the bacteria to (sludge) particles is confirmed by a significant difference between the unfiltered and filtered (4.4 μm) sample.

FIGURE 4.18 EFFECT OF ADDITIONAL STORAGE ON E.COLI CONCENTRATION IN THE DIGESTED MEDIUM OF AC1 (LEFT) AND AC2 (RIGHT) IN THE PERIOD AFTER THE 2ND RUN STOP

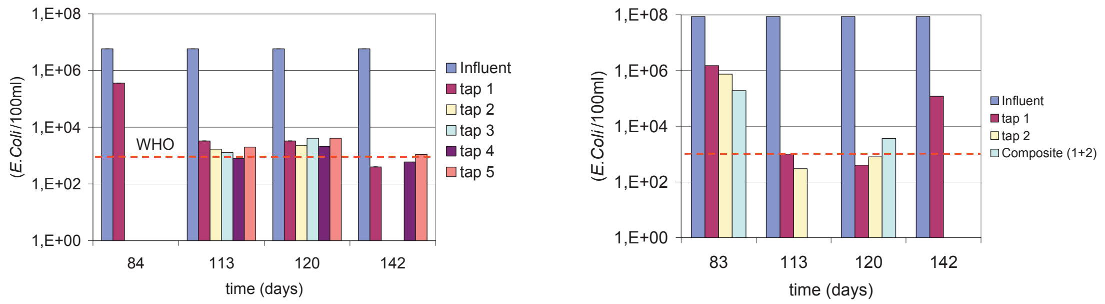
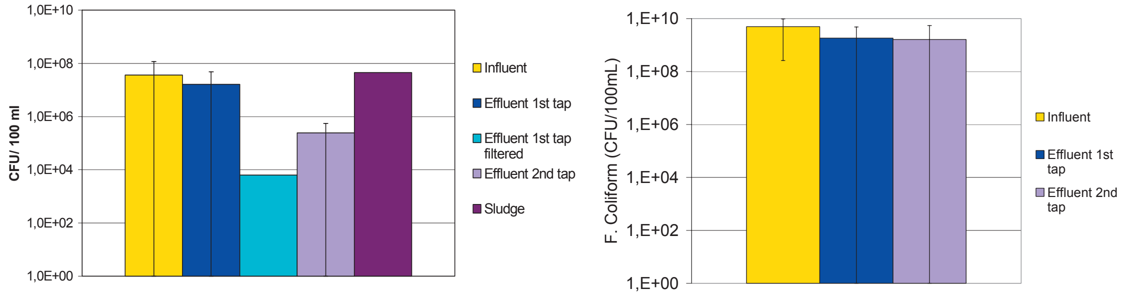


TABLE 4.8 FATE OF PATHOGENS IN THE AC2 (3RD RUN) IN CFU.100ML-1 (N=NUMBER OF SIGNIFICANT PLATES)

	influent	**1st tap, raw sample**	**1st tap, filtered sample**	**2nd tap**	**sludge**
E.coli	3.63E+07 ± 8.03E+07 (N=5)	1.62E+07 ±3.19E+07 (N=4)	6.30E+03 (N=1)	2.43E+05 3.09E+05 (N=3)	4.50E+07 (N=1)
F. coliform	4.97E+09 ± 4.71E+09 (N=5)	1.83E+09 ± 2.99E+09 (N=6)		1.64E+09 ± 3.80E+E09 (N=8)	

FIGURE 4.19 CONCENTRATIONS OF E.COLI AND F.COLIFORM IN THE AC2 DURING THE 3RD RUN (HALF WAY ACCUMULATION PERIOD)



The standard for unrestricted irrigation (WHO, 1989) amounts to less than 1000 *E.coli* .100 ml-1. The effluent *E.coli* concentrations are above this concentration value in AC2 during its operation. Even after filtration the *E.coli* concentrations are still too high. However, after a period of 142 days of unfed storage the AC1 content and the sludge meet this quality guideline. For AC2 this guideline is already met after a storage period of 113 days. A risk of a bacterial re-growth in the sludge however exists. Concerning reuse of digested medium and risk related to pathogens content, a separation between liquid and solid phase can be considered when reactor is being emptied after additional storage time. Liquid carrying less risk and complying with the standards can be directly reused while solid phase (being a much smaller volume) can be reused after hygenisation step.

The WHO standard for *F.Coliform* also amounts to less than 1000 *F.Coliform*.100 ml^{-1} (Asano and Levin, 1998). For AC2 the effluent *F.Coliform* concentrations exceed this value about a million times. However, the concentrations found in this study are not significant, and no information on the effect of storage on *F.Coliform* concentrations was obtained.

4.2 REACTOR DESIGN

The characteristic feature of AC system is that it is continuously fed with waste(water), without production of daily effluent. The effluent is normally obtained once during desludging (emptying) when the reactor is full, storage time is reached and treated mixture is needed as fertiliser. However, a portion of the sludge is left to act as an inoculum for the next cycle. Based on this definition the designed volume of the reactor (m^3) depends on:

- daily amount of provided wastewater (Q, $m^3.day^{-1}$);
- demanded accumulation period (t_{AC}, d)
- volume of inoculum sludge (V_{inoc}, m^3)

Volume of the reactor can be calculated thus using the following equation:

$$V_{AC}=V_{inoc} + Q\cdot t_{AC} \qquad (4.1)$$

When the AC system is fed with human excreta from a number of people, the volume of the reactor can be calculated as follow:

$$V_{AC}=V_{inoc} + n \cdot[(V_u + V_{wu} +V_f\cdot V_{wf}+V_k+V_{wk}) \cdot t_{Ac}] \qquad (4.2)$$

where:

n – number of people feeding the reactor

V – volume of waste provided, subscripts u,f,k stand for urine, faeces, kitchen refuse respectively

V_w – volume of water used to transport the considered waste to the reactor; additional subscripts u,f,k take into account different water use to flush different waste (for instance no-mix toilets use different quantities of water to collect urine and faeces; if kitchen refuse is co-digested with black water or brown water it will be possibly brought to the reactor using another system than toilet with another water consumption).

> **Example:** Daily volume of produced urine is 1,25 $l.person^{-1}$. One person visits 5 times per a toilet to urinate. The volume of faeces is 0,15 l. Statistically one person produces faeces once per day. If urine and faeces are to be treated together and each portion is collected by a vacuum toilet using 1l per flush, 7,4 l black water is produced daily by one person. One individual produces 0,2 kg wet kitchen waste per day. If kitchen waste is co-digested with black water and a kitchen waste grinder uses 0.5 l of water to transport this waste to the treatment, additional 0,7 l of wastewater will be brought to the reactor, giving 8,1 l concentrated wastewater per person. If required storage time is 6 months (180 days) the required volume of an AC reactor not including inoculum will be **1.4 $m^3.person^{-1}$.** Following the same pathway when brown water (faeces collected with vacuum) and kitchen refuse are to be co-digested (0.15+1+0.2+0.5) for the same period of 180 days the volume of the AC will be significantly smaller – **0.33 $m^3.person^{-1}$**. A settlement of 30 houses or approximately 100 people would need reactor of 140 and 33 m^3 excluding inoculum to treat black water with kitchen refuse and brown water with kitchen refuse respectively.

The mass balance over the reactor is determined by the fact that, the waste entering the system accumulates and methane is the only COD-flow leaving the reactor. The following relationship depicts the situation:

$$COD_{in(t)} = COD_{acc(t=x)} + \Delta COD_{methane(t=x)} \qquad (4.3)$$

Where:

$COD_{in(t)}$ = total COD input in reactor (g COD) over considered period (from t=0 to t=x)

$COD_{acc(t)}$ = total COD present in the reactor at time t=x (g COD)

$\Delta COD_{methane(t)}$ = cumulative methane production over considered period (from t=0 to t=x, g COD)

For calculation of the accumulation and conversion of the suspended COD fractions it is assumed that the accumulation system consists of several small batch reactors + one large inoculum batch reactor. The accumulated suspended COD fractions for the whole reactor can in principle then be calculated by the sum of the inoculum batch and the separate batch reactors from the daily feed. The conversion of the suspended, biodegradable COD follows the first order hydrolysis kinetics. Concentration of the particulate COD in the reactor at time = t can be calculated using equation 4.4.

$$COD_{part}(t)=(V_{inoc}\cdot f_B\cdot COD_{part,inoc}\cdot e^{-kH\cdot t}+\Sigma Q\cdot f_B\cdot COD_{part,inf}\cdot e^{-kH\cdot t})/V(t),\ (mgCOD.l^{-1}) \qquad (4.4)$$

Biodegradability (f_B) and first order hydrolysis constant (k_H) of the particulate substrate can be obtained from the batch test.

Simplification made here assumes that the amount of available biodegradable soluble substrate to be converted in methane is a sum of converted COD_{part} (eq. 4.4) and biodegradable COD_{sol} (hydrolysis is assumed to be rate limiting). The amount of methane that can be produced can be calculated using the following formule:

$$CH_4\ (m^3) = f_{CH4\text{-}COD}\cdot\Sigma COD_{VFA} \qquad (4.5)$$

where:

$f_{CH4\text{-}COD}$ = conversion factor to transform volume CH_4 at working temperature and pressure conditions to mass COD-CH_4.

ΣCOD_{VFA} – sum VFA from the influent and produced during the accumulation period from biodegradable COD_{part} and biodegradable soluble COD minus VFA from the influent.

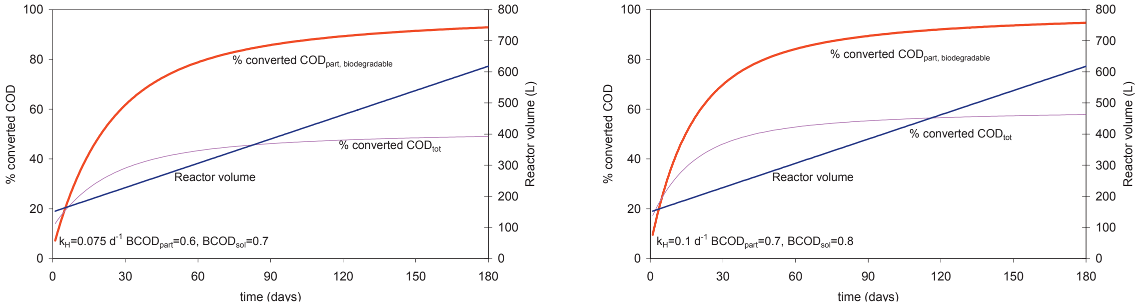
Temperature in this approach has influence of k_H value and conversion factor for methane production from biodegradable COD.

In Figure 4.20 the results of two example simulations are given for brown water. This influent was chosen because it seems to be the most appropriate medium to be treated in accumulation systems due to several reasons (small volumes, high concentrations). Except for the increase in reactor volume, percentage of converted particulate, biodegradable COD and percentage of total COD converted are shown in the figures. Data used for these two example scenarios can be found in Table 4.9.

TABLE 4.9 **DATA USED TO CALCULATE AC REACTOR FED WITH BROWN WATER AND KITCHEN REFUSE AS PRODUCED BY ONE INDIVIDUAL. REACTOR IS OPERATED AT 20°C. THE DIFFERENCES BETWEEN TWO SITUATIONS ARE DIFFERENT HYDROLYSIS CONSTANTS AND BIODEGRADABILITY OF PARTICULATE AND SOLUBLE COD**

Parameter	Unit	Scenario 1	Scenario 2
Input			
Volume inoculum	l	150	150
Influent flow	$l.d^{-1}$	2.6	2.6
Influent COD_{total}	$g.l^{-1}$	53.6	53.6
Influent COD_{sol}	$g.l^{-1}$	10.1	10.1
Biodegadability COD_{part}		0.6	0.7
Biodegadability COD_{sol}		0.7	0.8
Hydrolysis constant, k_H (T=20°C)	d^{-1}	0.075	0.1
Simulation period	D	180	180
Results			
Reactor volume	l	670	670
% inoculum in relation to end reactor volume		22	22
% converted biodegradable COD_{part}		93.6	95.2
% converted total COD		49.35	58.1
Biogas produced in considered accumulation period	m^3	4.4	5.2

FIGURE 4.20 **CALCULATION EXAMPLE OF THE AC REACTOR TREATING A MIXTURE OF BROWN WATER AND KITCHEN REFUSE (Q = 2.6 $L.D^{-1}$) WITH A WASTEWATER CHARACTERISTICS AS GIVEN IN TABLE 3.4. TWO SCENARIO'S WERE SIMULATED. THE DIFFERENT INPUT PARAMETERS WERE HYDROLYSIS CONSTANT AND BIODEGRADABILITY OF SOLUBLE AND PARTICULATE FRACTION OF THE INFLUENT COD_{TOT}.**



It is generally agreed that during the anaerobic conversion of concentrated domestic wastewater streams the hydrolysis of the biodegradable particulate COD is the rate-limiting process. This is however only the case if the reactor has been started with inoculum with sufficient methanogenic activity to convert the VFA at any time. The methanogenic activity provided with inoculum sludge must be sufficient to convert the sum of the produced VFA from biodegradable COD_{part} and the VFA supplied with the influent. The required amount of inoculum (V_{inoc}, l) can be estimated from equation 4.6.

$$V_{inoc} = (\text{daily VFA produced from } f_B \cdot COD_{part,inf} + \text{daily VFA influent})/(VS_{inoc} \cdot SMA_{inoc}) \qquad (4.6)$$

where:

VS_{inoc} = volatile suspended solids content of the inoculum ($g.l^{-1}$)
SMA_{inoc} = specific methanogenic activity of the inoculum (at applied reactor temperature) (g COD.g VS^{-1}.day-1)

Example: The concentration of particulate COD in a mixture of brown water and kitchen refuse to be provided to an anaerobic accumulation system operating at 20ºC is 43.4 gCOD.l^{-1} ($COD_{part,inf}$). The biodegradability of particulate COD is assumed to be 0.7 ($f_{B,part}$). The concentration of VFA in the influent is 1 g.l^{-1}. The concentration of COD_{sol} is 5.3 gCOD.l^{-1} with the biodegradability of 0.7 (($\mathbf{f}_{B,sol}$). The wastewater flow to the reactor amounts at 2.6 l.d^{-1}. Since the wastewater has to be accumulated in a reactor for a period of 180 days the reactor volume needed to treat and store the influent is 468 l. The potential amount of VFA daily produced from COD_{part} can be calculated as: VFA_{COD} (g.d^{-1}) = $f_{B,part}$ COD_{part} $e^{-kH(t=1d)}$+ $VFA_{COD,in}$+ $f_{B,sol}$ (COD_{sol}-VFA). The latter equation is proposed under assumption that hydrolysis is a process the rate limiting step and that conversion of hydrolysate and biodegradable COD_{sol} will lead to the production of VFA. With the hydrolysis constant for 20ºC as high as 0.1 d^{-1}, 94 gCOD.l^{-1} of VFA should be daily produced. The inoculum to start up the reactor is characterised by a specific methanogenic activity of 0.1 gCOD$_{CH4}$.gVS^{-1}.d^{-1} and has a VS concentration of 20 g.l^{-1}. Implementation of equation 4.6 results in a volume of inoculum of 47 l what constitutes approximately 9% of the total reactor volume.

4.3 DISCUSSION

4.3.1 ORGANIC FRACTIONS REMOVAL

Due to the specific character of an accumulation system the efficiency for a given COD fraction is of limited importance. More important parameters are digestion rate (methanisation) of total influent COD, gas yield as well as stability of the reactor content. The maximum estimated methanisation within different accumulation runs was 58% for the AC fed with a very concentrated mixture of brown water and kitchen refuse.

The content of the AC reactor is strongly stratified if no mixing device is installed but during emptying is mixing of both phases difficult to avoid. When however, for comparison reasons, one wants to estimate the removal of different COD fractions, the liquid phase can be considered as effluent and traditional removal efficiency can be assessed.

The conversion of organics into methane can be considered as satisfactory in the accumulation reactors, when comparing with maximum biodegradability. Due to a very long retention time the methanogenesis was not the limiting step and hardly any VFA were measured in the reactor content after a certain time of accumulation. The hydrolysis process was the limiting factor regarding the total COD conversion. The combination of a long retention time, the presence of adapted inoculum can compensate for a lower operational temperature.

4.3.2 BIOGAS AND POTENTIAL ENERGY PRODUCTION

One of the drawbacks of anaerobic digestion in practice, is that the produced methane is often not utilised for energy production. To illustrate the potentials of this energy source to operate the system independent of external energy calculations were made based on the performance of AC2, operated at 20ºC and a filling time of 150 days.

ENERGY NEED FOR PUMPING

- The vacuum pump operated approximately 10 sec per flush, with 600 flushings, which made a total operational time of 6000 sec in 150 days. The power of the vacuum pump is 0.55 kW. Therefore the vacuum pump needs $3.3 * 10^6$ Joule (0.92 kWh).
- The shredder/pump operates approximately 20 seconds per run. It pumps 10 l of wastewater per run. To fill the AC reactor with 600 l, it makes 60 runs á 20 seconds giving a total operation time of 1200 seconds. Since the power of this pump is 1.1 kW, the shredder requires energy of $1.32 * 10^6$ Joule (0.37 kWh) in 150 days.

ENERGY NEED FOR HEATING THE WASTEWATER

- According the law of thermodynamics, a mix of 1.0 l of flushing water and 0.2 kg of physiological waste of respectively 15 °C and 37 °C has a temperature of 20 °C.
- To keep the wastewater at 20 °C the reactor has to be isolated. However isolation is not sufficient to prevent heat losses. Assume use of an isolation material of 5 cm (dx) with a thermal conductivity coefficient (λ) of 0.04 $W.m^{-1}.K^{-1}$ (e.g. mineral wool from glass fibres), environmental temperature of 15 °C (dT=20-15°C) and an effective reactor surface area (A) of 3 m^2. The need for heat can be calculated according to the Fourier law of heat conduction, $\Phi = -\lambda * A * dT/dx$.

  $\Phi = - 0.04 (K) * 3 * 5/0.05 = 12$ W, 43.2 kWh in 150 days (or $1.56*10^8$). That is exclusive the heat loss during transport and in the storage/equalisation tank.

ENERGY PRODUCTION

- Pure methane has calorific value of 9100 $kcal.m^{-3}$ at 15.5°C and pressure of 1 atmosphere ($35.8*10^6$ $J.m^{-3}$, at 273 K and $p=p_0$).

  Over 150 days AC2 produced 10.53 m^3 methane, at 293 K. This is 9.8 m^3 at 273 K.

  The total amount of energy, which is produced over the operation time, is $350 * 10^6$ Joule (97,3 kWh).

Above illustrates that anaerobic digestion of concentrated black water in combination with kitchen refuse gives a surplus energy of approximately $160 * 10^6$ Joule (52.1 kWh) over 150 days, which can be potentially (re)used in the household. The conversion of the methane in energy (electricity, heat) will determine the efficiency of the use of this reusable gas.

4.3.3 NUTRIENTS

The supernatant of the AC reactors contains relatively low biodegradable organic compounds but is rich in soluble nitrogen and phosphate. The sludge, next to latter dissolved compounds, is rich in organic carbonaceous particulate material (far stabilised), particulate nitrogen and precipitated phosphorus.

The fact that the nutrients are mainly present in a soluble form, make reuse of the effluent in agriculture an attractive option, when transport distances are limited. When distances to agricultural fields are long, further processing is necessary to form precipitated complexes, such as calcium phosphate ($CaOHPO_4$) or struvite ($MgNH_4PO_4.6H_2O$) using appropriate additives to enhance the precipitation (Ca, Mg, P). The precipitated phosphate complexes are valuable compounds especially when considering the problem of depletion of natural sources of phosphate (Steen, 1998 and Johnston, 2003). However it requires a comprehensive management of the effluent of the reactors to meet the balance of what is added and required to minimise the environmental effect.

The addition of kitchen refuse to the anaerobic reactors fed with concentrated black water seems to enhance the formation of precipitation complexes. The phenomenon could be especially observed in the reactor fed with concentrated brown water and kitchen refuse and to less extent in that fed with black water. The soluble phosphate concentration in the AC systems is lower compared to the influent wastewater, while the total phosphate concentration increase significantly at the bottom of the reactors. Followed fate of nutrients in the batch tests for determination of hydrolysis constant using different types of influent proved that precipitation of complexes (P but also NH_4) occurs in each reactor to lesser or higher extent.

4.3.4 CONSTRAINTS IN AGRICULTURAL REUSE

The salinity of the AC reactor content (expressed in conductivity of the sample) was 1.46 $dS.m^{-1}$. This value is in the usual range for not effecting soil or crops. Accumulation of salt in the soil should be prevented by proper management regarding application of treated concentrated domestic wastewater residues and crop selection to minimise environmental effects.

For direct application in agricultural field, the fate of pathogenic organisms and heavy metals in the sludge should be taken into account. Brief studies on the fate of pathogens and heavy metals in digesters treating concentrated domestic wastewater, showed that E.coli can be far reduced in accumulation reactors especially when some additional post-digestion, un-fed (storage) period is secured. The sludge contained, as expected, the highest concentration of heavy metals when comparing with supernatant, not always in the acceptable range for use in agriculture.

The way of application of digested effluent need to be chosen in such away that possible contact between the worker and the effluent is minimal, such as sub-surface irrigation and slurry digestion. The selection of non-edible crop and crops that require cooking before consumption will also minimize the health impact of reused digested effluent.

5

UASB SEPTIC TANK UNDER 15 AND 25^{0}C (UASB$_{ST}$ 1 AND 2)

5.1 RESULTS

5.1.1 PROCESS CONDITIONS AND OPERATIONAL PROBLEMS

The performance of the UASB septic tanks discussed in this report is mainly based on the second half of the year of their operation. This period is here considered as ‘adapted’ period. As a result of the increasing sludge volume in time UASB septic tanks are by definition do never operate in ‘steady state’. During that time the UASB$_{ST}$ 1 and 2 received the assumed amount of faeces and urine (black water) equivalent to the production by 1 person weekly (approximately 7 portion of faeces and 35 portion of urine, all flushed with around 1 l of water).

The average daily flow was calculated from the registered contributions of given waste (faeces or urine) multiplied by the sum of their volumes and flush water volumes. The sum was divided by 7 days. The hydraulic retention time (HRT) was calculated based as the average daily flow divided by the reactor volume. The organic loading rate (OLR) was estimated based on the average calculated daily flow, the average measured total COD influent and the total volume of the reactor.

During the operational period several technical problems occurred. Starting at 267th day malfunction of the flush button of the UASB$_{ST}$ 1 toilet caused higher water consumption (1.6 instead of 1 l), hence diluting the influent. The estimated influent wastewater flow in that period was 10.3 $l.day^{-1}$ instead of 7.0 $l.day^{-1}$. The change in flush water consumption decreased the HRT of the UASB$_{ST}$ 1 from 29 to 19 days. The estimated OLR was 0.33 kg $COD.m^{-3}$.

UASB$_{ST}$ 1 faced a problem with scum layer formation inside the gas, solid and liquid (GSL) separator. Because of the scum layer formation inside the GSL separator, gas production could not be measured. The scum was quite compact and difficult to destroy or remove because the GSL was difficult to assess. The TS of scum was 84 $g.l^{-1}$ with a VS content of 86%. A pump was used to remove the scum but within one or two days it was formed again.
On the 297th day of operation, the pressure release vessel of the UASB$_{ST}$ 1 collapsed. The operation was stopped since the reparation required the UASB$_{ST}$ 1 to be dismantled completely. The reactor’s content was stored and brought back to the reactor after the reparation was finished. The vessel was fixed within one day and the reactor remained kept unfed for few days to allow the reactor content to stabilise. This reparation enabled to remove the deposition in the GSL separator. The scum was also returned to the reactor together with the rest of the reactor content. No sign of scum formation was observed after the reparation until the end of this experimental period.

No technical problems occurred during the operational period of the $UASB_{ST}$ 2. However, there was also scum formed for some time on the top layer of the reactors (257th – 284th operational day). The scum was quite thick and compact with a TS content of 90 $g.l^{-1}$ (83% VS), quite similar to the characteristics of scum in $UASB_{ST}$ 1. However after several manual removals and returning the scum layer into the reactor (through the vacuum toilet), it did not appear anymore.

For a short period at the end of research period (from 309th day) the $UASB_{ST}$ 2 received also kitchen refuse (K) in an amount equivalent to the production by 1 person. In the beginning the kitchen refuse was added gradually to avoid/minimise shock loading to the reactor. The first two weeks the reactor received only 3 portions (=600g) and afterwards it was increased to 7 portions (=1400 g). The influent wastewater flow before kitchen refuse addition was 6.8 $l.day^{-1}$ and after 7.4 $l.day^{-1}$. The reactor operated with HRT of 29 days. After kitchen refuse addition the HRT decreased to 27 days. The OLR of the $UASB_{ST}$ 2 was 0.42 kg $COD.m^{-3}.d^{-1}$ and after kitchen refuse addition it doubled to 0.85 kg $COD.m^{-3}.d^{-1}$.

TABLE 5.1 **OPERATIONAL CHARACTERISTICS OF $UASB_{ST}$ 1 AND $UASB_{ST}$ 2**

Parameter (average)	unit	$UASB_{ST}$ 1	$UASB_{ST}$ 2 before K addition	$UASB_{ST}$ 2 after K addition
Feed loading	N° individuals	1	1	1
Flow, Q	$l.d^1$	7 (10.3)*	6.8	7.4
Flush water used	$l.flush^{-1}$	1 (1.6)*	< 1	< 1
HRT	d	29 (19)*	29	27
OLR	$kgCOD.m^{-3}.d^{-1}$	0.33	0.42	0.85
Nuisances caused by scum formation		Compact layer occurred and remained few times; after system got adapted it disappeared definitively	Occurred but to less extent; disappeared and did not return through the last few months of operation	Not observed but too short experimental period

* measured after flush water consumption increased

5.1.2 EFFLUENT CHARACTERISTICS AT OPERATION OF UASB SEPTIC TANKS WITH BLACK WATER

$UASB_{ST}$ 1 (15⁰C)

The results of the effluent COD fractionation (Figure 5.1) clearly showed that for the majority of experimental period the $COD_{soluble}$ was the main contributor to the effluent COD_{total}.

FIGURE 5.1 **COD FRACTIONS OF $UASB_{ST}$ 1 EFFLUENT DURING EXPERIMENTAL PERIOD**

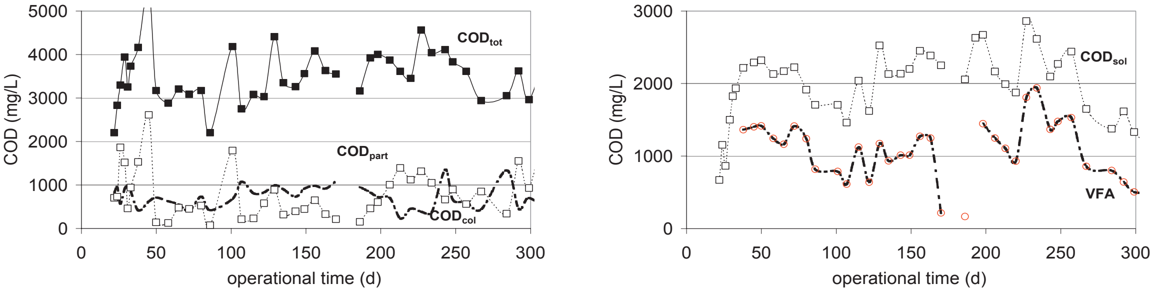


Table 5.2 presents the average effluent quality and removal efficiency of COD fractions. The results show that the UASB septic tank is very efficient in removing particulate COD even when operating under low temperature conditions. Until day 299 the average effluent COD fractions for the $UASB_{ST}$ 1 were approximately 3.7, 1.0, 2.1 and 0.64 $g.l^{-1}$ for total, particulate, soluble and colloidal COD respectively. This corresponded with a COD_{total} removal efficiency of 61% while the COD_{part} removal efficiency was 88%. Obtained removal efficiencies (Table 5.2) were quite comparable to the results obtained in other studies using a UASB septic tank. Bogte et al., (1993) and Luostarinen et al., (2004) using an UASB septic tank treating black water collected with a conventional toilet (COD_{total} 1.7 – 2.7 $g.l^{-1}$) at a temperature of 14 – 17°C, obtained 60 – 69% and 71 – 77% regarding total and COD_{part} removal efficiency respectively.

The main contributor to the effluent total COD was COD_{sol}. The COD_{sol} originated from the influent and from the hydrolysis of COD_{part}. Through the majority of the operational period, the effluent COD soluble was higher than in the influent indicating that acidogenesis, acetogenesis or methanogenesis were the rate limiting step. The lower operational temperatures (15°C) affects the methanogenic activity of the sludge. The average effluent VFA concentration was - 1.25 $g.l^{-1}$, with acetate as prevailing fraction (above 70%).

TABLE 5.2 AVERAGE EFFLUENT QUALITY IN TERMS OF DIFFERENT COD FRACTIONS (IN MG $COD.L^{-1}$) AND REMOVAL EFFICIENCY OBTAINED BY THE $UASB_{ST}$ 1

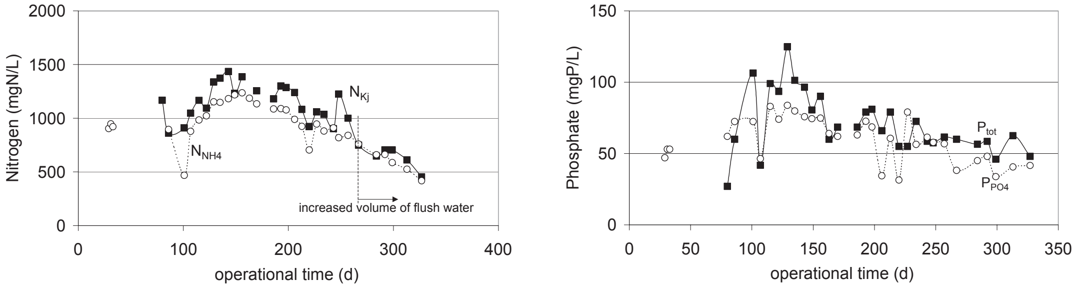
COD fraction	effluent	% removal efficiency
COD_{total}	3699 ± 464	61
COD_{part}	976 ± 366	88
COD_{sol}	2086 ± 449	
VFA	1245±406	
COD_{col}	636 ± 375	

Starting from day 267th the effluent COD_{sol} and VFA concentrations started to decrease. The average VFA concentration in the effluent at this period was 0.4 $g.l^{-1}$ (Figure 5.1). This was not impaired by higher biogas production and was rather caused by influent dilution (mentioned problems with flush button).

The pH of the effluent was stable with an average value of 7.77±0.15. This pH was higher than the optimum pH for an anaerobic digestion process being in a range of 6.5 to 7.5 (Lettinga et al., 1993). As mentioned in chapter 2, anaerobic digestion can be exposed to pH related toxicity (free ammonia and VFA). However, calculation on the concentration of free NH_3 (26 $mg.l^{-1}$) and unionised VFA (1.03 $mgCOD.l^{-1}$) showed that these concentrations were below the inhibition range.

The average effluent N_{Kj} and N_{NH4} of $UASB_{ST}$ 1 were 960±198 $mg.l^{-1}$ and 826±116 $mg.l^{-1}$ respectively. The average values of the effluent N_{Kj} were slightly lower than from the influent while the average values for N_{NH4} were higher due to hydrolysis of particulate organics converting the organically bound nitrogen (urea) into ammonia. 88% of the nitrogen in the effluent was in the form of NH_4^+-N while 83% of total P was in the form of orthophosphate. The average concentrations were 62±8 $mg.l^{-1}$ and 52±14 $mg.l^{-1}$ for P_{total} and P_{PO4} respectively. Figure 5.2 presents the course of nutrients in the effluent of the $UASB_{ST}$ 1.

FIGURE 5.2 NUTRIENTS COURSE IN THE UASB$_{ST}$ 1 EFFLUENT MEASURED DURING EXPERIMENTAL PERIOD. FROM APPROXIMATELY 270TH DAY THERE WAS DECREASE IN CONCENTRATIONS DUE TO INFLUENT DILUTION



UASB$_{ST}$ 2 (25^{0}C)

The average effluent COD fractions from the UASB$_{ST}$ 2 were 2.7, 0.6, 1.4 and 0.8 g.l^{-1} for total, particulate, soluble and colloidal COD respectively (Figure 5.3). The COD$_{total}$ and COD$_{part}$ removal efficiency were 78% and 94% respectively. In the last period of operation kitchen refuse was added to the influent for a short period. The average effluent COD fractions after this 'amendment' increased to 4.16, 0.98, 1.72 and 1.0 g.l^{-1} for total, particulate, soluble and colloidal COD respectively. However, the total and particulate COD removal efficiency remained high, 82% and 94% respectively. Table 5.3 presents the average effluent quality and efficiency of the reactor regarding COD fractions based on the measured influent characteristics before kitchen refuse addition and estimated influent characteristics after addition. The efficiency obtained in this study was similar as compared to the results obtained in other studies, using a UASB septic tank. Lettinga et al., (1993), obtained more than 90% COD$_{tot}$ removal efficiency using an UASB septic tank treating black water (5.5 g COD.l^{-1}) at tropical conditions at HRT of 15 days.

FIGURE 5.3 COD FRACTIONS IN THE EFFLUENT OF UASB$_{ST}$ 2 DURING EXPERIMENTAL PERIOD. LAST POINTS REFER TO THE PERIOD WHEN KITCHEN REFUSE WAS ADDED TO THE INFLUENT

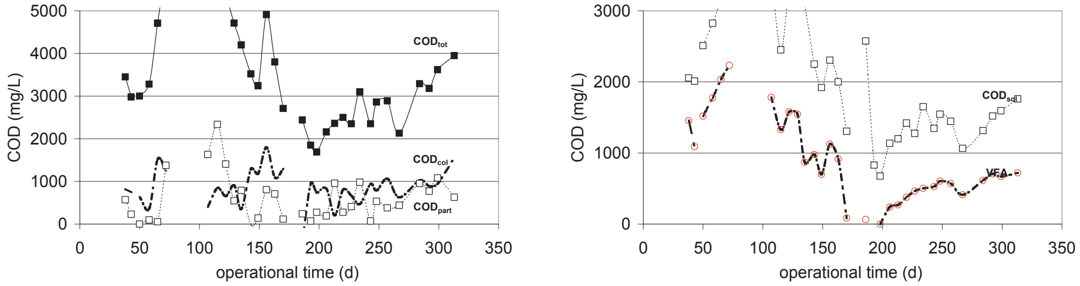


TABLE 5.3 AVERAGE EFFLUENT QUALITY REGARDING DIFFERENT COD FRACTIONS (MG.L^{-1}) AND THEIR REMOVAL EFFICIENCY (%) IN THE UASB$_{ST}$ 2

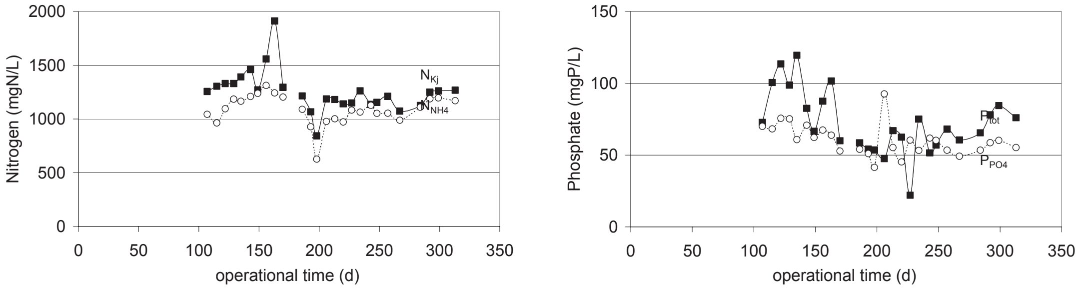
COD fraction	before kitchen refuse addition effluent quality	removal efficiency	after kitchen refuse addition effluent quality	removal efficiency
COD$_{tot}$	2733 ± 491	78	4160 ± 297	82
COD$_{part}$	586 ± 346	94	980 ± 495	94
COD$_{sol}$	1376 ± 186		1718 ± 60	
VFA	529±161		708±30	
COD$_{col}$	771 ± 247		1463 ± 138	

Similar to the $UASB_{ST}$ 1, the soluble and the colloidal COD were the main contributors to the total COD in the effluent of the $UASB_{ST}$ 2. The COD_{part} in the effluent attained relatively low values. The $UASB_{ST}$ 2 showed a better and more stable performance compared to the $UASB_{ST}$ 1 regarding COD removal efficiency. Biological processes are temperature dependent and tend to proceed faster at higher temperatures. This counts to all anaerobic digestion steps: hydrolysis, acidogenesis, acetogenesis and methanogenesis. This can be seen in a higher reduction of particulate and soluble organics. Lower concentration of VFA in the effluent in combination with higher total COD removal indicated higher methanogenic potential of the reactor. The high concentration of COD_{col} was probably due to incomplete hydrolysis of part of the COD_{part} and partly from the unconverted influent COD_{col}.

After kitchen refuse addition into the $UASB_{ST}$ 2 the COD_{total} removal efficiency increased slightly. The average VFA concentration in the effluent increased however from 0.5 to 0.7 g.l^{-1} (Figure 5.3) indicating mainly insufficient methanogenic potential to utilize high VFA (acetate prevailing) concentrations from the influent and produced (short experimental period with kitchen refuse addition).
The pH of the effluent from $UASB_{ST}$ 2 was stable but rather high with an average value of 8.24±0.15 before kitchen refuse addition and slightly lower after kitchen refuse addition - 8.16±0.11 The pH of $UASB_{ST}$ 2 was higher than of the effluent from $UASB_{ST}$ 1, probably because the influent entering $UASB_{ST}$ 2 was less diluted. Higher pH and temperature conditions could expose the process in $UASB_{ST}$ 2 to pH related toxicity (free NH_3). However, similarly to the first UASB septic tank, the calculation on the concentration of free NH_3 and unionised VFA showed that the concentrations were not in the inhibition range (108 mg N.l^{-1} and 0.45 mgCOD.l^{-1} respectively).

The average effluent N_{Kj} and N_{NH4} concentrations before kitchen refuse addition were 1178±59 and 1068±77 mgN.l^{-1} respectively, while after kitchen refuse addition these concentration increased to 1289±32 and 1125±64 mgN.l^{-1} respectively. The average effluent N_{Kj} value was slightly lower than in the influent while the average N_{NH4} concentration was higher due to hydrolysis of urea and organically bound nitrogen in the reactor. The same occurred with phosphorus compounds. The average P_{total} and P_{PO4} in the effluent were 62±16 and 59±12 mgP.l^{-1} respectively. After kitchen refuse addition the effluent P_{total} increased slightly to 73±5 mgP.l^{-1} while the orthophosphate, on the other hand, decreased to 54±2 mgP.l^{-1}. The decreased amount of soluble orthophosphate after kitchen refuse addition would confirm the observed phenomenon of phosphate precipitation, intensified with presence of kitchen refuse. Figure 5.4 presents the course of nutrients in the effluent of the $UASB_{ST}$ 2.
In the $UASB_{ST}$ 2 90% of total N was in the form of ammonium indicating that hydrolysis of protein was higher when comparing to the $UASB_{ST}$ 1. The soluble phosphate fraction of 76% was, on the other hand, lower than in $UASB_{ST}$ 1, indicating more precipitation. The higher pH inside the $UASB_{ST}$ 2 could enhance the phosphate precipitation.

FIGURE 5.4 **NUTRIENTS COURSE IN THE UASB$_{ST}$ 2 EFFLUENT DURING EXPERIMENTAL PERIOD**



Soluble nutrient fractions are readily available to the plants (FAO, 1992 & UNEP, 2000), when considering direct reuse of digested effluents in agriculture. Presence of majority of nutrients in the soluble form, is also promising for further resource recovery, such as for instance struvite or calcium phosphate precipitation.

Since the phosphate precipitation in UASB$_{ST}$ 2 seemed to be stronger than in UASB$_{ST}$ 1 an estimation of the prevalence of any phosphate precipitate form the ashes from UASB$_{ST}$ 2 was done. Except of phosphate also Ca and Mg ions were determined. The calcium concentration in the sludge from UASB$_{ST}$ 2 was more excessive than the magnesium concentration with molar ratio between P:Ca and P:Mg of 1:0.9 and 1:0.5 respectively (Table 5.4). Regarding thus the abundance of ions present in the sludge, in UASB$_{ST}$ 2 phosphate was most likely precipitated as calcium phosphate or hydroxyapatite ($CaOHPO_4$).

TABLE 5.4 **IONS CONCENTRATION IN THE SLUDGE ASHES FROM UASB$_{ST}$ 2**

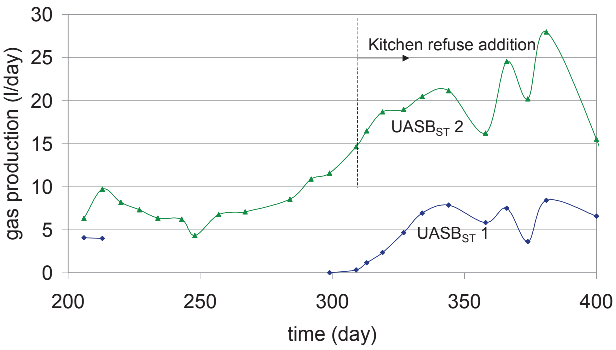
P$_{PO4}$ MGP.MG ASH^{-1}	CA MGCA.MG ASH^{-1}	MG MGMG.MG ASH^{-1}
0.01154	0.01369	0.00385

5.1.3 CONVERSION OF ORGANIC COMPOUNDS

The biogas production in UASB$_{ST}$ 1 was not monitored for the majority of the experimental run. The main reason was formation of a very thick, compact sludge layer inside the GSL separator, preventing release and collection of biogas. The gas production over the last period (day 327 to 408) amounted to 6.4 ± 1.5 l.day^{-1} of biogas.

Before kitchen refuse addition to UASB$_{ST}$ 2, the average measured daily biogas production from UASB$_{ST}$ 2 was 8.3±2.8 l.day^{-1}. The methane content was 72%. After kitchen refuse addition, the biogas production increased to an average of 19.5±4.0 l.day^{-1}. The methanisation rate calculated based on the average amount of biogas produced would be only 18 and 22% respectively. This allows to suppose that especially in UASB$_{ST}$ 2 there was under-estimation of biogas production, since the removal rates of total COD were significantly higher in the latter reactor.

FIGURE 5.5 **DAILY BIOGAS PRODUCTION IN THE UASB$_{ST}$ REACTORS**

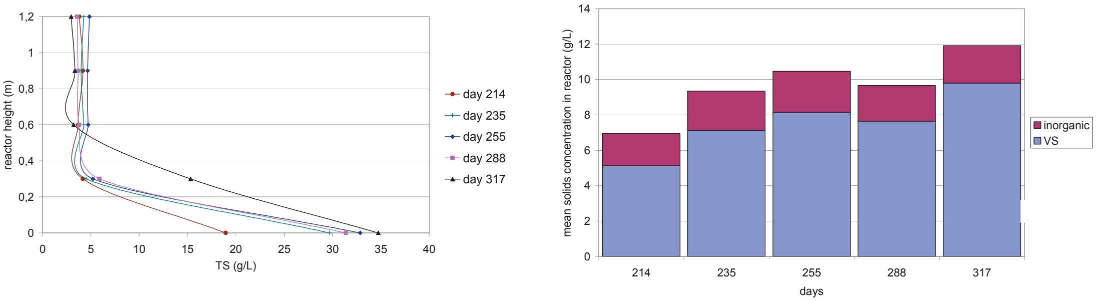


5.1.4 SLUDGE BED DEVELOPMENT

During the experimental period the sludge bed development in the reactors was followed (Figure 5.6). The total solid concentration inside the reactor was increasing in time due to accumulation of particulate (non-degraded and not degradable) organics and non-organic matter. At the end of experimental period (approximately 1 year after the start up) the sludge bed UASB$_{ST}$ 1 occupied less than half of the reactor volume. The concentration of total solid (TS) on the bottom of reactor was 35 g.l^{-1} with 86% volatile solids (VS) content and gradually decreased to 3 g.l^{-1} with 61% VS content at the 5th tap.

The average TS concentration inside the reactor was 12 g.l^{-1} of which 82% was VS. The percentage of the VS was increasing from the beginning of the experimental period. At the beginning, the VS was 74% and increased to 82% of the total solids at the end of the experimental period. This indicates that the sludge bed was more stabilised at start-up (inoculum from other long operating UASB septic tank under ambient temperature conditions).

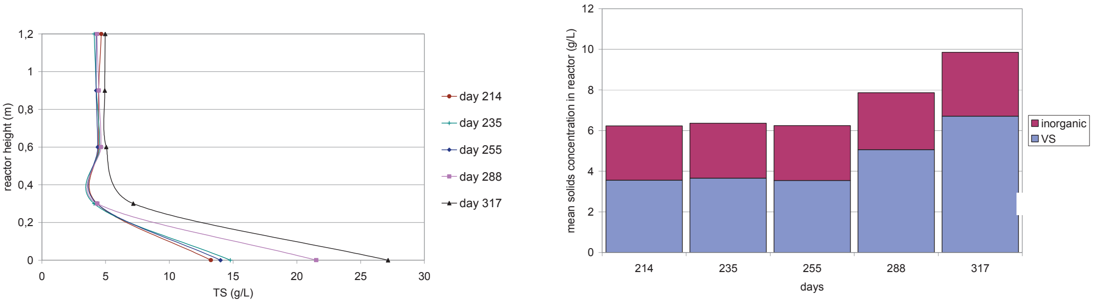
FIGURE 5.6 **SLUDGE BED DEVELOPMENT OVER THE HEIGHT OF UASB$_{ST}$ 1 AND THE AVERAGE AMOUNT OF TS AND VS IN THE REACTOR DURING THE LAST MONTHS OF THE EXPERIMENTAL PERIOD**



In Figure 5.7 the sludge bed development over the height of UASB$_{ST}$ 2 and the average TS concentration during the experimental period are presented. The TS concentration inside the reactor increased in time due to accumulation of particulate matter fraction from the influent and only its partial solubilisation. At the end of the experimental period the sludge bed inside UASB$_{ST}$ 2 occupied almost one third of the reactor height and the sludge was less compact (lower TS concentration) in comparison to UASB$_{ST}$ 1. The lower sludge bed height in UASB$_{ST}$ 2 was because the reactor was not inoculated at the start-up and the hydrolysis process proceeded faster at a higher temperature. The concentration of TS at the bottom of the reactor before kitchen refuse addition was 22 g.l^{-1} with 82% VS and it was gradually decreasing to 4.3 g.l^{-1} with only 42% VS at the 5th tap. The average TS inside the reactor was 8 g.l^{-1} with 64% of VS. The lower VS/TS ratio of the sludge as compared to the reactor operated at 15°C, indicates its higher stability.

After addition of kitchen refuse the concentration of TS at the bottom of the reactor increased rapidly to 27 g.l^{-1} with 83% of VS and was gradually decreasing throughout the reactor height to 5 g.l^{-1} (45% VS) at the 5th tap. The average TS inside the reactor increased to 10 g.l^{-1} (68% VS). The VS fraction was increasing from the beginning of the experimental period. The VS/TS ratio of the average total solids inside the UASB$_{ST}$ 2 was lower than the UASB$_{ST}$ 1 indicating a higher mineralisation rate of accumulated sludge in the first reactor.

FIGURE 5.7 **SLUDGE BED DEVELOPMENT IN UASB$_{ST}$ 2 AND THE INCREASE OF THE AVERAGE AMOUNT OF TS AND VS IN THE REACTOR DURING THE LAST MONTHS OF EXPERIMENTAL PERIOD**

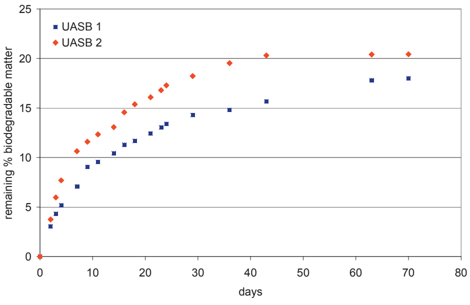


The sludge bed development from the UASB septic tanks indicated that the accumulation of particulate matter was higher in UASB$_{ST}$ 1 compared to UASB$_{ST}$ 2. At higher temperature, the hydrolysis rate increases. The sludge bed development is an important aspect regarding the frequency to withdraw a part of the accumulated particulate matter (sludge). The reactor needs to be emptied when the sludge bed contributes to the deterioration of effluent quality.

5.1.5 SLUDGE STABILITY

The sludge stability is an important property when considering the application of digested sludge in agriculture but also for disposal or further treatment. Sludge having a low stability would lead to on-going biological conversions in the following sludge management step with emissions of CH_4 and CO_2 into the atmosphere and offensive odour problems. Figure 5.8 presents the course of sludge digestibility (% stability = 100 - % digestibility) from both reactors during 70 days of incubation at 35°C. The sludge from both UASB septic tanks had similar digestibility after 70 days incubation, viz. period of 18% and 20 % for UASB$_{ST}$ 1 and UASB$_{ST}$ 2 respectively.

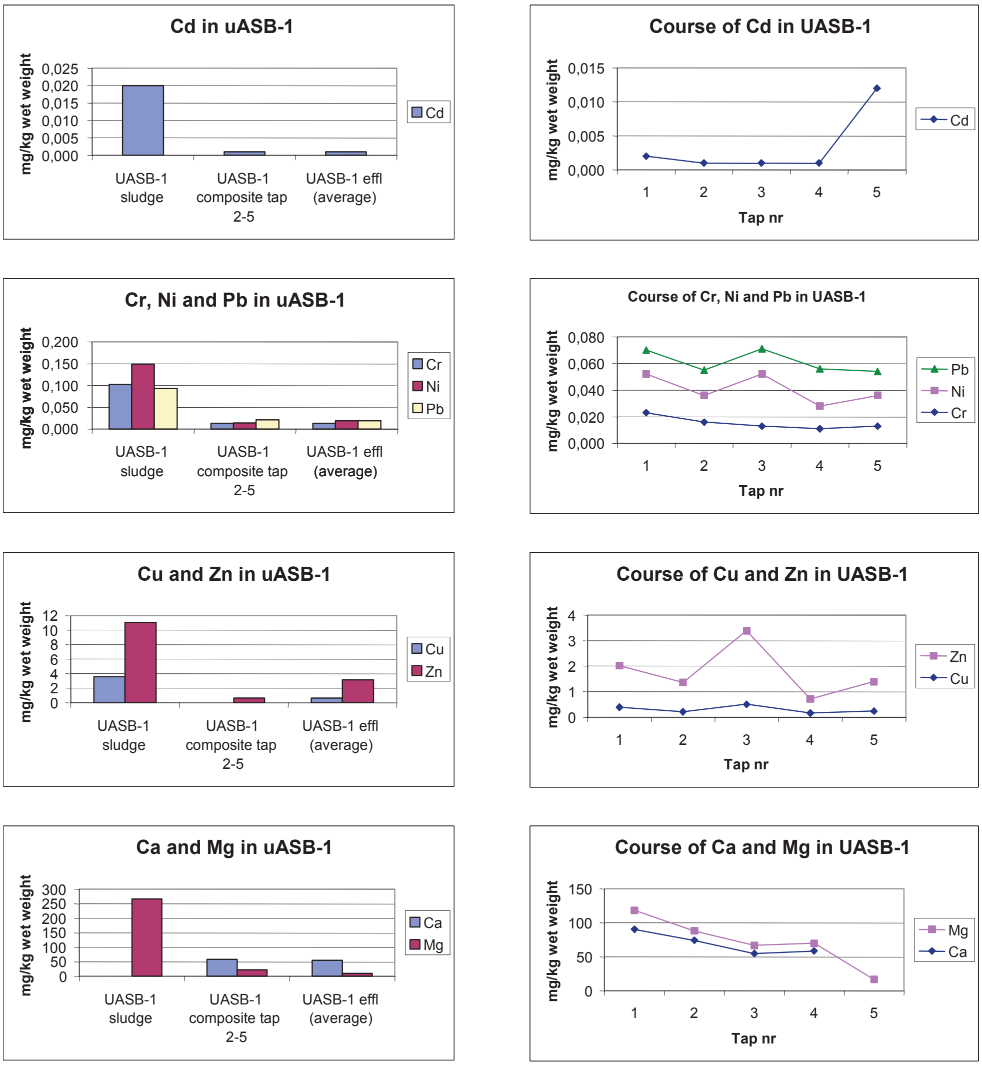
FIGURE 5.8 **THE SLUDGE DIGESTIBILITY OF THE UASB SEPTIC TANKS DURING 70 DAYS OF INCUBATION AT 35°C (SLUDGE SAMPLES TAKEN AT 295TH DAY OF REACTOR OPERATION)**



5.1.6 HEAVY METALS

The distribution of heavy metals in different reactor's phases and their profiles over reactor height are presented in Figures 5.9 and 5.10.

FIGURE 5.9 DISTRIBUTION AND HEAVY METALS PROFILES IN UASB$_{ST}$ 1



The metal concentrations found in this study (table 5.5) can be compared with values in literature describing other types of effluent (appendix 4). For Cd, Ni and Pb the UASB$_{ST}$s effluent concentrations are in the range of the WWTP effluent concentrations found in Asano and Levine (1998), while the concentrations of Cr, Cu and Zn exceed this range. When comparing the metal concentrations in the sludge from both UASB septic tanks reactors with concentrations in WWTP sludge as reported in literature (Snyman, 2001 ; Balmer, 2001; Aulicino *et al*, 1998; Keller *et al*, 2002), they are in the same range. For Cr and Cu, the concentrations are below the range.

In Table 5.5 the effluent from UASB$_{ST}$ 1 and UASB$_{ST}$ 2 are compared with irrigation water quality guidelines (Asano and Levine, 1998) and a commercial fertilizer (Palmquist and Jönsson, 2003). For majority of measured metals, the concentrations in the effluent of UASB$_{ST}$ 1 and UASB$_{ST}$ 2 are below the critic concentrations when comparing to irrigation water. However the ratio metal:phosphorus is often exceeded.

FIGURE 5.10 COURSE OF HEAVY METALS IN UASB$_{ST}$ 2

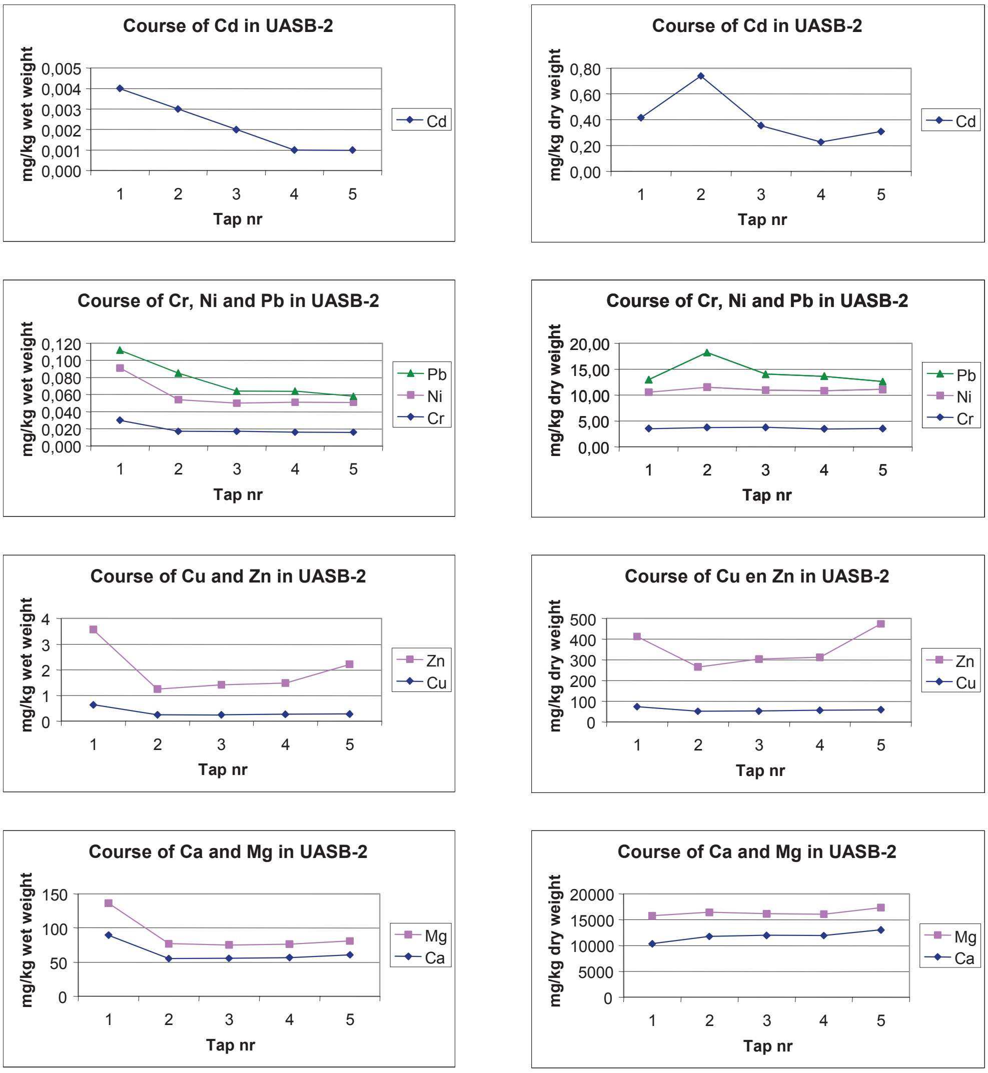


TABLE 5.5 EFFLUENT HEAVY METALS CONCENTRATIONS FROM UASB$_{ST}$ 1 AND 2 AS COMPARED TO EXISTING STANDARDS FOR IRRIGATION (ASANO AND LEVINE, 1998) AND COMPOSITION OF COMMERCIAL, MINERAL FERTILISERS (PALMQUIST AND JÖNSSON, 2003)

	Cd	Cr	Cu	Ni	Pb	Zn
Water quality for irrigation, mg.l^{-1}	0.01	0.1	0.2	0.2	5	2
UASB$_{ST}$ 1 effluent	0.001	0.012	0.137	0.019	0.018	0.53
UASB$_{ST}$ 2 effluent	0.002	0.007	0.186	0.025	0.013	0.773
Commercial fertiliser, mg.kgN^{-1}	0.08	12	2.3	7	0.7	25
UASB$_{ST}$ 1 effluent	0.02	3	4	5	5	15
UASB$_{ST}$ 2 effluent	0.04	0.15	4	1	0.28	16
Commercial fertiliser, mg.kgP^{-1}	0.24	37	6.9	22	2	76
UASB$_{ST}$ 1 effluent	0.34	5	60	8	8	231
UASB$_{ST}$ 2 effluent	0.55	2	58	8	4	242

5.1.7 FATE OF PATHOGENS

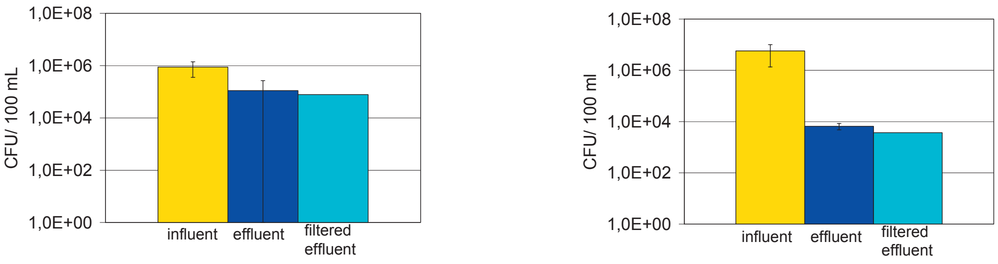
Some reduction of *E.coli* was observed in the UASB reactors, higher for higher operational temperature (Table 5.6). A little difference in *E.coli* concentration between the raw and paper filtered (4.4 μm) UASB effluent samples was shown, but the suspended solids concentration in the effluent was also quite low. No sludge samples from the UASB reactors have been analysed for the presence of microbial indicators.

Compared to concentrations found in literature *F.Coliform* concentrations found in this study were high. There was also a very low reduction in both reactors. The standard for unrestricted irrigation (WHO, 1989) amounts to less than 1000 *E.coli* .100 ml^{-1}. The effluent *E.coli* concentrations are above this concentration in the effluent of both UASB septic tanks. Even after filtration the *E.coli* concentrations are still too high. The WHO standard for *F.Coliform* also amounts to less than 1000 *F.Coliform*.100ml^{-1} (Asano and Levine, 1998). For UASB septic tanks the effluent *F.Coliform* concentrations exceed significantly the standard.

TABLE 5.6 **E.COLI AND F. COLIFORM CONCENTRATIONS IN INFLUENT AND EFFLUENT OF UASB$_{ST}$ 1 AND UASB$_{ST}$ 2**

	UASB$_{ST}$ 1			**UASB$_{ST}$ 2**		
	influent	**effluent**	**filtered effluent**	**influent**	**effluent**	**filtered effluent**
E.coli (CFU.100ml^{-1})						
Number of plates	5	2	1	3	4	1
average	8.84^{E} +05	1.11^{E}+05	7.80^{E}+04	5.70^{E}+06	6.60^{E}+03	3.70^{E}+03
st.dev.	5.26^{E} +05	1.54^{E}+05		4.33^{E}+06	1.81^{E}+03	
log 10	5.95	5.04	4.89	6.76	3.82	3.57
F.coliform (CFU. 100ml^{-1})						
Number of plates	6	7		7	8	
average	1.91^{E} +09	1.64^{E}+09		2.26^{E}+09	1.15^{E}+09	
st.dev.	3.06^{E} +09	3.28^{E}+09		4.52^{E}+09	2.24^{E}+09	
log 10	9.28	9.22		9.35	9.06	

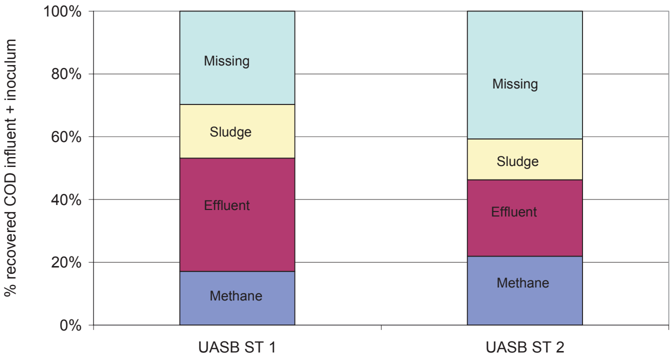
FIGURE 5.11 **EFFICIENCY OF E.COLI REMOVAL IN UASB$_{ST}$ 1 (LEFT) AND 2 (RIGHT)**



5.1.8 MASS BALANCE

The result of estimated mass balance for period of 300 days for both UASB$_{ST}$s is shown in Figure 5.12. According to these estimations for UASB$_{ST}$ 1, only 17.1% of the measured influent COD$_{total}$ (including inoculum) was converted into methane, 17.1% remained in the reactor and 36.1% left the reactor as the effluent. The UASB$_{ST}$ 2 showed a better performance with 21.9% of the influent COD$_{total}$ converted into methane. According to estimation only 13% of total influent COD remained in the reactor as accumulated sludge and 24.4% left the system as effluent. The large gaps in the mass balances (29.7 and 40.7 %) could be most likely due to the under measurement of the biogas production (gas meter malfunction, leakage in the gas collector hose or entrapment of biogas into scum and its escape from the GSL separator. Another reasons could be un-controlled sludge losses during period of scum forming, taking sludge samples for analysis and batch tests and to less extent over-estimation of influent composition).

FIGURE 5.12 COD BALANCE OF UASB$_{ST}$ 'S BASED ON TOTAL INFLUENT COD OVER PERIOD OF 300 DAYS



If the missing fraction was considered only as methane its maximal yield would be 17 and 31 $l.p^{-1}.d^{-1}$ for UASB$_{ST}$ 1 and 2 respectively. Especially the latter value is not realistic since assuming that the daily loading of COD of black water from one individual as 94.3 $g.p^{-1}.d^{-1}$ (Table 3.11) and methanisation of 60% a maximum amount of methane produced at 25°C would be 21.6 $l_{CH4}.p^{-1}.d^{-1}$.

5.2 DESIGN

When designing a conventional septic tank, the main aim is to provide a sufficient hydraulic regime to secure maximal removal of settleable solids. The settled material forms a sludge bed (accumulation) at the bottom of the reactor that (once upon a time) needs to be for a large part removed. Although high removal of particulates can be accomplished in a septic tank, the removal of soluble organic components is low because of the limited contact between biomass and wastewater. The configuration of UASB septic tank is such that the up-flow mode improves the overall removal of organic compounds. For design of the UASB septic tank two criteria can be considered:

1 accumulated sludge has to be stabilised to an extent that at the moment of discharge no nuisance is occurring when disposed.
2 methanogenic activity of the sludge should be sufficient to minimise the VFA content in the effluent at set % of inoculum.

Sludge can be considered as completely stabilised when it does not contain any biodegradable material. It is assumed that the hydrolysis of particulate, biodegradable material ($f_B{\cdot}COD_{part}$) proceeds according to first order kinetics. The ($f_B{\cdot}COD_{part}$) content in the sludge bed can be described using the following equation:

$$COD_{part}(t)= (\Sigma Q\cdot f_B\cdot COD_{part,inf}\cdot e^{-kH\cdot t})/(Q{\cdot}t+V_{inoc}) \quad (mgCOD.l^{-1}) \qquad (eq.5.1)$$

where: $COD_{part,biod}(t)$ concentration of degradable solids in the reactor at time t=t; $f_B{\cdot}COD_{part'inf}$ – concentration of degradable solids in influent ($gCOD.l^{-1}$), Q – influent flow ($l.d^{-1}$), k_H – first order hydrolysis constant of $COD_{part,biod}$, V_{inoc} – volume of inoculum at the start-up of the reactor (l). It is assumed that V_{inoc} does not contain any X_{deg}.

The indication of a required amount of inoculum can be calculated using simplified eq. 5.2:

$$V_{inoc} = (VFA_{CODpart,biod+CODsol,degr})_{inf} + VFA_{infl})) \cdot Q/(SMA \cdot X_{VS}) \qquad \text{(eq.5.2)}$$

where: SMA – specific methanogenic activity of the sludge ($gCOD.gVS^{-1}.d^{-1}$), X_{VS} – volatile solids concentration in inoculum sludge

Modifying the formule of Zeeman and Lettinga (1999) the minimum volume of the UASB septic tank ($V_{min,UST}$) can be calculated according to equation 5.3.

$$V_{min,UST} = (Q \cdot COD_{part})/X \cdot R \cdot (1-H_{max}) \cdot t_{Ac,\,min} + V_{inoc} \qquad \text{(eq. 5.3)}$$

Where: COD_{part} – particulate COD fraction in the influent ($gCOD.l^{-1}$), X – solids concentration in UASB septic tank ($gCOD.l^{-1}$), R – fraction of particulate matter retained in the sludge bed, H_{max} – max fraction of solids that can be hydrolysed, $t_{Ac,min}$ – minimal desired accumulation period of solids (d).

Example: Calculation of the minimal volume of a UASB septic tank to treat concentrated black water from one individual. Daily load of constituents is taken from Table 3.2, where COD_{tot} of concentrated black water amounted to 9.5 $g.l^{-1}$ with 83% suspended COD (= 7.9 $gCOD.l^{-1}$). This characteristics describes black water mixture consisting of 1 portion of faeces (0.138 $g.p^{-1}.d^{-1}$) and 5 portions of urine (0.25 $l.time^{-1}$), each flushed with approximately 1 l of water. The daily flow of this mixture was measured and amounted at 7.0 $l.d^{-1}$, which is close to the theoretical value (=5 · 1.25 + 1 · 1.138 = 7.4 $l.d^{-1}$).

The daily loading of particulate COD is:

$Q \cdot COD_{part} = 7.0\ l.d^{-1} \cdot 7.9\ gCOD.l^{-1} = 55.3\ gCOD.d^{-1}$. Assuming that 70% of COD_{part} is biodegradable, daily $COD_{part,\,degr}$ is 38.7 $g.d^{-1}$.

The minimum demanded accumulation time of the sludge ($t_{Ac'min}$) will mainly depend on the influent characteristics (concentration of biodegradable, particulate organics, influent loading rate) and process temperature (influence on hydrolysis constant) (eq. 5.1). Assuming that reactor ensures stable constant T of 20°C and k_H is 0.1 d^{-1}, approximately 250 days is necessary to ensure 95% of sludge stabilisation.

Inoculum volume, V_{inoc}: It is assumed that inoculum sludge has VS of 20 $g.l^{-1}$ and is characterised by a SMA of 0.05 $gCOD_{CH4}.gVS^{-1}.d^{-1}$ (20°C).

The daily load of VFA from influent biodegradable COD_{sol} (Table 3.2) is $COD_{sol'degr} = 7.0\ l.d^{-1} \cdot (1.4-0.5) \cdot 0.8\ gCOD.l^{-1} = 5.04\ gCOD.d^{-1}$ assuming that this fraction in 80% biodegradable and influent characteristics from Table 3.2. The daily loading of VFA from influent is $VFA_{inf} = 7.0\ l.d \cdot 0.5 = 3.5\ gCOD.d^{-1}$. The volume of inoculum is then $V_{inoc} = (38.7+5.04+3.5)/(20*0.05) = 47$ l.

Finally the minimum volume of UASB reactor needed to reach sludge stabilisation and total reduction of VFA (produced and from the influent) is calculated using equation 5.3. The results of few simple simulations are given in Table. The obtained volumes are per one person.

Q, $l.d^{-1}$	7	7	7	7	7	7	7
COD_{part} $g.l^{-1}$	7,9	7,9	7,9	7,9	7,9	7,9	7,9
X, $g.l^{-1}$	20	20	10	15	20	15	15
X_{COD}, $gCOD.gVS^{-1}$	1,4	1,4	1,4	1,4	1,4	1,4	1,4
R	0,8	0,8	0,8	0,8	0,9	0,8	0,9
H_{max}	0,7	0,7	0,7	0,7	0,7	0,8	0,8
$t_{min,Ac}$, d	250	360	250	250	250	250	360
V_{inoc}, l	47	47	47	47	47	47	47
$V_{min,UST}$, l	165,5	217,6	284	205	180,3	152,3	217,6

5.3 DISCUSSION

5.3.1 ORGANICS REMOVAL

Operation of UASB septic tank operated at two different temperatures demonstrated good particulate COD removal efficiency. The COD_{part} removal efficiency of 88% at 15°C and 94% at 25°C was due to settlement and entrapment of the suspended material in the sludge bed and then its subsequent (partial) conversion.

Regarding the COD_{total} removal and effluent quality, the performance of the reactors was in addition determined by the conversion of COD into biogas. The $UASB_{ST}$ 1 was characterised by an increased VFA concentrations in the effluent compared to the influent. This indicated the production of intermediate products, but only their partial, low conversion to biogas resulting in accumulation and presence in the effluent. The $UASB_{ST}$ 2 effluent was characterised by a lower VFA concentration than the influent, because of the higher methanogenic potential of the reactor contributing to a higher consumption of acetate (prevailing fraction of measured VFA) and higher production of biogas. To get the highest benefit from the UASB septic tank reactor treating very concentrated influent, optimal conditions for enhanced methanogenic activity are required (higher operational temperature, sufficient HRT, optimal contact between sludge and influent).

Higher concentration of the colloidal COD in the effluent of the $UASB_{ST}$ 2 showed possibly higher desintegration) hydrolysis rate of particulate matter.

Short period when kitchen refuse was added to $UASB_{ST}$ 2 (25°C) did not negatively affect the suspended COD removal in the reactor. This period was however too short to draw stronger conclusions. More research is needed on the longer term effect of kitchen refuse addition on the performance of UASB septic tank reactor. The attention has to be paid then on the effluent quality, biogas yield, rate of build-up of sludge bed, scum forming, etc.

5.3.2 BIOGAS PRODUCTION

The difference in operational temperature influenced the biogas production in UASB septic tanks. During its ‘adaptation’ phase the $UASB_{ST}$ 1 could produce 4.7 l $biogas.day^{-1}$ while the $UASB_{ST}$ 2 produced average amount of biogas equal to 8.3 l $biogas.day^{-1}$. In the last period the biogas production increased to 6.3 $l.day^{-1}$ and 10.4 $l.day^{-1}$ respectively. After kitchen refuse was added to $UASB_{ST}$ 2 the production increased to average 18 $l.day^{-1}$. These are rather low values indicating around 20% of influent methanisation. Gaps in the mass balances may be also partially explained by underestimation of biogas.

Analogously to considerations on potential energy reuse as presented in section 4.3 for accumulation systems the energy balance was done for UASB septic tanks (Table 5.7). This energy balance was estimated for the purpose of vacuum installation and reactors when they need to be heated to e.g. 25°C to obtain a better process efficiency. The energy balance calculation showed again a surplus energy, which can be reused in households. However in the real situation there are always energy losses depending on which way the methane is collected and utilised as energy source (heat, electricity). The addition of kitchen refuse into the reactor increases the margin of surplus energy.

TABLE 5.7 ENERGY BALANCE FOR EACH REACTOR USED IN THIS STUDY

	Vacuum pump	Wastewater pump	Heating	Methane Production	Energy balance
	Joule.week^{-1}				
UASB$_{ST}$ 1	2.475E+06	1.554E+06		8.320E+06	4.291E+06
UASB$_{ST}$ 2	2.475E+06	1.170E+06	9.677E+06	1.431E+07	9.868E+05
	2.640E+06*	1.848E+06*	9.677E+06	3.108E+07*	1.692E+07*

*after kitchen refuse addition

5.3.3 NUTRIENTS AND ORGANICS IN THE EFFLUENT AND SLUDGE

The effluent of the UASB$_{ST}$ reactors contained biodegradable organics, high ammonium content (around 1.0 gN.l^{-1}) and soluble phosphate (around 60 mgP.l^{-1}). Sludge bed on the other hand is rich in carbonaceous compound (well stabilized) and soluble and particulate nitrogen and phosphorus.

The fact that the nutrients are present mainly in a soluble form, make reuse of the effluent in agricultural field an attractive option. The soluble nutrients are readily available for the plant and can be further processed to form precipitated complexes, such as calcium phosphate ($CaOHPO_4$) or struvite ($MgNH_4PO_4.6H_2O$) using appropriate additives to enhance the precipitation (Mg, P). The precipitated phosphate complexes are important compounds regarding the fact of approaching depletion of the natural sources of phosphate (Steen, 1998 and Johnston, 2003).

The addition of kitchen refuse to the anaerobic reactors fed with concentrated black water seems to enhance the formation of precipitation complexes. The phenomenon could be especially observed in the accumulated influent of AC2 but also in the UASB$_{ST}$ 2 at the last described period of operation when kitchen refuse was dosed. Followed fate of nutrients during the batch tests for determination of hydrolysis constant using different types of influent proved that precipitation of complexes (P but also NH_4) occurs in each reactor to lesser or higher extent.

5.3.4 CONSTRAINTS IN AGRICULTURAL REUSE

The effluent from the UASB septic tanks had salinity (expressed in conductivity of the sample) of 0.67 and 1.15 dS.m^{-1} for UASB$_{ST}$ 1 and 2 respectively. These values are still in the usual range of soluble salts content not giving a side effect to soil or crop. However, accumulation of salt in the soil still could occur, hence for purpose of reuse in agricultural field, proper management regarding application and crop selection is required to minimise the environmental effect. Diluting the effluent with rainwater or treated grey water could also be an option (but then also nutrients diluted) when considering irrigation. The salinity will decrease due to dilution and the mixture will contain sufficient nutrients for the crops.

For direct application in the agricultural field, the fate of pathogenic organisms and heavy metal in the sludge should be taken into account. Brief studies on the fate of pathogens and heavy metals in UASB septic tanks treating concentrated black water under 15 and 25°C, showed that E.coli are concentrated in the reactor sludge and there occurs just a small reduction. Regarding presence of pathogens in the effluent and associated health risk, the effluent needs the most probably to be further treated to meet the standards or stored for a longer period. According to this information for instance anaerobic digestion at approximately 15 days at 35ºC to 55ºC could be sufficient to treat the sludge before applied in agricultural field. Secondary treatment is connected however with additional costs.

The concentration of heavy metals in the effluent did not exceed the standard for irrigation. When comparing with commercial fertiliser few elements tended to be higher in the effluent of UASB septic tank especially when related to phosphate content.

The way of application of digested effluent need to be chosen in such away that possible contact between the worker and the effluent is minimal, such as sub-surface irrigation. The selection of non-edible crop and crops that require cooking before consumption will also minimize the health impact of reused digested effluent.

6

GENERAL DISCUSSION

Application of vacuum toilets can result in a significant reduction of clean water consumption. Conventional toilets consume 6-9 $l.flush^{-1}$, while some types of vacuum toilets only 1 $l.flush^{-1}$. The total water consumption to collect and transport toilet waste would be then 7.5-10.5 $l.p^{-1}.day^{-1}$ resulting in 70-80% reduction in water consumption for flushing or approximately 20% in relation to total drinking water consumption in the Netherlands (NIPO/VEWIN, 2002).

Separation of black water from other wastewater streams and its separate treatment minimises diffusion of pollutants. For instance pathogens, pharmaceutical residues and hormones are mainly present in toilet wastewater and can be better controlled by remaining concentrated. Also nutrients, nitrogen and phosphate, are concentrated for approximately 90 and 75% respectively in black water. If retained in this small volume they can be better recovered, reused or removed if required.

Implementation of vacuum toilets increases the strength of wastewater when comparing to the traditional approach. High organic content implies that anaerobic digestion can be applied as a core technology, minimizing energy consumption and preserving valuable resources (energy, nutrients). In addition to black water, kitchen refuse can be co-digested, improving the biogas production as it contains a high content of highly biodegradable organic matter. In this research, two types of simple anaerobic digesters were tested, an accumulation system and a UASB septic tank, to evaluate their applicability in a DESAR concept. Three types of concentrated wastewater mixtures were used as influent: black water, black water with kitchen refuse and brown water with kitchen refuse.

The accumulation system is designed to provide storage and digestion in one volume and is continuously fed, by which its effective (digestion) volume is increasing in time. In this sense extra volume usually needed for storage of digested effluent required in continuous systems is avoided. The advantage of the long retention times applied in accumulation systems is that even at a lower temperature (like applied in this research - 15°C) high efficiency can be achieved. After the maximum volume or the demanded storage time is reached, the reactor is emptied. If still possible or required it can be left without feeding for further stabilisation - additional storage. Since the primary objective to apply an AC reactor is to reuse the digested, nutrient-rich mixture in agriculture, the storage period is determined by the required frequency of fertilisation. In Dutch conditions it is forbidden to spread fertilisers in the winter period, therefore approximately 6 months of storage are required. When emptying the reactor after the required storage period a minimum volume of digested medium, serving as well adapted inoculum should be left for the next run.

An accumulation reactor is a very simple configuration, mixing is not necessary devices and when storage time is long enough no heating is needed. The produced biogas can be reused on-site as energy source (heat, electricity) when the application scale is large enough to make it economically feasible. In that case facilities for collection, treatment and further management of generated biogas are needed.

The performance of the accumulation system is influenced by the amount and activity of inoculum, operational temperature, wastewater composition and reaction-storage period. To get the best possible performance the type and amount of inoculum should be chosen carefully. The best way is to start up the reactor with already adapted to the actual conditions inoculum (especially higher ammonium concentrations). The factors to be taken into account are wastewater composition, its loading rate and operational temperature. If the sludge seed used to inoculate the reactor was not adapted, longer times will be required to reach a steady state of the system. An example calculation (section 4.2) suggests to use approximately 9% of inoculum in relation to total reactor volume at the end of accumulation period when system treats concentrated brown water and kitchen refuse at 20°C. The characteristics of inoculum used in that example was 20 $gVS.l^{-1}$ and SMA of 0.1 $gCOD_{CH4}.gVS^{-1}.d^{-1}$.

The maximum attained methanisation rate was 58% of the total influent COD in accumulation systems fed with brown water and kitchen refuse at 20°C. An additional storage without feeding can still be applied (if there is a possibility to use two tanks or when no influent is produced for a certain period like for instance in only seasonally occupied locations) leading even to the complete anaerobic stabilisation of the reactor content.
The highest measured biogas yields were approximately 35 $l_{biogas}.p^{-1}.day^{-1}$. The theoretical COD loading from faeces (50g $COD.p^{-1}.d^{-1}$) and kitchen refuse (60 $gCOD.p^{-1}.d^{-1}$) is 110 $g.p^{-1}.d^{-1}$. When assuming a digestion efficiency of 60% based on the obtained results, approximately 40 l of biogas per person could be produced daily. In all runs sufficient biogas was produced even under sub-optimal conditions, to cover the energy requirements of the reactor (heating) and the vacuum installation (electricity). Reuse of biogas and its transformation into any energy form will be determined by scale of application.

Digested medium consists of liquid and sludge. Stratification occurs in the reactor during the feeding. An AC system is not designed for separation between solids and liquid; mixing will take place during emptying resulting in relatively high effluent COD concentrations.
High concentrations of ammonium are present in the reactor content, even higher for a mixture of brown water and kitchen refuse than for black water and kitchen refuse. Precipitation of phosphate will occur in these types of digesters depending on influent composition. Co-digestion of kitchen refuse and brown water enhances this process.
Reduction of pathogens (*E.coli* was mainly determined) is a function of retention time and operational conditions. In an accumulation system with an additional post-digestion period (in this research 140 days) satisfactory removal of *E.coli* could be reached. Due to the lack of any other reference, WHO guidelines (irrigation water) were taken for comparison (below 1000 cfu.100ml). This implicates additional advantage of applying additional storage without feeding.
In sludge samples the critical concentrations for Cu and Zn were exceeded. When comparing metal concentrations with composition of commercial fertilisers, three elements tended to be higher – Cd, Cu and Pb.

When treating black water in an accumulation system still significant influent volumes are provided resulting finally in relatively large reactor volumes. To give an idea of required volume for 6 months storage approximately 1.4-1.6 m^3 reactor volume per person is needed to treat black water with kitchen refuse while when urine is separately collected only 0.3-0.5

m^3.person^{-1} is required. From this point of view an accumulation system is mainly recommended for concentrated media collected with a limited amount of flushing water, e.g. using vacuum separation toilet that needs to be developed.

The *UASB septic tank* is meant as an improvement of the conventional septic tank applied world wide for on-site treatment of domestic wastewater or black water. In a septic tank the removal of pollutants is mainly based on the separation of the particulate and soluble fraction by means of sedimentation and floatation. If the UASB principle to lead the wastewater through the sludge bed in up-flow mode is applied to the septic tank, a better contact between the wastewater and the sludge is obtained. In this way an improvement of physical removal of particulate matter and biological conversion of dissolved compounds is attained. A slight modification of a conventional septic tank into an UASB septic tank results in a better treatment efficiency of domestic wastewater. The long sludge retention time (SRT) enables high stabilisation of retained solids. Contrary to accumulation system a UASB septic tank provides an (efficient) separation of the solid and liquid phases. Continuously produced effluent contains nutrients, mainly in the soluble form of ammonium and phosphate. Depending on operational temperature and sludge-wastewater contact high removal of dissolved organic matter can be achieved. Heavy metals in the effluent do not constitute any problem for reuse in light of existing standards (e.g. irrigation). However concentration of pathogens will still be high. Further treatment of the effluent will be probably necessary. In some cases direct reuse for close locations can be considered (e.g. urban agriculture).
Solids are concentrated in the sludge bed that slowly develops. The frequency to remove part of the sludge will depend on process conditions, efficiency and required rate of sludge stabilisation. Maximal once per year part of the sludge bed will be discharged and can be reused in agriculture.

Comparison of the two UASB septic tanks operating under two different temperatures fed with the same type of influent show that a higher temperature is removing the reactor performance. At a higher operational temperature of 25ºC, a higher efficiency of COD_{total} removal and biogas production could be achieved than at 15ºC.
Since the operation of two reactors was followed for a period of one year, it is expected that further build up of he sludge bed, improving biomass-wastewater contact and increasing methanogenic potential could still improve the effluent quality.

Short lasting enrichment of the influent to $UASB_{ST}$ 2 (25ºC) with kitchen refuse contributed to the instantaneous increase in the biogas production. The effluent quality deteriorated but not proportionally to the increase in influent strength that was almost double. Further research would be of interest to observe a long term effect of co-digestion on the process performance. The following parameters are of importance: effluent quality and biogas yield but also operational aspects such as occurrence of scum and rate of sludge bed development and its stability.

Operation of a UASB septic tank at higher temperature has, except of giving a better performance, an advantage that the frequency of sludge withdrawal is lower than when operating the same reactor under lower (ambient) temperatures. Higher conversion rate in $UASB_{ST}$ 2 (25ºC) contributed to a lower build up of the sludge bed than in the reactor operated at 15ºC.

Application of a more constant feeding to the reactor could result in a better contact between sludge bed and the influent (up-flow velocity). It would provide a better mixing and could minimise the possible nuisances related to possibility of scum forming. More constant feeding of the reactor can be secured when the scale of application is larger.

7

CONCLUSIONS

Anaerobic digestion was applied for treatment of concentrated domestic wastewater streams in a DESAR concept. Two simple digesters were extensively tested: an accumulation reactor and a UASB septic tank.

Two accumulation digesters were fed with two types of mixtures: (1) black water and kitchen refuse and (2) brown water and kitchen refuse at both 15 and 20°C.

Implemented vacuum toilets, consuming 1l water per flush, enabled to obtain concentrated wastewater. The whole system consisting of a vacuum unit and a reactor was working trouble-less throughout all research runs. An anaerobic accumulation system ensured stable, robust digestion of concentrated waste(water).

The highest methanisation of 58% was obtained for the system fed with brown water and kitchen refuse at 20°C. Additional storage without feeding improved the digestion efficiency, up to 10% approaching maximum biodegradability of the substrate.

Consequently the accumulated mixture will be well stabilised. Soluble organic compounds were present in low concentrations and represented mainly anaerobic inert material.

Stratification occurs in the reactor during the feeding since no mixing devices are installed. Mixing of sludge and supernatant will occur during reactor's emptying and the effluent will contain both fractions.

The average ammonium concentrations in the reactors were between 1.5 and 1.8 $gN.l^{-1}$. High ammonium levels did not effect the process performance. Average ortho-phosphate concentrations in the digested mixtures were 100 $mgP.l^{-1}$. The total phosphate content in the sludge phase was gradually increasing, indicating precipitation of phosphorus. Phosphate precipitation tended to be stronger in the reactor fed with brown water and kitchen refuse.

It was observed that additional storage of the digested mixture inactivates *E.coli* significantly.

The concentration of some heavy metals (Cu, Zn) in the sludge exceeds Dutch standard level for field application. The digested mixtures (sludge + supernatant) comply with latter.

The application of an accumulation system is especially recommended to treat highly concentrated mixtures like brown water and kitchen refuse collected with vacuum system. As a result of the high concentration (low dilution) the reactor volume to be installed then is limited to 0.3-0.5 m^3 per person when 180 days filling period is required.

The operation of *two UABS septic tanks* fed with concentrated black water and operating at 15 and 25°C was monitored for a period of one year. Operating a $UASB_{ST}$ at higher temperature results in a COD_{total} removal of approximately 80% and COD_{part} of 94%. At lower temperature of 15°C removal of COD_{part} was 88% but removal of total COD attained only 61%.

The system performed trouble-less throughout the considered period. The formation of a scum layer during the start-up phase of both reactors was faced as the only operational nuisance.

The effluent ammonium and phosphate concentrations were 0.83 and 1.1 $gN.l^{-1}$ and 52 and

59 mgP.l^{-1} respectively. A major fraction of N and P was in a soluble form (~ 80-90%).
The build-up of the sludge bed was rather slow, implicating that sludge withdrawal does not need to take place often. Long accumulation of sludge makes it well stabilised.
Heavy metals content in the effluent complied with the irrigation standards. The inactivation of pathogens was not sufficient in UASB septic tanks.

Contrary to the AC system, the UASB septic tank is recommended to treat less concentrated influent such as black water. An additional 'polishing' unit, whether for further mineralisation of the effluent, nutrient recovery or pathogens removal may be required.

LITERATURE

Airvac (2004) http://www.airvac.com/toilets.htm

Alef, K. and Nannipieri, P. (1995). Urease Activity. In: Alef K. & Nannipieri P.(eds) Methods in Applied Soil Microbiology and Biochemistry. London: Academic Press Ink.

Almeida, M.C., Butler, D. and Friedler, E. (1999). At source wastewater quality. Urban Water 1, pp. 49-55.

Alvarez, E.A. ; Mochon, M.C. ; Sanchez, J.C.J. ; Rodriguez, M.T. (2002) Heavy metal extractable forms in sludge from wastewater treatment plants. Chemosphere Vol. 47, pp. 765-775.

Angelidaki, I. and W. Sanders (2004) Anaerobic biodegradability of macropollutants, Submitted to Reviews in Environmental Science and Biotechnology

Angelidaki, I. ,and Ahring, B.K. (1994) Anaerobic thermophilic digestion of manure at different ammonia loads: the effect of temperature. Wat. Res. 28 (3):727-731.

AOO/IPA (2000) Status report Program separated collection of household waste 1998. IPA 2000-2.

AOO/IPA (2001) Informatie punt afval. Statusrapportage programma gescheiden inzamelen van huishoudelijk afval 1999, AOO-IPA, 2001-77p, ISBN 90-5484-221-0, Utrecht

Asano, T. and Levine, A.D. (1998) Wastewater reclamation and reuse. Technomic Publishing Company, Inc., Lancaster, PA, USA.

Aulicino, F.A.; Colombi, A.; Calcaterra, E,; Carere, M.; Mastrantonio, A.; Orsini, P. (1998) Microbiological and chemical quality of sludges from domestic wastewater plants Int. Journal of Env. Health Research Vol. 8, pp 137-144, 1998

Balmer, P. (2001) Possibilities to improve the quality of wastewater sludges Wat.Sci.Techn. Vol. 44 No.10 pp 19-26.

Batstone, D.J., J. Keller, I. Angelidaki, S.V. Kalyuzhnyi, S.G. Pavlostathis, A. Rozzi, W.T.M. Sanders, H. Siegrist and V.A. Vavilin (2002) The IWA Anaerobic Digestion Model No 1 (ADM1) Water Science and Technology Vol 45 No 10 pp 65–73.

Bentz, R.R., Dimcheff, D.G., Nemiroff, M.J., Tsang, A. and Weg, J.G. 1975. The incidence of urine cultures positive for Mycobacterium tuberculosis in a general tuberculosis patient population. American Review ofRespiratory Disease 111(5):647-650.

Bergen, V. W. J. v.d, (1997). Challenges in urban water management. Proceedings of the third Japan-Netherlands workshop on integrated water quality management.

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

Bofill-Mas, S., Pina, S. and Girones, R. 2000. Documenting the epidemiologic patterns of polyomaviruses in human populations by studying their presence in urban sewage. Applied and Environmental Microbiology 66(1):238-245.

Bogte, J.J., Breure, A.M., van Andel, J.G. and Lettinga, G. (1993) Anaerobic treatment of domestic wastewater in small scale UASB reactors. Wat. Sci. Tech. 27 (9), 75-82.

BOOM (1998) Besluit van 30 januari 1998, houdende regels met betrekking tot de kwaliteit en het op of in de bodem brengen van overige organische meststoffen (Besluit kwaliteit en gebruik overige organische meststoffen)

CBS (2000) Central Bureau Statistics. Water quality management, part b. Treatment of wastewater (in Dutch) Voorburg/Heerlen, The Netherlands.

CDC (2000a). Leptospirosis. Centers for Disease Control and Prevention. http://www.cdc.gov/ncidod/dbmd/diseaseinfo/leptospirosis_g.htm. Last updated 31 March 2000.

CDC (2000b). Schistosomiasis. Centers for Disease Control and Prevention.

http://www.cdc.gov/ncidod/dpd/parasites/schistosomiasis/factsht_schistosomiasis.htm. Last updated 7 June 2000.

Christ, O., P. A. Wilderer, et al. (2000). "Mathematical modelling of the hydrolysis of anaerobic processes." Water Science and Technology 41(3): 61-65.

Cooley A., Stravinsky D. and James T. B. Tripp (1996) The Village of Bellport's Program for the Home Composting of Kitchen refuses Waste Reduction Coalition, and General Counsel, Environmental Defense Fund. http://www.environmentaldefense.org/pubs/Reports/compost.html

Dailloux, M., Laurain, C., Weber, M. and Hartemann, Ph. 1999. Water and nontuberculosis mycobacteria. WaterResearch 33(10):2219-2228.

Decentrale Sanitatie en Hergebruik op Gebouwniveau (DESAH), EET kiemproject 98115 (2000),

Wageningen Universiteit, Agrotechnologie en Voedingswetenschappen, Sectie Milieutechnologie.

Dinopoulou, G., Rudd, T. and Lester, J.N. (1988). Anaerobic acidogenesis of a complex wastewater, I. The influence of operation parameters on reactor performance. Biotechnol. Bioeng. 31 (9), 958-968.

Drenthe in cijfers (2003) http://www.cijfers.drenthe.nl/hoofdstukken/10/102.htm

Duncanson, M., Russell, N., Weinstein, P., Baker, M., Skelly, C., Hearnden, M. and Woodward, A. (2000). Rates of notified cryptosporidiosis and quality of drinking water supplies in Aotearoa, New Zealand. Water Research 34(15):3804-3812.

Duynhoven, van A.H.M. (1994). Verwijdering van organisch keukenafval. De beoordeling van drie verwijderingsmethoden voor het Academisch Ziekenhuis Nijmegen. Verslagen Milieukunde nr. 79

EHC (2001) Environment Health Criteria ZINC. First draft, Published under the joint sponsorship of the United Nations Environment Programme, the International Labour Organization, and the World Health Organization, and produced within the framework of the Inter-Organization Programme for the Sound Management of Chemicals. World Health Organization, Geneva, 2001

El-Gohary, F. A. (2001). DESAR treatment concepts for combined domestic wastewater. Decentralised Sanitation and Reuse; Concepts, systems and implementation. P. Lens, G. Zeeman and G. Lettinga (edts). London, IWA Publishing.

El-Mashad, H. (2003) Solar thermophilic anaerobic reactor (STAR) for renewable energy production. Phd Thesis, Sub-department of Environmental Technology, Wageningen University.

Elmitwalli, T.A. (2000). Anaerobic treatment of domestic sewage at low temperature. PhD Thesis, Sub-department of Environmental Technology, Wageningen University.

Elmitwalli T. A., Soellner J., Keizer A. de, Zeeman G., Bruning H. and Lettinga G. (2001). Biodegradability and change of physical characteristics of particles during anaerobic digestion of domestic sewage. Wat. Res., 35(5), 1311-1317.

Elmitwalli, T., S. Sayed, et al. Decentralised treatment of concentrated sewage at low temperature in a two-step anaerobic system: two upflow-hybrid septic tanks.

Eriksson, E., Auffarth, K, Henze, M, Ledin, A.(2002). Characteristics of grey wastewater. Urban Water , Volume 4 (1) , 85-104

EPA (1980) Design manual: onsite waste water treatment and disposal systems (EPA 625/1-80-012).

EPA report 822-R-02-038 (2002) Edition of the drinking water standards and health advisories. Office of Water, U.S. Environmental Protection Agency, Washington, DC, 2002.

FAO (1992) Wastewater treatment and use in agriculture – FAO irrigation and drainage paper 47 Pescod, M.B., Food and Agricultural Organisation of the United Nations, Rome.

Fittschen, I. and Hermann, H.H. (1998). Characterization of the municipal wastewaterpart human urine and a preliminary comparison with liquid cattle excetion. Water Science and Technology 38, pp. 9-16.

Friedler, E., D. M. Brown, et al. (1996). “A study of wc derived sewer solids.” Water Science and Technology 33(9): 17-24.

Feachem, R.G., Bradley, D.J., Garelick, H. and Mara, D.D. (1983) Sanitation and Disease - Health aspects of excreta and wastewater management. John Wiley and Sons, Chichester, UK.

Gopinath, R. and Keystone, J.S. 1995. Ascariasis, Trichuriasis, and Enterobiasis. In: Blaser, M.J., Smith, P.D., Ravdin, J.I. and Greenberg, H.B. (eds.) Infections of the gastrointestinal tract. Raven Press, Ltd., New York, NY, USA. p. 1167-1177.

Grange, J.M. and Yates, M.D. 1992. Survey of Mycobacteria isolated from urine and the genitourinary tract in south-east England from 1980 to 1989. British Journal of Urology 69:640-646.

Guyton A.C. (1992). Human Physiology and Mechanisms of Disease. Philadelphia: W.B.Saunders Company

Haas, C.N., Rose, J.B. and Gerba, C.P. 1999. Quantitative Microbial Risk Assessment. John Wiley and Sons, Inc., New York, NY, USA.

Haytoglu, B. , Demirer, G.N., Yetis, U. (2001) Effectiveness of anaerobic biomass in adsorbing heavy metals.

Wat. Sci. Techn. Vol. 44, No. 10, pp. 245-252.

Halalsheh, M. M. (2002) Anaerobic Pre-treatment of strong sewage, a proper solution for Jordan. Phd Thesis, Sub-department of Environmental Technology, Wageningen university.

Hellström D. and Kärrman E. (1996). Nitrogen and phosphorus in fresh and stored urine. Proceedings from the 2nd International Conference on Ecological Engineering for Wastewater Treatment, September 18-22 1995. Waedenswil: Environmental Research Forum pp 221-226.

Helström, D. (1998) Nutrient management in sewerage systems: investigations of components and exergy analysis. PhD thesis, Luleå University of Technology.

Henze, M. (1997) Waste design for households with respect to water, organics and nutrients. Wat. Sci. Tech. 35(9), 113-120.

Höglund, C (2001) Evaluation of microbial health risks associated with the reuse of source-separated human urine, Doctoral thesis Royal Institute of Technology (KTH), Department of Biotechnology, Applied Microbiology, Swedish Institute for Infectious Disease Control (SMI) Department of Water and Environmental Microbiology

Huitema, H. 1969. The eradication of bovine tuberculosis in cattle in the Netherlands and the significance of man as a source of infection for cattle. Selected papers of the Royal Netherlands Tuberculosis Association 12:62-67.

InfoMil, (2002) Dutch notes on BAT for the incineration of waste. Ministry of Housing, Spatial Planning and the Environment

Janssen, A., A. Mels, et al. (2003). Anaerobic digestion of wastewater after primary separation, experiments on anaerobic biodegradability of raw and settled wastewater. Wageningen, Lettinga Associates Foundation (LeAF) for Environmental Protection and Resource Conservation: 29.

Jawetz, E., Melnick, J.L. and Adelberg, E.A. 1987. Review of medical microbiology, 17th ed. Appleton & Lange, East Norwalk, CT, USA.

Jefferson, B, Judd, S, Diaper, C. (2001). Treatment methods for grey water. In: Decentralised Sanitation and Reuse-, Concepts, Systems and Implementation. Eds. P. Lens, G. Zeeman and G. Lettinga. IWA Publishing, UK.

Johnston, F.A. and McMillan, T.J. (1952). The amount of nitrogen retaines by 6 young women on an intake of approximately 70 gm of protein a day. Journal of Nutrition 35, pp. 425-433

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

Johnston, A.E. (2003) Understanding potassium and its use in agriculture. European Fertiliser anufacturers Association Publications, Brussels, Belgium.

Keller, C. ; McGrath, S.P.; Dunham, S.J. (2002) Trace metal leaching through a Soil-Grassland system after sewage sludge application. Journal of Env. Quality 31: 1550-1560.

Kirchmann, H. and Pettersson, S. (1995). Human urine - chemical composition and fertilizer use efficiency. Fertilising Reseach 40, pp. 149-154.

Koning, de J. (2003) Afval het riool in. De 22e Vakantiecursus in Riolering en Afvalwaterbehandeling

Kujawa-Roeleveld, K., Elmitwalli, T., Gaillard, A., Leeuwen, M. van and Zeeman, G. (2003) Co-digestion of concentrated black water and kitchen refuse in an accumulation system (AC) within DESAR concept Wat. Sci. Tech. 48(4): 121-128.

Kujawa-Roeleveld, K., T. Elmitwalli, et al. Anaerobic digestion of physiological waste and kitchen refuse towards resource management in the DESAR concept. Wageningen, Wageningen University.

Leeuwen, M. V. (2002). Feasibility of Accumulation reactors for the digestion of concentrated blackwater in combination with swill. A study within the DESAR concept. Sub-department of Environmental Technology. Wageningen University: 58.

Lentner, C. and Wink, A. (1981). Units of Measurement, Body Fluids, Composition of the Body, Nutrition. Geigy Scientific tables. Basle: Ciba-Geigy

Lettinga, G. and Hulshoff Pol, L.W. (1991). UASB-process design for various types of wastewaters. Wat. Sci. Tech. 24 (8), 87-107.

Lettinga, G., K. Van Knippenberg, et al. (1991). Upflow Anaerobic Sludge Blanket (UASB); Low Cost Sanitation Research Project in Bandung/Indonesia. International Institute for Hydraulic and Engineering, Delft. Department of Water Pollution Control, Agriculture University of Wageningen. St. Borromeus Hospital, Bandung: 123.

Lettinga, G. and Haandel, van A.C. (1993) Anaerobic digestion for energy production and environmental protection, in Renewable energy; sources for fuels and electricity, T.B. Johansson, et al., Editors, Island press, California: London, p. 817-839.

Lettinga, G., Man, A. d, Last, A. R. M. v. d., Wiegant, W., Knippenberg, K., Frijns, J. and Buuren, J. C. L. v. (1993). Anaerobic treatment of domestic sewage and wastewater. Wat. Sci. Tech., Vol 27(9): 67-73.

Lettinga, G., Hulshof Pol, L.W. and Zeeman, G. (1993). Collegedictaat biologische waterzuivering, onderdeel anaerobe zuivering, Wageningen agricultural University, Department of Environmental Technology.

Lettinga, G., L. W. Hulshoff Pol, et al. (1998). Lecture notes: Biological Wastewater Treatment Part I Anaerobic Wastewater Treatment. Wageningen, Sub-department of Environmental Technology. Wageningen University.

Lettinga, G., Lens, P., et al. (2001). Environmental protection technologies for sustainable development. Decentralised Sanitation and Reuse; Concepts, systems and implementation. P. Lens, G. Zeeman and G. Lettinga (edts). London, IWA Publishing.

Lettinga, G., Rebac, S. et al. (2001). "Challenge of psychrophilic anaerobic wastewater treatment." Trends in Biotechnology 19(9): 363-370.

Lewis-Jones, R. and Winkler, M. 1991. Sludge parasites and other pathogens. Ellis Horwood Limited,

Chichester, UK.

Luostarinen, S., Sanders, W., Kujawa-Roeleveld, K. and Zeeman, G. (2004). Anaerobic treatment of black water in UASB-septic tank systems, under preparation

Mahmoud N.(2002) Anaerobic Pre-treatment of sewage under low temperature (15 ^{o}C) conditions in an integrated UASB-digester system. Ph. D. thesis, Sub-department of Environmental Technology, Wageningen University, The Netherlands.

Marshall, M.M., Naumovitz, D., Ortega, Y. and Sterling, C.R. 1997. Waterborne protozoan pathogens. Clinical Microbiology Reviews 10(1):67-85.

Mes, T. (2001) Personal communication. Faecal weight.

Mes, T.Z.D., Stams, A.J.M., Reith, J.H. and Zeeman, G. (2003) Methane production by anaerobic digestion of wastewater and solid wastes. In Bio-methane and bio-hydrogen. Edited by Reith, J.H., Wijffels, R.H. and Barten, H. Dutch Biological Hydrogen Foundation.

Mead, P.S., Slutsker, L., Dietz, V., McCaig, L.F., Bresee, J.S., Shapiro, C., Griffin, P.M. and Tauxe, R.V. 1999 Food-related illness and death in the United States. Emerging Infectious Diseases 5(5):607-625.

Mgana, S. M. (2003). Towards sustainable and robust on-site domestic wastewater treatment for all citizens. Sub-Department of Environmental Technology. Wageningen, Wageningen University: 95.

Morgenroth, E., R. Kommedal, et al. (2002). “Processes and modelling of hydrolysis of particulate organic matter in aerobic wastewater treatment - a review.” Water Science and Technology 45(6): 25-40.

Mufson, M.A. and Belshe, R.B. 1976. A review of adenovirus in the etiology of acute hemorrhagic cystitis.Journal of Urology 115(2):191-194.

Murray, P.R., Drew, L.W., Kobayashi, G.S. and Thompson, J.H. (1990) Medical Microbiology. Wolfe Publishing, Ltd., USA

NIPO/VEVIN (2002) Milieu en natuur compendium http://www.rivm.nl/milieuennatuurcompendium/nl/i-nl-0037-03.html

Ollinger-Snyder, P. and Matthews, M.E. 1996. Food safety: Review and implications for dietitians and dietetic technicians. Journal of the American Dietetic Association 96:163-168,171.

Otterpohl, R. (2000). Design of highly efficient source control sanitation and practical experiences. IWA publishing, ISBN 1 900222 47 7.

Palmquist, H. and Jönsson, H (2004) Urine, faeces, greywater, and biodegradable solid waste as potential fertilisers. Proceedings of the 2nd International Symposium, 7th – 11th April 2003, Lübeck, Germany

Pavlostathis, T.S. and Giraldo-Gomez, E. (1991). Kinetics of anaerobic treatment. Water Science and Technology 24, 35-59.

Reith, J.H., Wejffels, R.H. and Barten, H. (2003) Bio-methane & bio-hydrogen. Status and perspectives of biological methane and hydrogen production. Dutch Biological Hydrogen Foundation.

Reuschenbach, P., U. Pagga, et al. (2003). "A critical comparison of respirometric biodegradation tests based on OECD 301 and related test methods." Water Research 37: 1571-1582.

Rioned (2002) Stichting Rioned. Het riool in cijfers. http://www.riool.net/rioned/binary

Roediger Vacuum + Haustechnik (2000) Roevac vacuum technology, the low cost option. Vacuum sewer system Roevac (outdoor) for the collection of waste-water. Comercial brochure

Sanders, W.T.M. (2001). Anaerobic hydrolysis during digestion of complex substrates, PhD thesis Wageningen University, Wageningen.

Schönning, C. (2001) Evaluation of microbial health risks associated with the reuse of source separated human urine. PhD thesis, Department of Biotechnology, Royal Institute of Technology, Stockholm, Sweden. ISBN 91-7283-039-5.

Seghezzo, L., G. Zeeman, et al. (1998). "A review: the anaerobic treatment of sewage in UASB and EGSB reactors." Bioresourse Technology 65: 175-190.

SMI (2000) Infectious Diseases 1999. (Smittsamma sjukdomar 1999). Swedish Institute for Infectious Disease Control, Stockholm, Sweden. (In Swedish).

Snyman, H.G. (2001) Characterization of sewage sludge metals for classification purposes using the potential leachable metal fraction. Wat.Sci.Techn. Vol. 44, No. 10 pp 107-114, 2001

Steen, I. (1998) Phosphorus availability in the 21st century. Management of a non-renewable respurces. Phosphorus and potassium, issue No 217.

STOWA (2001) Separate urine collection and treatment. Options for sustainable wastewater systems and mineral recovery, 2001-39.

Swedenviro report (2001) Market survey – extremely low flush toilets Plus urine diverting toilets and urinals for collecting of black water and/or urine, 2001:1

Tauxe, R.V. and Cohen, M.L. (1995). Epidemiology of diarrheal diseases in developed countries. In: Blaser, M.J., Smith, P.D., Ravdin, J.I. and Greenberg, H.B. (eds.) Infections of the gastrointestinal tract. Raven Press, Ltd., New York, NY, USA. p. 37-51.Tong and Lam, 2000

Tortora, G.J., Funke, B.R. and Case, C.L. (1992) Microbiology: an Introduction. The Benjamin/Cummings Publishing Company, Inc., Redwood City, California, USA.

Trémolières J., Bonfilis S., Carré L. & Sautier C. (1961). Une méthode d'étude de la digistibilité chez l'homme, le fécalogramme. Nutritio et Dieta; European Review of Nutrition and Dietetics 3, pp. 281-289.

UNEP (2000) International Source Book on Environmentally Sound Technologied for Wastewater and Stormwater Management. United Nationds. International Environmental Technology Center, Osaka, Japan.

Velsen, van, A.F.M. (1981) Anaerobic digestion of piggery waste. Ph.D. thesis. Wageningen University, The Netherlands.

Vinnerås B., Höglund C., Jönsson H. & Stenström T.A. (1999). Characterisation of sludge in urine separating sewerage systems. In: Klöve B., Etniner C., Jenssen P. & Maehlum T.(eds) Proceedings of the 4th International Conference – Managing the Wastewater Resource Ecological Engineering for Wastewater Treatment. Norway, June 7-11, 1999.

Vinnerås, B. (2002) Possibilities for Sustainable Nutrient Recycling by Faecal Separation Combined with Urine Diversion PhD thesis, Agraria 353, Department of Agricultural Enginering, Swedish University of Agricultural Sciences.

Veeken, A. and Hamelers, B. (1999) Effect of temperature on hydrolysis rates of selected biowaste components. Bioresource Technology 69, 249-254.

Verkerk, G. (1986) Binas, information book for education in nature science.

Wang, K. (1994). Integrated anaerobic and aerobic treatment of sewage. Sub-Department of Environmental technology. Wageningen, Wageningen University

WHO (1989) Health guidelines for the use of wastewater in agriculture and aquaculture. Technical report series No. 778. Geneva, World Health Organisation.

Wellinger, A. and Kaufmann, R. (1982) Biogasproduktion aus Schweingülle in nicht beheitzen Anlagen. Blätter für Landtechnik, 198: 1-12 .

Wheeler, J.G., Sethi, D., Cowden, J.M., Wall, P.G., Rodrigues, L.C., Tompkins, D.S., Hudson, M.J. and Roderick, P.J. 1999. Study of infectious intestinal disease in England: Rates in the community, presenting to general practice, and reported to national surveillance. British Medical Journal 318:1046-1050.

Wijst, M. van der Groot-Marcus, A.P. 1998. Huishoudelijk afvalwater: Berekening van de zuurstofvraag, Household and consumer studies onderzoeksrapport 1, Wageningen Agricultural University (STOWA 98-40)

Wiegant, W. M. (2001). "Experiences and potential of anaerobic wastewater treatment in tropical regions." Water Science and Technology 44(8): 107-113.

Wilderer, P. A. (2001). Decentralized versus centralized wastewater management. Decentralised Sanitation and Reuse; Concepts, systems and implementation. P. Lens, G. Zeeman and G. Lettinga. London, IWA Publishing.

WRS, Water Revival Systems Uppsala AB (2001) Market Survey – extremely low flush toilets. Plus urine diverting toilets and urinals, for collection of black water and/.or urine. Swedenviro 2001:1.

WUR-MT (2001) Wageningen Universiteit, Milieutechnologie, Projectplan DESAR-PHASE 1. Decentrale Sanitatie en Hergebruik. EET project, Wageningen University

Zeeman G. (1991) Mesophilic and psychrophilic digestion of liquid manure. PhD thesis Wageningen University, The Netherlands.

Zeeman, G. and Lettinga, G. (1999). The role of anaerobic digestion of domestic sewage in closing the water and nutrient cycle at community level. Water Science and Technology, 39 (5), 187-194.

Zeeman, G., Sanders, W. and Lettinga, G. (2000). Feasibility of the on-site treatment of sewage and swill in large buildings. Water Science and Technology, 41 (1), 9-16.

APPENDICES

APPENDIX 1

LIST OF COMMONLY FOUND PATHOGENS IN EXCRETED URINE

(ADAPTED FROM HÖGLUND, 2002)

Organisms	**Source**	**Infection risks, transmition routes**
Leptospira interrogans	Feachem et al, 1983; CDC2000a	Bacterial infection with influenza like symptoms, transmitted by urine from infected animals; occupational hazard for WWTP workers and farm workers in developing countries. Human urine not significant transmission route since the prevalence of infection rather low
Salmonella typhi Salmonella paratyphi	Lewis-Jones and Winkler 1991,	Infection risks only during the epidemy of typhoid and paratyphoid fevers when bacteria proliferate in the blood – situation not common for developed country. Even during epidemy urine-oral transmission unusual in comparison with faecal-oral transmission.
Schistosoma haematobium	CDCb	Schistosomiasis, major human parasitic infection mainly occurring in Africa. Eggs are excreted in urine, sometimes during the whole life of host. The eggs released to environment and the larvae infect aquatic snail species living in fresh water. After several development stages, larvae emerges the snail and may infect human through penetration of the skin (not in Europe or US)
Mycobacterium tuberculosis Mycobacterium bovis	Bentz et al., 1975, Grange and Yates 1992, Dailloux et al., 1999, Huitema 1969	Tuberculosis mainly transmitted by air. Exceptionally isolated in nature but it was found in wastewater from hospitals. Humans may infect cattle (reported).
Microsporidia (protozoa group)	Marshall et al., 1997, Haas et al., 1999	Mainly in HIV-positive individuals. Urine is a possible transmission route. No water or foodborne outbreaks reported.
Cytomegalovirus (CMV) Polyomaviruses: JCV, BKV	Jawetz et al., 1987	Infection from person to person and not by food and water. CMV infects large populations (age of 40). Found in sewage.
Hepatitis A, B	Bofill-Mas et al., 2000 Ollinger-Snyder and Mathews 1996	One hepatitis A outbreak as result of eating lettuce contaminated by urine. Hepatitis B was also found in urine. Urine suggested as potential transmission route
Adenovirus	Mufson and Belshe 1976	Urine of children with hemorrhagic cystitis, transplant patients, HIV positive individuals. No health concern recognised.

APPENDIX 2

LIST OF COMMONLY FOUND PATHOGENS IN EXCRETED FAECES

(ADAPTED FROM HÖGLUND)

Organisms	**Source**	**Infection risks, transmition routes**
Campylobacter, Salmonella, Giardia,Yersinia,Giardia lamblia, Shigella, Entamoeba, enterohaemorrhagic *E.coli* O157 (EHEC O157) Aeromonas spp Cryptosporidium parvum Entamoeba histolytica	SMI 2000; Tauxe and Cohen 1995	diarrhoeal diseases
Protozoa: microsporidia, Cyclospora, Entamoeba histolytica C. parvum	Duncanson et al. 2000	GI infections, C. parvum: 2.5 and 24.2 cases per 100 000 protozoa like microsporidia and Cyclospora
Viruses: small round shape viruses (SRSVs, Norwalk-like virus), rotavirus, enteroviruses, reovirus, hepatitis Aenteric adenovirus, reoviruses and hepatitis A virus	Wheeler et al.1999 Mead et al. 1999 Tauxe and Cohen 1995 Feachem et al. 1983	GI infections More than 120 different types of viruses may be excreted in faeces; a major cause of gastrointestinal infections; enteric v. transmission from person to person (zoonotic transmission not proved neither excluded)
Human helmints:	Feachem et al. 1983; Gopinath and Keystone 1995	Mainly in developing countries major cause of morbidity and mortality;
Ascariasis (most common helminthic infections) Fertilised eggs		Most common, one billion cases Develop in soil, causes disease by ingestion
Hookworm (ancylostomiasis) Whipworm (trichriasis)		

APPENDIX 3

EFFECT OF CHOSEN METAL EXPOSURE ON/ TO HUMANS

Wastewater and sludge can contain different species of metals, some of which are needed for growth by plants and animals (Ca, Mg), some of which are needed but in smaller quantities (Cu, Fe, Ni and Zn) and some of which are not essential for growth (Pb, Cd, Hg). Too high concentrations of specific metals can have toxic effects, also in humans. Metals of serious concern are Cd, Hg and Pb. Also Ni Cr and Zn can cause human toxicity. The effects of metals on human health are listed in Table. Mentioned there effects are dose-dependant. A high exposure can be reached by ingesting a high concentration in a short time, but also by ingesting a low concentration during a chronic exposure.

When heavy metals are brought into the soil through the use of contaminated sludge or wastewater, three scenario's are possible: 1) metals accumulate in the soil, 2) metals are taken up by crops and accumulate in the crop, 3) metals are leaching into the groundwater.
The second and third scenario are most likely to impose risks for human health, because there is a reasonable chance that people will eat contaminated crops or that contaminated groundwater is used as drinking water source. The chance of people other than agricultural workers to contact contaminated soil is neglectable, unless land use will change. Whether metals are taken up by crops and in which concentrations depends on their chemical structure. Metals bound to organic and mineral matter in the soil are not easily available for plants. On the other hand metals in soluble form have a high plant-availability and will be taken up quickly. Some metals are taken up more easily as they are similar in size and chemical form to essential metals. For example chromium, cadmium and manganese compete with iron. Latter can also be the cause of iron-deficiency in humans (Vinnerås, 2002).
Since the most significant exposure route for metals is food chain transfer (Asano,1998, Alvarez *et al*, 2002), bioavailability is an important factor in health risk characterization. It is important to know to what extent metals are taken up by crops, and transferred to humans. In a study on bioavailability of different metals brought into soil, Alvarez *et al* (2002) found that Zn and Mn were bioavailable, while Cd, Mo and Ti were bound to soil particles and not bioavailable. In Table four general groups of metals are distinguished.

METAL CLASSIFICATION IN RELATION TO THEIR EFFECT ON CROPS AND PLANTS

Group	Metals	Characteristics
1	Ag, Cr, Sn, Ti, Y, Zr	low solubility in soil and therefore are not taken up by plants
2	As, Hg, Pb	strongly absorbed or adsorbed by soil colloids. They can be absorbed by plant roots, but do not reach the other plant tissues
3	B, Cu, Mn, Mo, Ni, Zn	easily taken up by plants, but that are toxic to the plant at concentrations that pose little risk to human health
4	Cd, Co, Mo, Se	easily taken up by plants, but can cause adverse human health effects at concentrations that are not toxic to the plant

Another study, on metal uptake by wheat plants after fertilizing with different types of biosolids, reports that the wheat uptake of only Cu and Zn had increased, while the concentration of Cu, Zn, Pb and Cr in the surface layer of the soil had increased after the application of biosolids on a field. None of the studied metals (Cu, Zn, Pb, Ni, Co, Cr and Cd) had leaked into the groundwater (Benitez *et al* , 2000). Snyman (2001) and Asano (1998) consider Cd the most serious contaminant because of its high toxicity, relatively high mobility in the soil and high risk of food chain transfer. However, Alvarez *et al* (2002) consider Cd to be safe regarding potential health effects, because they found its bioavailability to be low. Also Mo, Co, Ni and Ti were regarded as safe. According to McLaughlin *et al* (1999), Pb and Hg are not of serious risk either, because human intake is limited by soil retention and the “soil-plant barrier”.

Heavy metals in domestic blackwater usually originate from faeces. This is because most of the metals that are consumed by humans are not taken up, and directly leave the body via the faeces. Household cleaning water contains high concentrations of metals. If cleaning water is flushed through the toilet, it contributes greatly to the heavy metal content of black water (Vinnerås, 2002). Heavy metals tend to accumulate in biomass (Haytoglu *et al*, 2001) and therefore a higher metal concentration is to be expected in sludge. Several studies have been conducted on the heavy metal content of different wastewater streams. They are listed in appendix 4 together with guidelines when wastewater and sludge is to be used as fertilization on agricultural land. For comparison the guidelines for drinking water are included as well.

INTAKE WAY OF CRITICAL METALS AND THEIR EFFECTS ON HUMAN HEALTH

Metal	Way of uptake	effects
Cd	Ingestion	accumulate in kidney and liver where it causes renal dysfunction
	chronic exposure	nephropathies, emphysematous alterations in the lung and cardiovascular diseases human carcinogen neurotoxic (together with Pb)
	fumes inhaled	
Hg	long-term exposure	highly immunotoxic, irreversible brain damage, kidney damage and damage to the developing fetus neurotoxicity, especially in young children whose brain is growing at maximal speed
Pb	Chronic lead exposure even at low levels	various metabolic and cognitive disorders, decrements in school performance, learning disabilities, neuropsychological deficits, decreased intelligence, behavioural and developmental impairment, and disturbances of growth increased blood pressure and nephropathies
Ni	Accumulation	nickel allergy in the form of contact dermatitis lung fibrosis, cardiovascular and kidney diseases, and carcinogenic effects
Cr		allergic reactions on the skin
Zn	Short term exposure in water or beverages	Gastrointestinal distress, nausea, diarrhoea, occasionally leading to death (after the inadvertent intravenous administration of large doses of zinc); copper deficiency when disproportionate intake of zinc in relation to copper;

APPENDIX 4

Heavy metal concentrations of different wastewater components, guidelines found in literature and examples of standards for different disposal/management practices

Concentrations		Cd	Cr	Cu	Ni	Pb	Zn
Urine, separate collection (a)	mg.person/y	0.08-0.49	0.16-10.29*	17.2-1489.95*	4.2-32.69*	4.2-15.33**	107-150.01*
Faeces, separate collection (a)	mg/person/y	5.7-6.3	47-68.04	628-1059.66	81-109.62	13.0-459.9	4859.82-16940***
Biodegradable waste (b)	mg/person/y	2.5	182	162	89	40	675
Blackwater (c)	mg/person/d	0.013-0.025	0.05-0.08	1.4-3.3	0.2-0.27	0.05-0.08	11.9-15.0
Blackwater (d)	µg/L	0.171-0.513	<0.902-3.7	61.9-162	5.76-12.1	0.705-3.67	213-767
Urine, normal flush toilet (d) daily load	mg	0.002		0.05		0.03	0.5
Faeces, normal flush toilet (e) daily load	mg	0.03		2.0		0.3	11.0
Sludge (f)	mg/kg DM	<0.1-15	23.2-1868	97-1579	1.6-705	0.5-943	42.8-79.13
WWTP effluent (g)	mg/L	0.0002-0.0023	0.003-0.008	0.005-0.148	0.003-0.073	0.0004-0.022	0.017-0.571
Guidelines, standards, criteria for different waters and purposes							
Swedish designing urine (h)	mg/person/y	0.25	3.7	37	2.6	0.73	16.4
Swedish designing faeces (h)	mg/person/y	3.7	7.3	400	27	7.3	3900
Swedish designing biod. waste (h)	mg/person/y	2.7	137	549	82.3	275	700
Max pollutant concentration in soil (i)	mg/kg DW	7	3200		850	150	
Water agricultural irrigation (j)	mg/L	0.01	0.1	0.2	0.2	5	2
Drinking water (k)	mg/L	0.005	0.1	1.3		0.0	
Max loading with WWTP sludge meadows (l)	g/ha/y	1.25	75	75	30	100	300
Max loading with WWTP sludge, agriculture (l)	g/ha/y	2.5	150	150	60	200	600
WWTP sludge	mg/kg DW	1.25	75	75	30	100	300
Compost	mg/kg DW	1	50	60	20	100	200
Very clean compost	mg/kg DW	0.7	50	25	10	65	75

* probably due to misplaced domestic cleaning water and corrosion of pipe material; ** probably due to deposition of lead from propeller plane fuel used at the local airfield; *** probably due to contamination from zinc-galvanised pipes

References: a: (Vinnerås, 2002 ; Palmquist and Jonsson, 2001) ; b: (Palmquist and Jonsson, 2001); c: (Balmer, 2001) d: (Palmquist and Hanaeus, 2001) e: (Austrian data) f: (Snyman, 2001 ; Balmer, 2001; Aulicino et al, 1998; Keller et al, 2002) g: (Asano and Levine, 1998); h: (Vinnerås, 2002) i: (Asano and Levine, 1998) j: (Asano, H9) k: (EPA, 2002), l: BOOM (2002)

APPENDIX 5

BATCH TEST: MAXIMUM SUBSTRATE BIODEGRADABILITY

Literature prescribes a method for determining the biodegradability of a substrate, using inoculum for the substrate that is being tested (Angelidaki and Sanders, 2004). When a substrate contains a sufficient amount of biomass to ensure complete degradation of the biodegradable material inoculum is left out.

Serum bottles (500 ml) were filled with 400 ml substrate (influent, effluent or sludge) (tests were performed in duplo). The bottles were closed with a rubber septum and an aluminium cap and flushed with nitrogen for 10 minutes to create anaerobic conditions in the bottle. The bottles were stored in a 35 °C cabinet. When enough gas had been produced (resistance had been built up under the septum), the serum bottles were connected to a displacement bottle filled with NaOH by tubes and syrinx needles. The displaced amount of NaOH was measured in a graduated cylinder. The gas that enters the displacement bottle consists mainly of CH_4 and CO_2. As the NaOH absorbs CO_2, only CH_4 will cause a volume of NaOH solutionto leave the displacement bottle. The volumetric amount of methane is equal to the displaced volume of NaOH.

The measured volumetric amount of methane was calculated to the equivalent amount of COD. By using the ideal gas law $P * V = n * R * T$ and the fact that one mole of methane is equivalent to 64 g COD, equation (5.1) was derived.

$$COD - CH_4 = \frac{P \cdot V_{CH4}}{R \cdot T} * 64 \tag{5.1}$$

Where:

$COD\text{-}CH_4$ = Chemical Oxygen Demand of the produced volume of methane [g COD]

P = pressure [Pa] (at sea level 100 000 Pa)

V_{CH4} = cumulative volume of methane produced [m^3]

R = universal gas constant [Pa m^3 mol^{-1} K^{-1}] (value 8.3144)

T = temperature [K]

The biodegradability was then determined according to equation (5.2).

$$Bio\deg radability(\%) = \frac{COD - CH_{4(t=end)}}{COD_{tot(t=0)}} * 100 \tag{5.2}$$

After the experiment the serum bottle content was analysed for COD_{tot}, NH_4^+ - N, PO_4^{3-} - P, pH, and VFA. This was done amongst others to check on possible inhibition of methane production due to the accumulation of VFA or free ammonia. The COD_{tot} concentration was used to verify the biodegradability percentage according to equation 5.3.

$$Bio\deg radability(\%) = \frac{COD_{tot(t=0)} - COD_{tot(t=end)}}{COD_{tot(t=end)}} * 100 \tag{5.3}$$

APPENDIX 6

BATCH TEST: SLUDGE STABILITY

When sludge from anaerobic reactor is used in agriculture it should be digested in order to prevent further going mineralisation and bad odours. To determine the fraction of solids that are still degradable, stability tests are conducted.
Serum bottles (250 ml) are filled with 200 ml sludge (tests in duplo). The bottles were closed with a rubber septum and an aluminium cap and flushed with nitrogen (10 minutes) to induce anaerobic conditions. The bottles were stored in a 35ºC cabinet. When enough gas had been produced (resistance had been built up under the septum), the serum bottles were connected to a displacement bottle filled with NaOH by tubes and syrinx needles. The volumetric amount of methane was measured in a graduated cylinder.
The gas that enters the displacement bottle consists of methane and CO_2. As the NaOH absorbs CO_2, only methane will cause a volume of NaOH solution to leave the displacement bottle.

The stable fraction of the sludge is determined by equation 6.1

$$Stability(\%) = 100\% - \frac{COD - \breve{C}H_4}{COD_{tot}} \cdot 100 \tag{6.1}$$

After the experiment the sludge form the serum bottles was analysed for COD_{TOT}, NH_4^+ - N, PO_4^{3-} - P, pH, and VFA. This was done to check on possible inhibition of methane production due to for example the accumulation of VFA. The COD_{TOT} concentration was used to verify the stability percentage according to equation (6.2).

$$Stability(\%) = 100\% - \frac{COD_{tot,t=0} - COD_{tot,t=end}}{COD_{tot(t=0)}} \cdot 100 \tag{6.2}$$

Stability and digestibility are two related variables. Where stability is the fraction of sludge that can't be degraded anymore, digestibility is the fraction that remains degradable after storage in the $UASB_{ST}$.

APPENDIX 7

BATCH TEST: MAXIMUM SPECIFIC METHANOGENIC ACTIVITY

The specific methanogenic activity (SMA, gCH_4-COD.gVS^{-1}.d^{-1}) is the maximum potential of the biomass to produce methane (CH_4) from a certain substrate in a certain time interval. The SMA is high when a lot of methanogenic bacteria are present in the sludge and environmental conditions are favourable for them to digest their substrate.
The sludge was taken from a reactor and analyzed for VFA, NH_4^+-N, PO_4^{3-} - P, pH, TS, VS. Oxitop bottles were filled with 100 ml of sludge (tests in duplo). To prevent substrate inhibition, excess amount of acetate was added to the sludge. The bottles were flushed with nitrogen gas for 10 minutes to obtain an anaerobic condition. A rubber tool was installed containing NaOH crystals to capture CO_2 and measure exclusively CH_4 production. The bottles were closed with an Oxitop measuring head, which measures the pressure difference in the bottle. The bottles were stored in a cabinet at a temperature of 35ºC. During the measuring period, possible pH inhibition was monitored in an additional control bottle containing sludge and acetate not closed with an Oxitop head.
The pressure differences were recorded some, after which the heads were taken of and the sludge characterized again. Data were retrieved from the measuring heads using a remote control. It can show retrieved data (graphically) at all times. After the run the data were brought from the controller to a PC using an interface cable.
The calculation procedure is based on the ideal gas law:

$$P \cdot V = n \cdot R \cdot T \tag{7.1}$$

Where: P = gas pressure (Pa), V = volume of gas (m^3), n = the amount of mols of gas formed, R = universal gas constant (Pa m^3 mol^{-1} K^{-1}, value 8.3144), T=temperature (K).

The gradient of *n* is the parameter looked for. During the test the gradient of *P* is measured. Based on reaction 7.2 the production rate of methane (expressed in gCOD.d^{-1}) is calculated from equation 7.3 and the SMA from equation 7.4.

$$CH_4 + 2O_2 \rightarrow CO_2 + H_2O \tag{7.2}$$

$$production \quad rateCH_4 - COD = 64 \cdot \frac{V_{gasbottle}}{R \cdot T} \cdot \Delta P \tag{7.3}$$

$$SMA = \frac{production\ rate \quad CH_4 - COD}{VS_{sludge \quad bottle}} \tag{7.4}$$

APPENDIX 8

BATCH TEST: DETERMINATION OF HYDROLYSIS CONSTANT

The knowledge on hydrolysis constant enables process design and modelling. The substrate hydrolysis rate (hydrolysis constant) during anaerobic digestion of concentrated black water and kitchen waste was determined at a range of temperatures. The experiment is carried out in bottles under methanogenic conditions.
Two types of serum bottles are used, bottle for sampling and determination of soluble compounds and bottle for biogas production measurement. They contained mixtures of substrate, tap water and inoculum. In addition bottles containing only inoculum and tap water were used as control for the k_H values from hydrolysis of fractions in the inoculum.

The required volume of inoculum was estimated based on formula:

$$V_{inoculum} = \frac{X_{ss}.V_{ww}.\,k_H}{VS_{inoculum}.\,SMA_{inoculum}} \quad ,\ \mathrm{l} \tag{8.1}$$

where:

X_{SS} = concentration of hydrolysable substrate ($gCOD.l^{-1}$)
V_{ww} = volume of wastewater in the bottles (l)
k_H = hydrolysis constant (d^{-1})
$V_{inoculum}$ = volume of methanogenic inoculum (l)
$VS_{inoculum}$ = volatile solids content of the inoculum ($gVS.l^{-1}$)
$SMA_{inoculum}$ = methanogenic activity of the inoculum ($gCOD.gVS^{-1}.d^{-1}$)

Total COD of the sample was measured at the beginning of the experiment. To monitor the hydrolysis process during the experiment, measurement of the $COD_{soluble}$, VFA, $NH_4^+ - N$ and $PO_4 - P$ from each substrate bottles were conducted. The measurements were done by taking 5 ml sample from the soluble bottle using a syringe (bottle was not open) and distributed into three 1.5 ml eppendorf. The tubes then centrifuged for 10 minutes at 12 500 rpm. The supernatant was used for dilution and then the dilution was used for the determination of the parameters. The measurement in the first days was done more frequently since the concentration of biodegradable substrate is highest at the beginning of the experiment so does the hydrolysis. Afterwards the measurement was done once a day and then decreased to once a week. The experiment was run for 50 days and all the produced methane was followed through the biogas bottles using the sodium hydroxide (NaOH) displacement method.
Hydrolysis rate is assumed to be a first order constant accordimng to the following formula:

$$X_t = X_0 \cdot e^{-k_H \cdot t} \tag{8.2}$$

The biodegradable COD particulate at t=0 (X_0) can be described as:

$$X_0 = CH_{4,max} + COD_{sol(t=end)} - COD_{sol(t=0)}$$

The biodegradable COD particulate at t=t (X_t) can be described as:

$$X_t = X_0 - (CH_{4,\ at\ t=t} + COD_{sol(t=t)} - COD_{sol(t=0)})$$

The k_h is the natural logarithm of the ratio between X_t and X_0. Based on the linear part of the gas production curve, and covering at least 70% of the total gas production, the natural logarithm of (X_t/X_0) is plotted against time. The hydrolysis constant is calculated based on the linear trend line through the plotted points.

APPENDIX 9

ANALYTICAL METHODS

COD FRACTIONS (DR. LANGE METHOD)

Influent and effluent samples of the reactors were analyzed for COD fractions (COD_{total}, $COD_{particulate}$, $COD_{colloidal}$ and $COD_{soluble}$) using Dr. Lange method according to NEN 6633.

COD_{total} : Raw samples were disintegrated for 1 minute. After that, influent samples were diluted 20 times and effluent samples were diluted 10 times. Dr. Lange tubes, type 514, were used. After sample preparation COD was measured spectrophotometically with use of a Xion 500 spectrophotometer.
$COD_{particulate}$: Raw samples were filtered through a 125 mm Φ, 4.4 μm pore size paper filter, diluted, and measured using Dr. Lange. The measured value was noted as $COD_{filtered}$.
COD_{part} was calculated as COD_{total} minus $COD_{filtered}$.
$COD_{soluble}$: Filtered samples (125 mm Φ, 4.4 μm pore size paper filter) were again filtered through a 50 mm Φ, 0.45 μm pore size membrane filter, diluted, prepared and measured.
$COD_{colloidal}$: $COD_{colloidal}$ was calculated as $COD_{filtered}$ minus $COD_{soluble}$.

Volatile Fatty Acids (VFA)
Samples were filtered through a 125 mm Φ, 4.4 μm pore size paper filter, and after that filtered again through a 50 mm Φ, 0.45 μm pore size membrane filter. Filtered samples were diluted. 1 ml sample together with 0.1 ml of a 50% formic acid was pipetted into a 2 ml glass flask to be analyzed by GC. The chromatograph (Hewlett Packard, model 5890A, Palo Alto, USA) was equipped with a 2 x 2 mm (inner diameter) glass column, packed with Supelco port (100-120 mesh) and coated with 105 Fluorad FC 431. Operating conditions were: column, 130 ºC; injection port, 200 ºC; flame ionisation detector, 280 ºC. Nitrogen (N2) saturated with formic acid at 20 ºC was used as a carrier gas (30 ml/min). Acetate, propionate, butyrate (n/i) and pentate (b/n) were determined in mg COD/litre.

Total Kjeldahl-N
The total Kjeldahl-N was determined according to a titrimetric method (*Standard Methods for the Examination of Water and Wastewater, 20th Edition*.

Ammonium (NH4-N)
Diluted influent and effluent samples were analysed photometrically for NH_4-N using an auto-analyzer (SKALAR SA-1050) according to NEN 6472, which describes a photometric method.

Total Phosphate
Destructed samples were analysed for PO_4-P with an auto-analyzer (SKALAR SA-1050) according to NEN 6472, which describes a photometric method.

Soluble Phosphate (PO_4-P)
Soluble phosphate in diluted samples were determined photometrically on an auto-analyzer (SKALAR SA-1050) according to NEN 6472.

Pathogens (indicator organisms)

E.coli were determined according to the method described by Havelaar *et al.* (1988) and Sharpe *et al.* (1981). Sample volumes (or dilutions in 0.1% peptone-saline) were filtered through 0.45 µm membrane filters. The membrane filters were placed in petri dishes containing tryptone soya agar and incubated at 37 ± 1 ˚C for 4 hours. After that the membranes were transferred to petri dishes containing tryptone bile agar and incubated at 44 ± 0.5 ˚C for 20 hours. After incubation the membranes were placed on filter papers saturated with indole reagent (0.5 g 4-dimethylaminobenzaldehyde in 100 ml 1 mol / 1 HCL) according to Sharpe *et al.* (1981) and placed under a UV lamp for 15 minutes. Colonies were scored as indole positive, negative or doubtful., A dish was accepted when the number of colonies on a dish was between 20 and 80.

Faecal coliforms were determined according to method 9211 B from *Standard Methods for the Examination of Water and Wastewater, 20th Edition*. Sample volumes (or dilutions in distilled water) were filtered through 0.45 µm membrane filters, placed on a M-7 h FC agar plate and incubated at 41.5 ˚C for 7 hours. Faecal coliform colonies are yellow (indicative of lactose fermentation).

Staphylococci were determined according to method 9213 B from *Standard Methods for the Examination of Water and Wastewater, 20th Edition*. Sample volumes (or dilutions in distilled water) were filtered through 0.45 µm membrane filters, placed on a Baird-Parker agar plate and incubated at 35 ± 0.5 ˚C for 48 hours. Staphylococci form slate gray to jet black, smooth, entire colonies.

Streptococci and Enterococci were determined according to method 9230 from *Standard Methods for the Examination of Water and Wastewater, 20th Edition*. Sample volumes (or dilutions in distilled water) were filtered through 0.45 µm membrane filters. Membranes were transferred to petri dishes containing m Enterococcus agar and left standing for 30 minutes. After 30 minutes, plates were inverted and incubated at 35 ± 0.5 ˚C for 48 hours.
All light and dark red colonies were counted as enterococci.

Pseudomonas aeruginosa were determined according to method 9213 E from *Standard Methods for the Examination of Water and Wastewater, 20th Edition*. Sample volumes (or dilutions in distilled water) were filtered through 0.45 µm membrane filters and placed on a M-PA agar plate. Plates were inverted and incubated at 41.5± 0.5 ˚C for 72 hours.
Typical colonies are 0.8 to 2.2 mm in diameter and flat in appearance with light outer rims and brownish to greenish-black centers.

Ascaris suum eggs were determined according to the method described by De Victorica and Galván (2003). A sample volume of 1,000 ml was centrifuged at 480 x g for 5 minutes, supernatants were discarded and sediments were collected in a 50 ml centrifuge tube.
Samples were washed 2-3 times with 0.85% saline solution, centrifuging at 480 x g for 5 minutes each time. Final supernatant was discarded and eggs in the sediment were recovered by floating with ZnSO4 (ρ 1.2) and centrifuging at 480 x g for 5 minutes. Floated eggs were poured through a 25 mm Φ, 8.0 µm pore size membrane filter (using a plastic syringe), using gentle pressure. Then 1-3 ml of a 0.1% solution of vital stain was added and filtered as above. The filter was mounted on a clean glass slide and was allowed to dry for a few minutes. After that, the filter was cleared with 1 drop of glycerol, was allowed to stand for a few minutes

and covered (preventing bubbles). Eggs were counted under the microscope, using 10x and 40x objectives. Stained eggs were counted as non-viable, whereas unstained eggs were counted as viable.

Heavy metals

Heavy metal (Cd, Cu, Cr, Mg, Ni, Pb, and Zn) and also Ca and K concentrations were determined by ICP. Samples had to be prepared for measuring with ICP by using microwave destruction. The microwave used was Milestone Ethos E. Five ml of sample was pipetted into a special microwave-vessel and weighed. The sample with the highest organic content was pipetted into a special vessel with temperature sensor. Then 10 ml of acid, consisting of 7.5 ml HCl (37%) and 2.5 ml HNO_3, was added. The vessels were put in protection shields and placed in the microwave. After destruction and cooling down, the samples were quantitatively transferred to a 25 ml flask and filled up to 25 ml with demi-water and transferred to a measurement tube, to be analysed with ICP.

Types and frequencies of analyses for UASB reactors' performance

	Samples of UASB-1 and UASB-2			
	Influent	**Effluent**	**Reactor profile**	**Sludge**
Analyses				
PH	Two times	Two times	-	-
COD fractions	Weekly	Weekly	-	-
VFA	Weekly	Weekly	-	-
TS	Obtained from parallel experiment	Obtained from parallel experiment	Obtained from parallel experiment	Obtained from parallel experiment
VS	Obtained from parallel experiment	Obtained from parallel experiment	Obtained from parallel experiment	Obtained from parallel experiment
N Kjeldahl	Weekly	Weekly	-	-
NH_4-N	Weekly	Weekly	-	-
Total-P	Weekly	Weekly	-	-
PO4-P	Weekly	Weekly	-	-
K	Once	Once	Once	Once
E. Coli *and* Faecal Coliform	Three times	Seven times	-	-
Heavy metals (Ca, Cd, Cu, Cr, Mg, Ni, Pb, and Zn)	Once	Once	Once	Once

Types and frequencies of analyses for AC reactors' performance

	Samples of AC		
	Influent	**Reactor content**	**Sludge**
Analyses			
PH	frequently	frequently-	-
COD fractions	Weekly	-	-
VFA	Weekly	-	-
TS	Obtained from parallel experiment	Obtained from parallel experiment	Obtained from parallel experiment
VS	Obtained from parallel experiment	Obtained from parallel experiment	Obtained from parallel experiment
N_{Kj}	Weekly	-	-
N_{NH4}	Weekly	-	-
P_{tot}	Weekly	-	-
P_{PO4}	Weekly	-	-
K	Once	Once	Once
E. Coli and Faecal Coliform	Nine times	Eight times	Two times
Heavy metals (Ca, Cd, Cu, Cr, Mg, Ni, Pb, and Zn)	Once	Once	Once