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Options for reuse of nutrients from waste water in the Mekong Delta, Vietnam

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Abbreviations and Technical Terms

#	Number
α	Level of significance
a	Year, from Latin annus
AB	Experimental site An Binh, District of Can Tho City (and/or former Can Tho Province)
Active composting	composting with regular turning of the pile or heap, additional air supply
ADB	Asian Development Bank
AF	Anaerobic Filter
ANOVA	Statistical test for comparison of means
ANZECC	Australian and New Zealand Environment and Conservation Council
ASEAN	Association of South East Asian Nations
AusAID	Australian Agency of International Development
Autotroph	Organism that uses direct sources of energy
B23	Name of student dormitory at Can Tho University where separation system was installed
Biosolids	wastewater treatment plant sludge
Black water	Toilet water: Faeces, urine and flushing water
BOD	Biochemical oxygen demand is the amount of oxygen required for complete (aerobic) biological decomposition of a material. It is a measure of biodegradability of liquid streams
Brown water	Faeces and flushing water
BS	Biogas Sludge
BV	Vermicompost out of biogas sludge
Cap	Capita, per head
CEC	Cation exchange capacity, unit: mmol l^{-1} or %
CERPAD	Centre for Residential Planning and Development
CERWASS	Centre for Rural Water Supply and Environmental Sanitation
CFU	Colony forming unit, plate or direct counts (CFU g^{-1} or CFU ml^{-1}): amount of viable microorganisms that grow to colonies on an agar plate
CM	Cow Manure
CMEA	Council for Mutual Economic Assistance (also COMECON or MEA)
CoA	College of Agriculture, Can Tho University
Cocoon	Egg case where fully developed earthworms emerge
COD	Chemical oxygen demand, necessary oxygen amount for oxidising all organic material present in a sample (by Potassium-di-chromate)
CoT	College of Technology, Can Tho University
CPVN	Communist Party of Vietnam
CSRR	Continuously stirred-tank reactor
CTU	Can Tho University
CV	Vermicompost of cow manure
CV	Coefficient of variation, equals the standard deviation divided by the mean, expressed as a percent
d	Day
DANIDA	Danish International Development Assistance
DARD	Department of Agriculture and Rural Development
$\Delta\Gamma$	Gibbs free energy (IUPAC recommended name: Gibbs energy or Gibbs function, originally called available energy), thermodynamic potential that measures the work obtainable from a system
DM	Dry matter
DOLISA	Department of Labour, Invalids and Social Affairs
DOST(E)	Department of Science and Technology (formerly: „..and Environment“)

DVC	Double Vault Composting (latrine)
E. coli	Escheria coli, bacteria used as indicator organism for faecal contamination
Excrements, excreta	Urine and faeces
FAO	Food and Agricultural Organization of the UN
FD	Fixed-dome reactor
FD-10	10 farms with FD-digester within the screening campaign in Long Tuyen, An Binh, My Khanh
FD-LT	Fixed-dome digester in Long Tuyen, part of the monitoring campaign
FYM	Farm Yard Manure
GDP	Gross Domestic Product
GoV	Government of Viet Nam
Grey water	Water from showers and sinks
GSO	General Statistical Office
HCMC	Ho Chi Minh City
Heterotrophic	Organism that uses organic substrates to get its chemical energy for its life cycle (all animals, fungi and many bacteria)
HRT	Hydraulic retention time, ratio of net volume of fermenter and amount of substrate
Hygienisation	Any process of disinfection to achieve a desired hygienic quality of a material (esp. in waste water treatment)
ICLARM	The International Center for Living Aquatic Resources Management
IPE	Institute of Plant Nutrition, Bonn University
IRC	International Water and Sanitation Centre
IWC	Influent wastewater concentration
KfW	Kreditanstalt für Wiederaufbau, German bank
LT	Experimental site Long Tuyen, District of Can Tho City (and/or former Can Tho Province)
LU	Livestock unit (equals 1 cattle of 500 kg)
MARD	Ministry of Agriculture and Rural Development
MDG	Millennium Development Goal
m_N^3, Nm^3	Normed gas volume at T=0°C, p=1013 mbar and free of water
MOC	Vietnamese Ministry of Construction
MOH	Vietnamese Ministry of Health
MOLISA	Vietnamese Ministry of Labour, Invalids and Social Affairs
MOSTE	Vietnamese Ministry of Science Technology and Environment
MPI	Vietnamese Ministry of Planning and Investment
MPN	Most Probable Number, MPN values are estimates (statistical in nature) by serial dilutions of the sample prepared and inoculated to determine the no. of organisms in the sample
MRC	Mekong River Commission
NGO	Non Governmental Organisation
Night soil	human excrements (faeces and urine)
NRWSS	National Rural Clean Water Supply and Sanitation Strategy up to Year 2020
N_{tot}	Total N, consisting of nitrogen fractions Norg, NH_4, urea (measured by Kjehldahl) plus NO_3 ; in substrates where no nitrate is present, e.g. slurry just Kjeldahl-N
n.d.	Not determined
NTU	Nephelometric Turbidity Unit
oDM	Organic dry matter, difference of mineral dry matter (received after drying at 550°C) and dry matter (drying at 105°C), unit used for loading rate of biogas digesters
OECD	Organisation for Economic Cooperation and Development
OES	Occupational exposure standard
OLR	Organic loading rate (Biomethanation)
OM	Organic Matter
Passive composting	No changes, e.g. turning during the composting process

Pasteurisation	Reduction of the number of viable pathogens so they are unlikely to cause disease
PE	Person equivalent, used for calculations in wastewater treatment
People's committee	Parliament of Province, district, ward, etc.
pH	Potentia hydro genii, negative log_{10} of H^+ concentration
PM	Pig Manure
PT	Plastic-tube biogas digester
PT-10	10 farms with PT-digester within the screening campaign in Long Tuyen, An Binh, My Khanh
PT-AB	Plastic-tube digester in An Binh, part of the monitoring campaign
PV	Vermicompost of pig manure
r^2	Correlation coefficent
RWSS	Rural Clean Water Supply and Sanitation Strategy (see also NRWSS)
SEARAV	Southeast Asian Research Association of Vietnam
SIWRP	Southern Institute of Water Resources and Planning (Vietnamese government agency in HCMC, in charge of water use planning in the Mekong Delta)
SNV	NGO from Netherlands, engaged in biogas technology, water treatment and others
SOFRI	Southern Fruit Research Institute of Vietnam
SRT	Sludge retention time
Sterilisation	any process that effectively kills or eliminates transmissible agents such as bacteria, fungi, viruses, spores, etc.; can be achieved through application of heat, chemicals, irradiation, high pressure or filtration
STP	Standard temperature and pressure; Volume at 0°C and 1 bar pressure, for calculations biogas
TCVN	Technical committee of Vietnam (Vietnamese institution responsible for water norms)
TDS	Total dissolved solids
(Compost) Tea	Aqueous extract of composts and vermicomposts, used by organic farmers to promote plant growth or suppress plant diseases
TKN	Total Kjeldahl nitrogen
TOC	Total organic carbon
TS	Total solids; sum of VS and ash components
TSS	Total suspended solids
UASB	Up-flow anaerobic sludge blanket reactor type
UN	United Nations
UNDP	United Nations Development Programme
VAC / VACB	Vietnamese Integrated farming system combining garden (vườn), aquaculture (ao) and livestock (chuồng) / VAC and biogas
Vacvina	Vietnamese NGO, engaged in biogas technology, agro-forestry and others
VBARD	Vietnam Bank for Agriculture and Rural Development
VFA	Volatile fatty acids, control parameter in biogas processes, produced within anaerobic degradation
VLSS	Viet Nam Household Living Standard Survey
VND	Vietnamese Dong, 20,000 VND ≈ 1 €
VNHS	Vietnamese National Health Study
VS	Volatile solids, difference between total solids (TS, drying at 105°C) and mineral solids (burning at 550°C for 2 h), equals organic dry matter; usually as percent of total solids (TS)
WATSAN	Water and Sanitation Program (Unicef and CERWASS)
WHO	World Health Organization
WRB	World Reference Base for Soil Resources, by FAO and UNESCO
WRI	World Resources Institute
WSP	Water and Sanitation Program for East Asia and the Pacific
WSS / WSSCC	Water Supply and Sanitation / WSS Collaborative Council
WTO	World Trade Organisation

1. Introduction

1.1. Motivation

Water resources are threatened by high consumption and increasing water pollution, due to growing population and "industrialisation" especially in fast developing countries like Vietnam.

Water is a precious resource and essential for living on earth. The importance is reflected in the proclamation of the International Decade for Action 'Water for Life' by the United Nations General Assembly for 2005 to 2015.
According to estimations of the WHO, 80% of all sickness in the world is attributable to waterborne disease. Each year there are approximately 4 billion cases of diarrhoea worldwide, most of them attributed to unsafe water supply, inadequate sanitation and hygiene (WHO 2007). Five million people die annually from water-borne diseases (Ongley 1996), 1.8 million deaths occur due to diarrhoeal diseases mostly in children under 5 years of age (WHO 2000, 2007). Infections are widespread throughout the developing world, in Southeast Asia, diarrhoea is responsible for 8.5% of all deaths, and for 4% worldwide (WHO 2000). This brings diarrhoeal diseases to the 5th most frequent reason for death ("Leading causes of death" (WHO 2008), more than one billion people do not have access to safe drinking water (EcosanRes 2008b).

Only 1.1 billion people are served by water-based sewage systems, 2.8 billon people are using pit latrines. 2.6 billion people lack basic sanitation, the majority of them being Asians (EcosanRes 2008b). Basic sanitation (as defined by WHO & UNICEF) is the lowest-cost technology ensuring hygienic excreta and sullage disposal and a clean and healthful living environment including safety and privacy in the use of these services. Improved sanitation facilities are public sewer connection; septic system connection; pour-flush latrine; simple pit latrine; ventilated improved pit latrine (WHO 2009).

The 2004 global image (Figure 1-1) shows that sanitation is lacking for large percent of the population in numerous contries, especially in sub-Saharan African, and numerous Asian countries (WHO & UNICEF 2006).

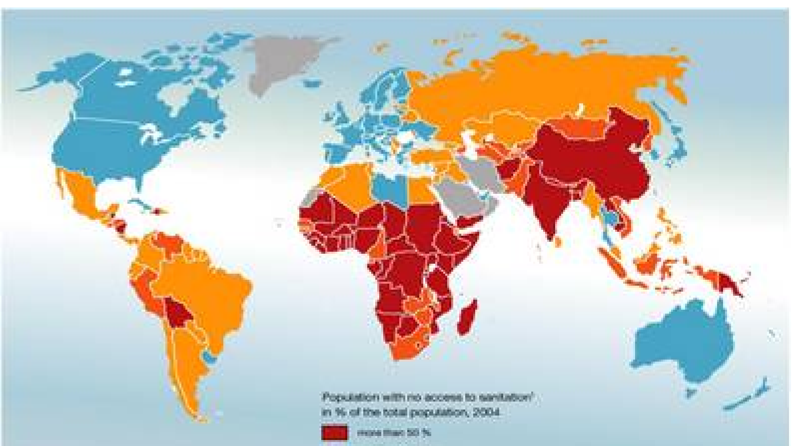

Figure 1-1: Population with no access to sanitation per country (WHO & UNICEF 2006)

In most developing countries less than 10% of waste water undergoes any treatment at all. Only ~40% of urban waste water receives treatment, in rural areas - areas characterized by farms, small towns, and unpopulated regions - it is a lot less (Hoek 2002). Figure 1-2 is the world map showing the percentage of citizens living in an urban environment in 2005. Vietnam belongs to the category with less than 35% living in urban environments.

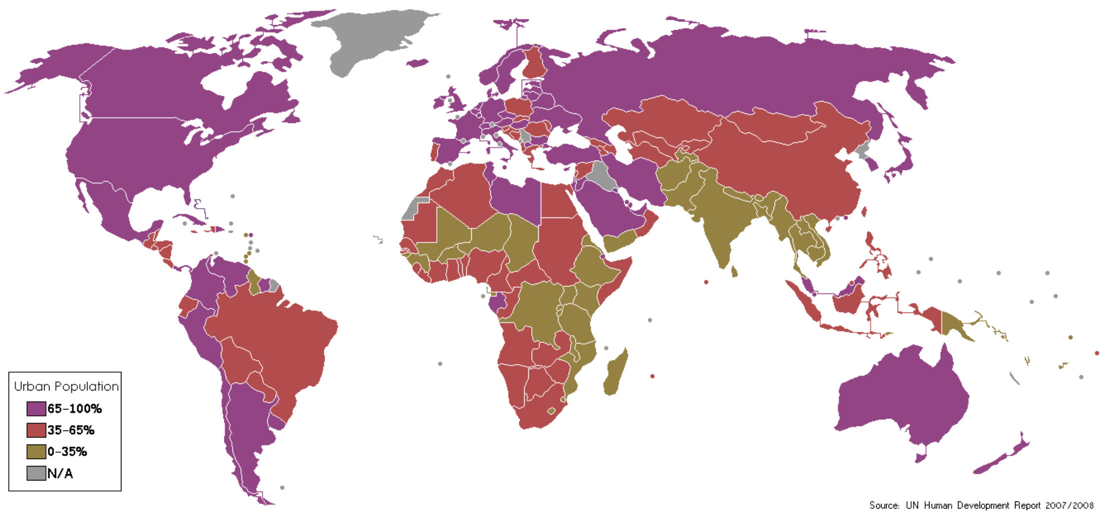

Figure 1-2: Population living in urban environments (WHO & UNICEF 2006)

Due to World Bank data, the priority of investment in the water sector in developing countries is water supply first, then sewerage and only then waste water treatment (Libhaber 2006).

Within the last 100 years, less than one third of the world population has been equipped with the "western type" of sanitation and waste water treatment (flush-toilets, sewer, and waste water treatment plant). High cost for construction and operation, lack of knowledge and a long construction process will not allow to apply this solution to all nations, and surely not until 2015 when the Millennium Development Goals are to be fulfilled (UNDP 2000). Cheaper and faster solutions are required to "reduce by half the proportion of people without sustainable access to safe drinking water and basic sanitation" (Target 7c).

Besides the endeavour for improving water resources, a second, most important aim has to be fulfilled by mankind: sufficient food for an increasing world population.

Increasing fertiliser demand, depletion of natural resources (e.g. phosphorus and potassium) combined with dramatically increasing prices are causing problems. Due to highest population growth rates in Asia and declining resources this area might become a "hot spot" – the risk of future escalations is more probable here than elsewhere.

Management of resources will become one of the most important issues in the world. (Molden & Billharz 1997). The UNESCO announced the years 2005-2014 as the Decade for education for sustainable development.

Worldwide there is a amount of nutrients from excrements already which will increase due to the world population´s development (Table 1-1). At the moment, the phosphorus in human excrements equals about 20-25% of P-fertiliser annually sold (Bundesanstalt für Geowissenschaften und Rohstoffe 2007; Steen 1998).

Table 1-1: P amount in human excrements worldwide (own calculation)

	2007[1,2]	2025	2050
Population	6,600,000,000	8,000,000,000	9,200,000,000
P exretion small (1.23 g pers^{-1} d^{-1})[3]	450	450	450
P excretion high (1.6 g pers^{-1} d^{-1})[4,5]	584	584	584
P small (t P a^{-1})	2,970,000	3,600,000	4,140,000
P high (t P a^{-1})	3,854,400	4,672,000	5,372,800

[1] (Wikipedia 2007), [2] (United Nations 2007) , [3] (Jönnson & Vinneras 2003), [4] (Metzner 2006), [5] (Gethke 2006)

By combining waste water treatment with recycling of nutrients for agriculture, several problems can be addressed:

- Pollution of natural waters can be reduced, thereby improving health conditions.
- Nutrient recycling contributes to reduce the world´s phosphorus depletion (Larsen *et al.* 2007; Tiessen 1995; Werner 2003) which is predicted in 60-90 years (Steen 1998; Tiessen 1995).
- Recycled nutrients may be an additional nutrient source especially in developing countries and in times of increasing fertiliser prices.

In this study it was examinined if and to what extent, the combination of wastewater treatment and recycling is feasible in the Mekong Delta, an Asian agriculturally distinct country.

1.2. Workflow

To suggest suitable solutions for the Mekong Delta area, how to treat waste water and recycle nutrients, the following steps were carried out (Table 1-2):

1. Characterising the situation and the threatening problems regarding water, sanitation and agriculture in the Mekong Delta. The general situation and the main issues regarding water and agriculture have to be understood for finding suitable solutions.
2. Evaluation of appropriate treatment techniques suitable for the local conditions and analysis of selected systems.
3. Balancing the recycling potential for the main nutrients, determine the conditions and limitations of the selected systems and give recommendations to select technologies according to the site and the specific situation.

Table 1-2: Experimental approach to identify appropriate technologies

Steps of the experimental approach
Status-Quo of technical, socio-economic conditions
Selection of treatment options and representative sites
Experimental phase: analysis and evaluating systems and products
Discussion with expert companies and scientists
Evaluating answers, creating database, giving recommendations

2. Status Quo in the Mekong Delta and in Vietnam

2.1. Political, Sociological and Economic Frame

Vietnam is a densely-populated developing country in South-East Asia, with around 85 million inhabitants (2008), growing by more than one-and-a-half million people every year.

The "Socialist Republic of Vietnam" is a single-party state where the Communist Party plays the central role in all organs of government, politics and society. The prime minister of Vietnam, Nguyen Tan Dung is the head of government, with three deputy prime ministers and 26 ministries and commissions.

After the war and the reunification of North and South Vietnam in 1975, the centrally planned economy was marked by an extremely low level of development. Substantial progress was achieved after the Communist Party of Vietnam implemented "Đổi Mới" (renovation) in 1986 which was leading towards a market-oriented economy (CIA 2008). From 1997 to 2004, the Gross Domestic Product (GDP) growth averaged 6.8% per year even against the background of the Asian financial crisis. Vietnam's membership in the ASEAN Free Trade Area and entry into WTO in January 2007 has provided Vietnam an anchor to the global market.

Deep poverty, defined as the population living under $1 per day, has declined significantly to the rate of 14.8% (2007 est.) and is now smaller than that of China, India, and the Philippines (CIA 2008). According to the Human Development Index (HDI) of the UNEP measuring achievements in terms of life expectancy, educational attainment and adjusted real income, Vietnam is ranked no. 108 of 177 countries worldwide (UNDP 2005).

Agriculture is one of the main economic sectors besides oil and clothing, although its economic output is shrinking, from about 25% in 2000 to less than 20% in 2007. Nevertheless, agriculture is still the main sector for occupation (56% compared to 18.9% in industry and 25.5% in services in July 2005 (CIA 2008)).

Agricultural production in Vietnam is based on the Mekong Delta. As the major food surplus region in Vietnam it is contributing about 80% of Vietnam's rice exports and more than 60% of seafood exports. The delta is developing export-oriented production (MDEC 2009).

With more than 420 pers/km^2 the Mekong Delta is also one of the most densely populated areas in the world (MONRE *et al.* 2003): around 17 million of Vietnam's inhabitants are living in this region, about 80% of them in rural areas, in villages structured along the canals.

The Mekong Delta includes 12 of 58 Vietnamese provinces (Figure 2-1). The former Can Tho Province now consists of Kien Giang province and the municipality of Can Tho, at the same level of a province. The provinces are divided into districts and towns, further into communes and wards. People´s Committees are regulating governmental affairs at each administrative level. Additionally, ministerial departments are in charge of the related affairs, e.g. DONRE (Department of Natural Resources and Environment), DOST (Department of Science and Technology), DARD (Department of Agriculture and Rural Development).

Figure 2-1: Vietnam and the Mekong Delta Provinces

2.2. Water Sources and Supply

In Vietnam different water sources are available and used for different purposes.

The main governmental strategy is the Water Resources Development Plan to 2000 and tentative development plan to the year 2010. The key legislation is the Law on Water Resources from 1998. According to (MONRE *et al.* 2003) Vietnam is lacking an overall integrated strategy on water resources.

Agriculture is the main water consumer; water is used for cultivation, aquaculture and animal husbandry (Table 2-1). The highest demand is caused by irrigation - most rice growing areas are irrigated - counting for around 84% of the total demand in Vietnam and 89% in the Mekong Delta. Aquaculture consumes around 4% and livestock 0.9% in the Mekong Delta. Only 1.3% are used for domestic purposes (MONRE *et al.* 2003).

Table 2-1: Water consumption in the Mekong Delta and Vietnam (MONRE *et al.* 2003)

%	Mekong Delta	Vietnam
Irrigation	89	84
Aquaculture	4	4
Livestock	0.9	1
Domestic use	1.3	2

Table 2-2 shows the access to safe water in Vietnam: only 52% of the urban and 36% of the rural population have access to safe water (MONRE *et al.* 2002). The government's target is to reach 85% of rural people using safe water in 2010; 60 l^{-1} cap^{-1} d^{-1} are estimated (SNV Vietnam 2008b).

Presently, water related diseases are causing serious damages. 6 Mio cases of cholera, typhoid, dysentery and malaria had been reported in the last 4 years - an estimated cost of 400 billion VND (MONRE *et al.* 2003).

Table 2-2: Access to safe water in Vietnam (MONRE *et al.* 2002, 2003; SNV Vietnam 2008b)

%	2002	2010 (intended)
Urban area	52	95
Rural area	36	85

2.2.1 Groundwater Use in the Mekong Delta

Although in Vietnam less than 5% of the groundwater reserves are used, there is an over-exploitation in the Mekong Delta (MONRE *et al.* 2003). Usually water is gained from the second aquifer in around 60-100 m depth as the first aquifer is not rich enough. Due to the increasing number of used wells (11% in 1992, 23% in 1998 and 28% in 2002 (Soussan *et al.* 2005) the artesian ground water level decreased. The level of this aquifer- a Pleistocene sediment layer overlaying older rocks – has fallen around 0.5 m a^{-1} (Nuber 2008). If this tendency continues, a huge amount of wells will stop working in the near future.

In Can Tho Province, more than 50.000 tube wells are registered by the Centre of Rural Water Supply (Nuber 2005). Additionally, there are a great number of non-registered tube wells. According to (Soussan et al. 2005) around 24% of the households in Can Tho Province have own wells. In the southern and western provinces of the Mekong Delta coverage with drilled wells is even higher. It is reaching more than 80% in Ca Mau and Bac Lieu, and only slightly lower numbers for Soc Trang and Kien Giang. Tra Vinh and Long An also reach a higher coverage than Can Tho (Soussan *et al.* 2005).
Besides the small tube wells, community wells and supply stations, established by the Centre for Rural Water Supply and Environmental Sanitation (CERWAS), provide water for 38% of the rural population in Can Tho Province. These stations usually supply ~ 100 households, with capacities of ~ 4 $m^3 h^{-1}$. At the station groundwater is filtered successively by sand, gravel and coal. Twice a year water analysis is carried out by the Preventive Health Care Centre (Loi 2005).

Due to the high chloride and iron concentration in many areas, groundwater can not be used for drinking water purposes. The small scale tube wells mainly serve for livestock and for other domestic purposes (Stolpe & Nuber 2005).

2.2.2 Surface Water Use in the Mekong Delta

The Mekong (Figure 2-3) is one of the world's biggest rivers - it flows 4,620 km from East Tibet to the South Chinese Sea (Hori 2000). With around 510 Mio m^3 per year it provides abundant water – sometimes even too much: water is distributed similar to the rainfall; around 90% water flows down during the rainy season. So every year from June to November flooding occurs in large areas of the Mekong Delta (in around 1.4 - 2 mio ha of 4 mio ha total area) being the main water related disaster (MONRE et al. 2003; Soussan et al. 2005). Figure 2-2 shows one of the main roads in Can Tho where flooding has to be faced regularly.

Figure 2-2: Flooding in Can Tho city, occurring regularly during the rainy season

The river carries a lot of sediment: 160 Mio t a^{-1} (Nuber 2005) which contributed to the fertility of the alluvial soils in the area but makes its use as a drinking water source less attractive.

More 60 million people solely in the Lower Mekong Basin are using the river for drinking water, food, irrigation, hydropower, transportation and commerce (Figure 2-3). Millions more in China and Myanmar and beyond the boundaries of the basin benefit from the river (UNDP Human Development Report 2006).

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Water quality in the Mekong river branches depends a lot on the upstream users. The Mekong river is crossing six countries and used for different purposes before reaching its delta (Figure 2-3).

But water quality not only depends on the upstream users. Quality is strongly influenced by the local use and discharge of domestic and agricultural waste water especially in the smaller canals (Stärz 2008).

Its quality is also influenced by the tidal regime. Tidal effects of the Gulf of Thailand and the South Chinese Sea which are reaching up to 4 m can be noticed far inland (Hori 2000). Quality is usually better in areas where the water in the canals and rivers are exchanged very often: better in O´Mon, Can Tho, Phung Hiep, Chau Thanh and Tho Not compared to Vi Thanh Vi Thuy, Long My and Chau Thanh A (Stolpe & Nuber 2005).

The influx of salt water from the sea during the dry season affects farming as well as drinking water supply in various regions.

Figure 2-3: The Mekong river crossing China, Myanmar, Laos, Thailand, and Cambodia before reaching Vietnam (MRC & GRIDA 2009)

Due to two Vietnamese surveys (Vietnamese National Health Study and Viet Nam Household Living Standard Survey), 57 % of the households in Can Tho Province are using surface water as the main source for water supply. Surface water use is higher and reaches up to 88 % in the provinces Dong Tap, Vinh Long, An Giang, and Ben Tre (Soussan et al. 2005). In the examined rural and peri-urban areas around Can Tho, 53 % of the households were using surface water as one kind of water supply (Wieneke 2005).

The most widespread method found for water treatment within the household was boiling. Flocculation with aluminium sulphate (~ 0.1 kg/m^3) was common. Sometimes filtration by ceramic filters was found. Hypochloride was sold for water treatment, too (Wieneke 2005; own observations). Quality of the examined samples differed and did not always provide drinking water quality (Rechenburg *et al.* 2005). But there are a lot of households who are not treating their water at all, sometimes even despite bad smell. More than 40 % of the rural population of the Mekong Delta use untreated surface water for daily drinking and cooking (Soussan et al. 2005).

According to (Fuchs et al. 2005) 17 % of the rural population are connected to supply stations. In urban areas of Can Tho, 80-90 % (up to 95 % in Ninh Kieu district) are supplied with treated surface water by the Can Tho Water Supply and Sewerage Company (CTWSSC) who runs a number of surface water plants with capacities from 2,500 m^3/d to 50,000 m^3/d. The treatment is not providing drinking water quality. In addition bottled water is sold by the same company at a price of 15,000 VND per 20 l.

2.2.3 Rain Water Use in the Mekong Delta

Rain water is abundant in the Mekong Delta (1200-2300 mm per year) but it is distributed unevenly. Most of the 120 days of rain occur from May to October, due to a south westerly monsoon. During this time rain water can be gained and used as a household's water source. During November to April when there is only little rainfall, the use of rain water depends on the size of the household's storage tank.

In the selected pilot areas of Can Tho Province, rainwater was used by 57 % of the interviewed households in the rainy season and by 27 % in the dry season (Wieneke 2005) as one source of water. Rainwater use during the dry season depends on the availability of large rainwater tanks. The largest ones are constructed of concrete (Figure 2-4), other ones are steel tanks or clay jars.

Figure 2-4: Rainwater storage at a rural houshold close to Can Tho

Due to a Vietnamese National Health Survey (VNHS) in 2002, only 10% of the households in the Mekong Delta were using rain water for water supply (Soussan *et al.* 2005). In the provinces Tien Giang, Ben Tre, Long An, and Kien Giang rain water supply was higher (up to 20%), in the other provinces lower.

The water quality depends mainly on the storage conditions. Measurements of storage tanks proved surprisingly quite good hygienic quality for many tested tanks (Rechenburg *et al.* 2005).

2.3. Sanitation, Waste and Waste Water Treatment

The national framework is fixed in the Vietnam National Rural Water and Sanitation Strategy 2006-2010 (RWSS NTP II). Further strategies are set up in the Socio-economic Development Plan 2006-2010 and the Vietnam Development Goals (SNV Vietnam 2008b).

In 2003 only 44% of the population had sufficient sanitation (MONRE *et al.* 2003; Soussan *et al.* 2005). To meet the national target - 70% of the population with basic sanitation facilities by 2010 - Vietnam is trailing to achieve the MDG targets.

2.3.1 Sanitation

In the Mekong Delta, most villages are developed in linear structures along the canals. The adjacent canal is used as bathroom, for washing dishes and clothes, cleaning vegetables or gear (e.g. motorbikes), and waste water is mainly disposed directly into the surface water. Animal excrements as well as the excreta from "sky toilets" may run into surface water without any treatment. In rural areas "sky toilets" above fishponds are common. Although the government wants to abolish this practice, the direct use of excreta in fishponds is still common and widely accepted. Even the Farmer's Union in Can Tho accepts this form of "waste water treatment" (personal communication, 2004).

The government's target was to increase the number of rural households having hygienic latrines to 50% in 2005 (NRWSS 2002).

The Ministry of Health (MOH 2005) stipulates one of the following toilet systems:

- Double vault composting toilet
- Dry toilet with septic underground tank with ventilation tube (in areas without flooding)
- Self-infiltration tank, flushing toilet
- Toilet with double vault septic tank

According to estimations of Mr. Ton That Khai, Director of Can Tho´s Department of Health (2003, personal communication), around 10% of the rural households in Can Tho Province have modern latrines, 90% discharge their sanitary waste without any storage or other treatment into the surface water.
In the survey of 218 households in Can Tho and Hoa An distict (Wieneke 2005), 58% of them were using fishpond toilets, like those in Figure 2-5. Waste water out of the animal husbandry was discharged into the ponds as well. Similar numbers were given later in two Vietnamese studies of 2002 (Soussan *et al.* 2005): the Viet Nam Household Living Standard Survey (VLSS) and the Viet Nam National Health Study (VNHS). Due to these studies less than 10% of the household in rural Mekong Delta own a septic tank. Open defecation is highest nationwide: nearly two third (59% VLSS, 75% VNHS) are using fishpond latrines. Another 10 - 20% (23% VLSS, 8% VNHS) are without any latrine.

Figure 2-5: Fishpond toilets in the Mekong Delta

The use of fishpond latrines is unique in the Mekong Delta. Compared to the rest of the country, three out of nine areas suffer from even lower latrine coverage (South Central Coast > Central Highlands > North West).

In the city centres the number of households having sanitary tanks is higher. According to the regulations, houses have to install a septic tank (MOH Decree No. 08-2005/QD-BYT). In the urban area of Can Tho around 50% of the population are connected (Kermer 2006) and the overflow is disposed via municipal sewerage pipes directly to the Mekong river. The accumulated sludge is pumped out of the sanitary tanks in irregular intervals and put on the municipality's landfill in Tan Long (Vinh 2004). Black and smelly dumps with H_2S emissions are often found in peri-urban situations where open canals take up waste water of the surrounding households and even livestock. But even in the city, a large number of houses are disposing their sanitary waste directly into the river or canal (Figure 2-6).

Figure 2-6: Toilets along the Can Tho Canal, a branch of the Hau River

2.3.2 Solid Waste

Collection is limited to central areas and the national coverage is estimated to be around 50%. Higher percentages are reached in the cities, lower in peri-urban and rural areas (MONRE et al. 2002, Anh 2002).
In Can Tho City municipal collection covers around 50% of the waste. The waste is not separated at the source but private collectors are taking out valuable parts like metal, bottles etc. in front of the houses, prior to collecting. Additionally, scavengers are active on landfills where the collected material is disposed of.

In 2002 the "municipal waste" collected per person and day was 0.5-0.8 kg for large cities, 0.3-0.4 kg for small towns. Anh (2002) recognized an increase of 20% per year since 1992. From 1997 to 1999 it even increased by 30% (MONRE *et al.* 2002; MONRE *et al.* 2004).

In HCMC 59% of the waste is produced by households, 24% originates from the market, 7% from the streets and 10% from elsewhere (Anh 2002). The municipal waste consists mainly of organic material – an estimated 50-60% (MONRE *et al.* 2002), or even 50-90% (Anh 2002).

Further treatment is rare. Even simple composting is only carried out for a minor part of the material. According to Anh (2002), HCMC has capacities for around 12% of the collected waste; 88%-100% are disposed of on at a landfill. In the Mekong Delta there is no composting plant for municipal waste (Vinh 2004). The existing landfill in Can Tho does not have a bottom layer or leachate collection. Therefore it is expected that leachate reaches the underground and adjacent surface waters.

The remaining – not collected half of the waste material - is discharged to rivers and canals mainly (Anh 2002). Besides the direct disposal, significant input of solid waste via gully pots and the sewerage system into the receiving waters has been observed. Mr. Nguyen Trung Nghia from the Preventive Health Care Centre Can Tho estimates most of the non-collected waste will be disposed of into river and canals, around 10% may be burnt (Nghia 2005). Wieneke´s study (2005) confirmed that some garbage gets burnt. He also found that plastic is used for land extension together with mud from the canal. In rural areas, organic waste is feed to animals to a large extent, mainly for pigs (43%) and in aquaculture (22%). One quarter of the interviewed households did not use organic waste at all, they disposed of it into the backyard and the canal close to their houses.

Besides the municipal, another type of waste is generated in the field and during food processing. This kind of waste - crop residues - is usually reused: leaves of corn, cassava, vegetables for feeding livestock, roots, and vegetable residues as green manure, rice straw for mulching and fungi production or like rice husks, stalk of corn and cassava, for heating (Anh 2002).

Besides, animal waste has to be considered. This type of waste is reused to some extent: around 60 Mio t animal manure and straw-manure are produced every year (Anh 2002). A further considerable part is discharged into surface waters, especially in the south.

2.3.3 Waste Water

In central parts of Can Tho city waste water is discharged by sewers (Figure 2-7). Houses connected to a sewer pipe usually have installed a septic tank at their house prior to the sewer. Due to the small slope of the canals, waste water is standing in the sewer and digestion occurs frequently, leading to foul smelling emissions. During the rainy season streets get flooded (Figure 2-2) at high tide by back-flowing sewers. Thereby water is moved in and out.

In the Mekong Delta no municipal waste water treatment plant has been established yet. Even the biggest cities in Vietnam, HCMC and Hanoi are just about to build-up waste water treatment systems. The waste water is therefore reaching the receiving canal or river untreated (Figure 2-7). Currently the German KfW is assisting several municipalities in the Mekong Delta - Can Tho, Soc Trang and Tra Vinh - to build up waste water treatment plants for their central city quarters. The plant in Can Tho will consist of a primary treatment system for 150,000 inhabitants (Sao 2005).

Figure 2-7: Sewers leading waste water untreated into the surface water of Can Tho Canal

In peri-urban and rural areas no sewage pipes are installed. Open sewers may lead waste water to the river or other surface waters (Figure 2-7, left). Depending on the amount of waste water discharged, these can be very putrid places. In the rural area, besides human waste, livestock and intense aquaculture contribute to waste water pollution. Some improvement is reached when water is passing extensive fishponds or biogas digesters.

Only industrial waste water is treated. According to the available sources, in Can Tho about 50% are equipped with a waste water treatment system (Fuchs *et al.* 2005). Industrial enterprises gathered in “Industrial Parks”, e.g. in Can Tho at Tot Not are more likely to run a treatment plant. Small businesses like rice noodle factories, being widespread in the area, rarely own a treatment plant despite legal regulations.

2.4. Agriculture and Farming

Vietnam strongly depends on agriculture - not only as the source of living but also as an important export factor. During the last decade Vietnam developed from rice-importing country to the 2nd biggest rice-exporting nation after Thailand (UASD 2008). Besides, fishery products contribute 9.4% to the nation's exports of 26.5 billion US$ in 2004 (Wikipedia 2008).
Besides paddy rice, fish and seafood, other crops provided for export are coffee, rubber, cotton, tea, pepper, soybeans, cashews, sugar cane, peanuts, bananas (CIA 2008).

2.4.1 Land Use and Farming Systems in the Mekong Delta

In 2004, about 74% of the Mekong Delta´s surface was devoted to agriculture, and rice farming constituted about 70% of the agricultural land (GSO 2004). Besides, fruit cultivation, especially in areas with alluvial soils is quite common. Other crops like sugar cane, corn, cassava, sweet potato, yams are of minor importance. The major part of rural households (80%) had their main activity in agriculture or fishery. Out of these, 90% were devoted to agriculture, 10% to fisheries, forestry was negligible (0.17%).

The land use in Can Tho Province - corresponding to ~1/10 of the Mekong Delta area – is shown in Figure 2-3. Here the ratio agriculture to fishery is 99% to 1% respectively (GSO 2003). Farm structure in Can Tho Province is based on annual crops mainly (93%); mixed activities are the remaining (7%). Compared to the rest of the Delta, agricultural area accounts for a larger share in Can Tho Province (86%). Rice is the dominant field crop. Fruit trees are quite frequent, often contributing to the main income of the household.

Table 2-3: Agricultural land use in Can Tho Province (GSO 2003)

Unit	ha	%
Annual crops	204,022	
Rice	191,478	75
Others	12,610	5
Gardens	1,636	1
Perennial crops	48,764	19
Meadow	-	0
Water area for fish culture	159	0.06

Although agriculture is practiced in small, family owned units (Anh 2002; Wieneke 2005), agriculture and aquaculture have shifted from subsistence to market-orientated production and been intensified progressively. Intensive rice culture expanded during the 1990s and is still the principal farming activity.

The distribution of farm sizes in Can Tho Province and in the Mekong Delta is very similar. Their sizes are slightly bigger compared to the whole Vietnam (Figure 2-8). In Can Tho Province 50% are smaller than 0.5 ha and 70% smaller than 1 ha (GSO 2003).

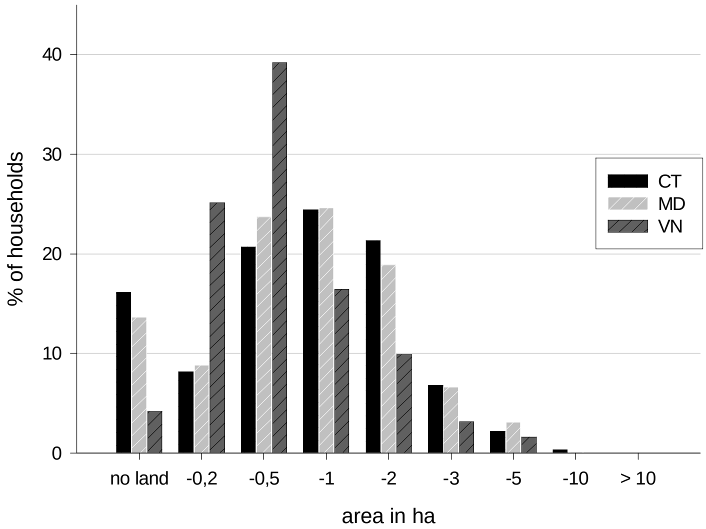

Figure 2-8: Farm size distribution in Can Tho Province (CT) compared to Mekong Delta (MD) and Vietnam (VN) (GSO 2003)

In the Mekong Delta the total amount of pigs was close to 3 million, poultry nearly 50 million heads. Only 30,000 cattle and buffalo were counted in 2001 (GSO 2003). A typical farming household (of 6 persons) owns 1-5 pigs (Figure 2-9).

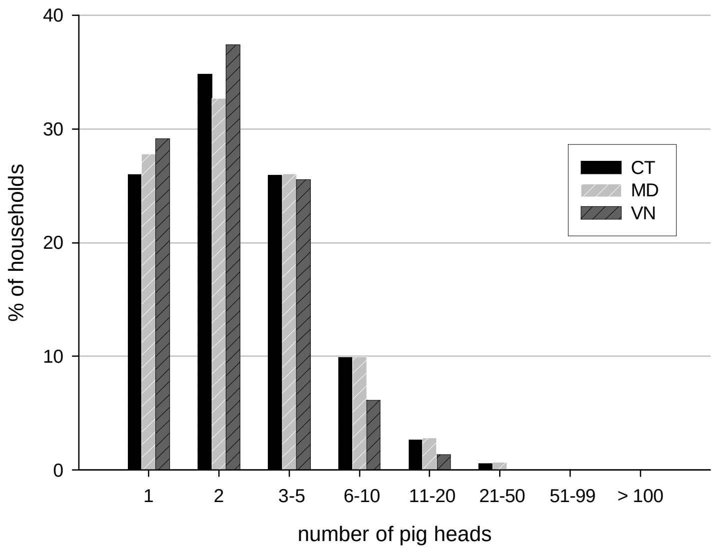

Figure 2-9: Number of pig heads per farm, in Can Tho Province (CT), the(Khoi & Tri 2003) Mekong Delta (MD) and Vietnam (VN) (GSO 2003)

Recognising the potential of aquaculture, the Vietnamese government has promoted diversification in agriculture since 1999, aiming to increase the contribution of aquaculture to economic growth (Nhan *et al.* 2007). Between 1999 and 2004, the growth rate of aquaculture production was rapid, annual growth rates of 31% for production and of 19% for farming area (GSO, 2003, 2005), suggesting a gradual intensification of aquaculture. Aquaculture increased to 600,000 ha and within eight years fish production has doubled, shrimp production nearly tripled (GSO 2004). Coastal shrimp farming and intensive fish culture upstream have been the main drivers of this expansion of aquaculture production, but there are indications that this growth is not sustainable, globally (Naylor et al. 2000) and regionally (Loc et al. 2007).

Like intensive rice farming, aquaculture practices are water-intensive (Boyd *et al.* 2002) and are leading to discharge of polluted water (Trung 2006). Besides, these effluents may also contain antibiotics and pesticides. On the contrary, extensive aquaculture may contribute to improve water quality (Mara et al. 1995).

2.4.2 Natural Conditions

The Mekong Delta area is classified as a "humid forest agro-ecological zone" (FAO 1978). Evaporation is well below the precipitation. Due to the uneven distribution of rainfall, deficiencies may occur from January to May.

Except for some hills at the border to Cambodia the Delta is plain with an altitude from 0-3 m. The geographical position on 8.5-12.5° north provokes a tropical monsoon climate which is characterised by high temperatures (monthly means 26.6 - 27.5°C) and high humidity (81-85%).

The main soil type in the Mekong Delta are Acid Sulphate Soils (WRB: Dystri-thionic Fluvisols; Sulfaquept) covering around 40% of the area, especially in the Plain of Reeds and in the area around Long Xuyen (Minh et al. 1996). The second prevalent soil type are Alluvial Soils (Fluvisols, Gleysols) along the Mekong and Bassac-branch (Figure 2-10).

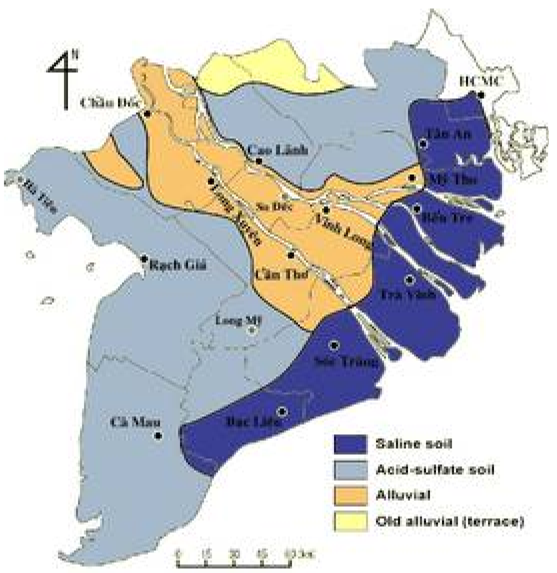

Figure 2-10: Soil Map of the Mekong Delta (http://cantho.cool.ne.jp/mekong/geo/geol_e.html)

Acid sulphate soils usually have a high clay content, like in Hoa An where 55-65% have been determined (Table 2-4). The pH is slightly above 4 at the top, and as low as 3.5 at the bottom of the profile. The topsoil of around 20 cm is rich in organic matter (10-15%). Usually it is underlaid by a firm layer with low permeability. Underneath a sulphuric horizon a sulphidic layer can be found (Becker *et al.* 2008; Hanhart & Ni 1994).
Agricultural use is limited and irrigation was used to improve Acid Sulphate Soils and make these suitable for agricultural use (Minh 1996). The low pH (Table 2-4) allows the cultivation of acid-tolerant crops only (rice, cassava, pineapple, sugarcane). Nutritional disorders occur due to aluminium toxicity, iron toxicity and phosphorus (and zinc) deficiency (Hanhart & Ni 1994). Phosphorus availability is low as existing P is fixed in low soluble Al- and Fe-Phosphates and firmly bond to organic material. As a consequence acid sulphate soils require more phosphate fertiliser than nitrogen fertiliser.

The alluvial soils emerged when river sediments were disposed as a "top layer". These soils are the most fertile soils of the area. Difficulties might be due to texture (high clay content) and flooding. Limiting factors in older, degraded soils are low availability of Ca, Mg, and P (Khoi & Tri 2003), due to low pH and high levels of free aluminium and iron. Besides, older, degraded alluvial soils, suffer from compacted soil and declining soil fertility.

Table 2-4: Selected physico-chemical properties of experimental soils (Becker *et al.* 2008)

Alluvial soil	Acid sulphate soil
Silty clay (57% clay, 34% silt, 9.0% sand)	Silty clay (44% clay, 55% silt, 1.0% sand)
pH: 4.9 – 5.1	pH: 3.3 – 3.5
C_{org}: 1.48 – 1.56%	Al_3^+: 37 mg $100g^{-1}$
N_{tot}: 0.14 – 0.17%	P_{avail}: 1.2 – 2.5 mg $100g^{-1}$

2.4.3 Fertiliser Use

Like in many other countries the use of organic fertilisers decreased in Vietnam due to the availability of mineral fertilisers. Although in traditional agriculture only organic fertilisers were used (Anh 2002) nowadays, an increasing amount of mineral fertilisers is applied. Consumption increased particularly during the last decades. Figure 2-11 shows the amount of fertiliser applied in Vietnam, Thailand and South East Asia.

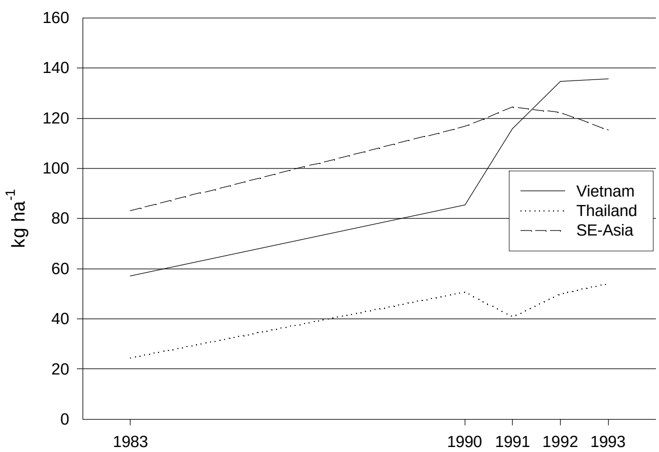

Figure 2-11: Development of mineral fertiliser used per ha and year in Vietnam, Thailand and South East Asia (Anh 2002; GSO 1996)

Vietnam's national production of mineral fertiliser is low, so the import of fertiliser has been increasing during the last years continuously. In 2003, 80% of N, 50% of P and 100% of K were imported (Son, 2004). Distributed to agricultural land, these figures amount to around 140 kg N ha-1 and year.
The money spent on imported fertilisers reached US$ 400-500 Mio in the late 90´s and increased to 1 billion dollars in 2008 (Saigon Times 2004; Vietnam Business News 2008).

Organic fertilisers contribute to the total nitrogen with ~210,000 t which equals around 16% of the applied N (Son & Ha 2006). Due to a calculation of Anh (2002) around one third of the necessary fertiliser in Vietnam is supplied by organic sources.
The use of manure - including human faeces - has a long history in Northern Vietnam (Humphries et al. 1997) similar to the Chinese tradition (Shiming 2002).

In the South, organic fertiliser from excrements was rarely used. Sometimes straw, ashes or green manure were applied (Wieneke 2005; Wohlsager et al. 2008). This might be due to cultural habits and the fact that annual flooding of the Mekong possibly provided enough nutrients formerly. Traditionally, the Khmer have been using manure. They are accounting for 1-40% of the poulation in the Mekong Delta (1 million Khmer according to Vietnamese government, 7 millions according to the Khmer Krom Federation, Wikipedia 2009).

Organic waste management could only be found at a small number of farms. Most of these were influenced by Can Tho University (own observations, Wieneke 2005): use of biogas and fishpond sludge for orchards, composting of pig manure or biogas sludge, vermicomposting of cattle manure.

2.4.4 Expected Development

It is most likely that agricultural intensification will continue due to increasing land and economic pressure (Bon 2003). Rice is expected to remain the main crop, although diversification, e.g. cash crops and a trend towards higher quality are to be expected (Trung & Arnold 2008). Besides, aquaculture will play a major role (Trung 2006).

Regarding the future development, the Mekong Delta Economic Cooperation writes (MDEC 2009):

"Mekong Delta provincial authorities would like exports to reach US$ 6.2 - 6.5 billion in 2010. They think that seafood will still be the major export of the delta in 2010. According to their plan, the delta is to have an aquaculture area of 0.85 - 1 million hectares. Some 2.24 million tonnes of seafood is to be farmed and caught, more than 60 percent of that farmed in 2010. Delta authorities target US$ 2.2 - 2.4 billion from seafood export earnings in 2010. There's also an interest in developing the textile/garment, footwear, plastics, fine art and handicraft and wood product industries to have a range of exports and increase total export revenue."

Increasing urbanisation will lead to higher demands in the city which might - depending on political decisions on land use planning and urban development - force a change towards urban agriculture (Bon 2003).

2.5. Conclusions Regarding the Status Quo in the Mekong Delta and Expected Development

The main findings regarding the status quo of water and related issues that have been identified in the Mekong Delta:

1. The main part of the population is dependent on agriculture as their source of living. Paddy rice is the main field crop.
2. Aquaculture and annual crops are practiced intensely already. Increasing population and higher demand will be followed by competitive and intensified land use and higher water pollution.
3. Surface water from canals and rivers is used for drinking and other household purposes. Only a minor part of the households receives treated water from a public supply. The knowledge and awareness of hygienic concerns, shortcuts etc. is limited.
4. At present no municipal wastewater treatment plant is operating in the Mekong Delta. Due to disposal of waste water, animal manure and solid waste into surface waters, black putrid canals are in evidence, especially in peri-urban areas.

3. Required Systems

3.1. Nutrient Demand in Agriculture

Generally, plants derive most necessary components for growth from water and air (C, O, H, 92%). A minor part (around 8%) is taken up from the soil, the plant nutrients N, P, K, Ca, Mg, S, Fe, Mn, Zn, Cu, B, Mo (Diepenbrock et al. 2005).

The macro nutrient nitrogen is plant available in the form of NO_3^- and NH_4^+. Urea that can also be taken up directly by the roots (Marschner 1995) is usually hydrolysed before uptake. It causes very acidic reaction (Finck 1992). In general, combined NO_3^- and NH_4^+ nutrition is best (Marschner 1995). On acid soils NO_3^- fertilisers should be preferred due to their alkaline effect (Finck 1992). Nitrogen can be synthesised from the air by the Haber-Bosch process.

Phosphorus may be present in different chemical forms. Its availability depends on the solubility and the pH level. For acid soils, slow-releasing (less soluble) phosphates are usually superior (Finck 1991a). Rock phosphate will be released slower and lead to pH increase. When supplied by mineral fertilisers, rock phosphates or processed, more soluble phosphates (e.g. Superphosphate $Ca(H_2PO_4)_2$, Novaphos $CaHPO_4$) are used.
Phosphorus can only be gained from natural resources: rock phosphate, in sediments and rocks. There are just a few countries owning the majority of resources (Maroq, China, USA). The resources are depleting, known reserves are estimated to last for another 80-100 years (Steen 1998; U.S. Department of the Interior 2007).

Potassium, the third macro element is usually supplied as KCl or K_2SO_4. K is mainly adsorbed by clay minerals just a minor part is related to the humic fraction. Due to the high clay content in the soils in the Mekong Delta, leaching of the water soluble element is rather limited.
Potassium is stored in the earth crust, Canada, Russia, USA and Germany. Reserves will last for another 300 years (Düe 1992).

These macro elements and micro-nutrients are important for plant growth. In agricultural soils where nutrients are removed with the harvest, they have to be supplied, by mineral, organic or organo-mineral fertilisers.

The fertiliser demand depends on the site specific local conditions (soil, crop, equipment, etc.). Besides their total amount, the availability of the nutrient is essential. For most nutrients, availability is best in the neutral pH range (Marschner 1995; Scheffer 2002).

For the soils in the Mekong Delta that are very acidic (see chapter 2.4.2) a fertiliser with alkaline effect or good buffer capacity should be preferred or additional liming has to be considered. Otherwise the existing nutrients may not be plant available.

Mineral fertilisers are quite popular as handling is convenient, nutrient concentration high and the dosage quite simple. Nutrients can be available exactly during the 4 - 6 weeks of the plant's main uptake.

The benefit of using organic substrates is their effect on the chemical, physical and biological soil status (Hsieh et al. 1990). By increasing the humic content of soils, structure and water holding capacity will improve; ion exchange capacity will rise, thereby holding nutrients in the soil (Finck 1992). Usually biological activity is enhanced, too (Gisi 1997; Guggenberger 2007), although there have been controversies (Finck 1992). Additionally micro-nutrients are provided.

If organic fertilisers or waste water are applied, hygienic concerns and potentially harmful substances have to be considered. Depending on the type of crop higher safety is required. Recommendations and handling instructions are given by FAO (Ongley 1996; Pescod 1992) and WHO (Blumenthal *et al.* 2000a; WHO 2006b) as domestic or agricultural effluent can never be completely free of pathogens.

3.2. Nutrient Recycling Potential

There are multiple reasons for enhancing technologies that recycle of nutrients from waste and waste water, e.g. decline of endless resources P and K, energy aspects etc.

Especially for waste water from households and farms where toxic load is usually low, reuse can be realised easily when remaining separated from industrial waste water.

Presently, most nutrients present in animal excreta are wasted, into the water bodies or elsewhere. Some nutrients are recycled in fishponds but a large part is wasted into canals (Watanabe 2000). Human waste is discharged into fishponds and into canals, too.
To find out about the recycling potential for nutrients from animal manure and human excrements, the following data were used:

- Amount of persons and animals, by the Statistical Bureau of the Province Can Tho (2002)
- Amount of nutrients excreted by animals, by the mean yearly values, following the data of Hydro-Agri (1993) and (Watanabe 2002)
- "Nutrient release" of humans according to the method of (Jönsson & Vinneras 2004): based on the nutrition data of the FAO (2002)

The nutrient release, i.e. the calculated amount of nutrients excreted per person resulted in 3.0 kg N a-1 and 0.44 kg P a^{-1}. This is less than in Europe, but more than e.g. in India or Uganda (Jönsson & Vinneras 2004).

Table 3-1 shows the amount of persons and livestock and the resulting nitrogen and phosphorus amounts excreted.

Table 3-1: Calculated Nutrient Fluxes in Can Tho Province and in the Mekong Delta

	Heads	N [t a^{-1}]	P [t a^{-1}]
Can Tho Province			
Persons	1,855,991	5,500	800
Pigs	289,159	3,200	947
Poultry	4,996,590	3,700	982
Cattle+ Buffalos	2,627	100	34
Total		12,500	1,963
Mekong Delta			
Persons	17,000,000	50,300	7,400
Pigs	2,900,000	33,400	9,496
Poultry	46,000,000	34,500	9,038
Cattle+Buffalos	30,000	1,000	393
Total		119,100	18,927

Per year, around 5500 t N from human and 7000 t N from animal excrements are potentially available in Can Tho Province and about 10 times in the Mekong Delta. The amount would be enough to supply half of the agricultural land of Can Tho Province (or the Mekong Delta, respectively) with ~100 kg N ha^{-1} from waste. Of course, in practice, not all waste can be recycled and - depending on the type of fertiliser substrate - N will be released slowly and not only in the first year.These amounts seem feasible when compared to calculations of Anh (2002) who calculated the amount of nutrients from livestock for Vietnam with 168,000 t N and 40,000 t P for 2002.

Interestingly, the distribution is different to the situation in European countries: in Germany, the nutrient fluxes by manures are much higher compared to those from human excrements (Eurich-Menden 1998).

Calculated by the price of urea (46% N, 7600 VND or 0.41 € kg^{-1} N) the nitrogen in waste of Can Tho Province equals 5 Mio € (95,000 Mio VND) which would be 3 € or 8 kg urea per household (6 persons).

3.3. Water Treatment Systems

Waste water contains substances that are undesired in water and therefore get removed in the waste water treatment process. Table 3-2 shows the substances, and the removal path that is usually carried out in recent (western standard) treatment systems. Typically a "train" of different treatments, including a mechanical, biological and chemical step is applied to fulfil all required cleaning steps (Horan 2001). Various techniques are available for each step.

Depending on the kind of waste water, the required goal, the available space, the technical know-how, or the economic potential, different systems and system combinations will be preferred (Gujer 2002; Mudrack & Kunst 2003).

Table 3-2: Removal of undesired substances from wastewater and potential recycling

Undesired Substance	Removal in a typical "classic" WWT system	Further use of recyclable substance	Limitations
Suspended solids	Mechanical 1. step (e.g. screen, sedimentation)		
Organic material	Mechanical 1st step and Biological 2nd step (e.g. activated sludge)	Transformation of removed sludge into biogas, reuse of sludge in agriculture	Due to heavy metal contents reuse in agriculture restricted
Nitrogen	Biological 2nd step	--	Lost as gas
Phosphorus	Biological 2nd step, Chemical 3rd step (Precipitation)	Phosphates as fertiliser	P in Fe, Al-phosphates not plant available (Simons 2008)

Other undesired substances as heavy metals, pathogens or organic contaminants may be removed partly within these steps, there is usually no further treatment in municipal WWTPs. Recycling is not desired or (in case of heavy metals) not cost-effective due to low concentrations. Other Elements, e.g. K, Ca, Mg, S, micronutrients etc. are not regarded as hazardous in waste water and therefore usually don't need to be removed. However, if waste water is recycled as fertiliser, these will provide nutrient supply to plants.

Among various solutions for improving the water quality, starting at the source – avoiding, reducing or improving waste water - seems an effective way. By this, drinking water treatment will be facilitated. Systems may be classified by different topics (Table 3-3).

Table 3-3: Classification of decentralised waste water treatment systems

Parameter	Options
Size	Central vs. decentralised
Function	Pre-treatment; main treatment and post-treatment
Process	Biological vs. chemical & mechanical treatment
Energy input	Extensive vs. intensive systems (regarding energy input)
Oxygen input	Aerobic vs. anaerobic treatment
Technical degree	Nature-orientated vs. technical
Technique	Dry vs. wet systems, Separation vs. combined treatment

3.4. Demand for Wastewater Treatment Systems

The present practice, e.g. open defecation, putrid canals, use of untreated surface water as drinking water may harm people and especially children. Epidemiological surveys proofed the occurrence of water borne diseases. Vietnam is one of the countries with highest worm prevalence (Feachem *et al.* 1983; Thien *et al.* 2007).

To improve the present situation, various systems may be used to treat waste water effectively. The selection should therefore be based on the characteristics of the area and the site. These have to be identified to determine the demands. Based on this information, potential solutions can be selected.

The main facts that are influencing the choice of suitable waste water treatment systems for the Mekong Delta are listed in Table 3-4 together with potential solutions.

Table 3-4: Main characteristics of the Mekong Delta regarding waste water treatment options

Factor	Finding	Demand for solution(s)
Water	High groundwater level, tidal influence, no slope	Avoid long pipe systems
	Abundant water at most places	Water recycling of minor importance
Society	Widespread settlings along open canals, one-storey buildings	Decentralised systems
	Modest technical know-how	Simple, low maintenance
	Wet Sanitation	Water-based system
	Rising energy demand, no stable electricity supply	System with low energy input or positive energy balance, use of available energy e.g. solar, biomass
Economics	Low economic status	Affordable, cheap or system that adds an extra value
Agriculture	Intense Agriculture (rice, orchard)	Fertiliser containing plant available nutrients
Soils	Acidification	Substrate with high buffer capacity, alkaline reaction
	Nutrient imbalances, declining soil fertility	Organic matter for increasing CEC
	Animal manure and human excreta contribute similar amounts to reusable nutrient potential	Solutions for both types of waste

With regard to the present technical knowledge and the status quo of infrastructure and energy supply in the Mekong Delta, focus should be put onto low-tech systems.

Looking at the wide-spread settlements and the flat land without slopes, as well as at the economic situation, first focus should be decentralised systems. A short overview of available decentralised systems is given.

Regarding the economic situation of the area, the systems should be affordable, or better, include an additional benefit for the operator. This can be reached by recycling of nutrients (N, P, K, micronutrients), organic substance and/or water which can amortise part of the investment cost. Besides many systems that just eliminate N (e.g. by denitrification to the air) and P (e.g. as insoluble precipitates), there are a number of systems with recycling potential.

Taking into account the factors practicability, environmental impact and costs, the following promising technologies seemed worth to be considered.

Table 3-5: Selected techniques, removal and reuse potential

Undesired Substance in waste water	Treatment technique	Potential Reuse	Product for reuse
Suspended solids	Biogas digester	see organic material or mineral substances	CH_4 for cooking and lightning
Organic material	Biogas digester	Biogas, Plant growth substrate	CH_4 Biogas sludge
N, P			Biogas effluent
Organic material	Composting Vermicomposting	Plant growth substrate Soil conditioner	CH_4, biogas sludge, compost, vermicompost
Nitrogen	Source separation (biogas digester)	N-fertiliser	Yellow water, or in biogas effluent
Phosphorus	Source separation (biogas digester)	P-fertiliser	Yellow water and its precipitates, biogas effluent, sludge

For the different wastewater streams of human and animal excrements several techniques were experienced for the recycling in the Mekong Delta (Figure 3-1).

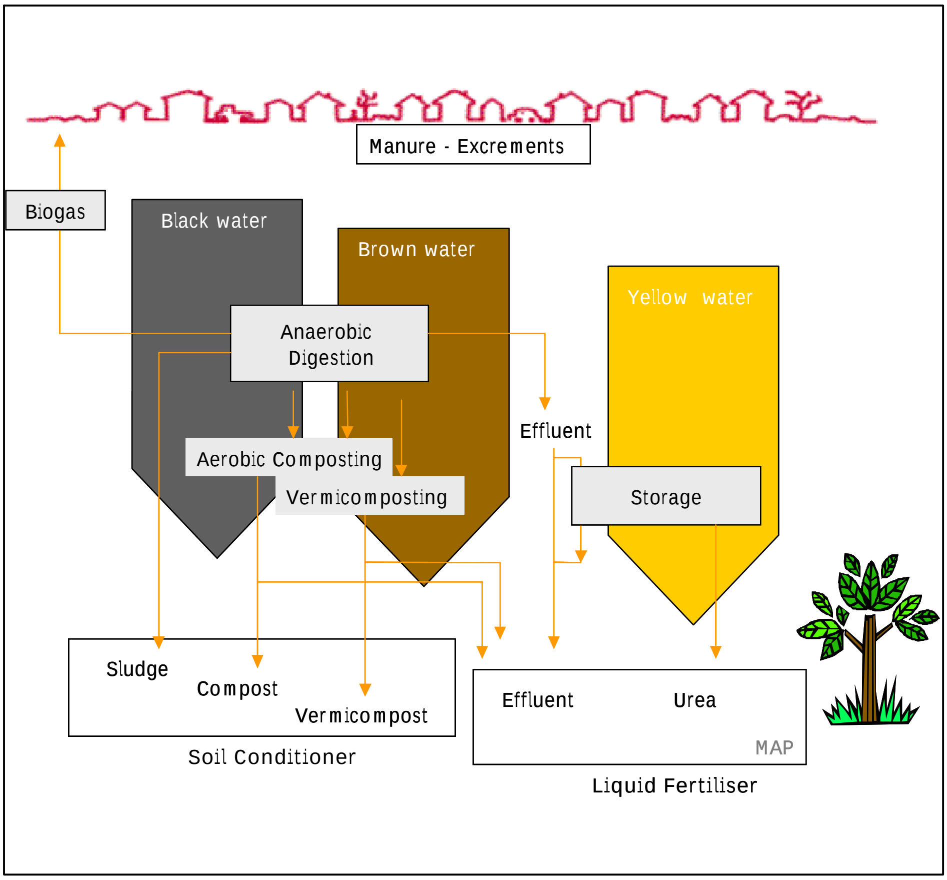

Figure 3-1: Wastewater treatment system with recycling potential for the Mekong Delta examined in this study

4. Anaerobic Treatment in Biogas Digesters

4.1. Background of Anaerobic Technology

Although anaerobic digestion (AD) has a long history, and septic tanks or lagoons have been used for many decades, the anaerobic process started to be considered a process suitable for municipal wastewater treatment only in recent years (Lens et al. 2001; Libhaber 2006).

Similarly, development of biogas technology for energy purposes has been intensified in developed and developing countries during the last decades. In Vietnam, too an increasing number of biogas systems have been introduced.

4.1.1 Process

Under anaerobic conditions, organic material is transformed to "biogas" by microorganisms. Biogas is a mixture of methane (CH_4), carbon dioxide (CO_2) and traces of some other gases (e.g. H_2S, N_2, H_2).

Suitable substrates for this process are wastewater, slurry, solid wastes and energy crops (Kaltwasser 1980; KTBL 2007; Nyns 1999). Depending on the material and the process, around 30-60% of the digestible material is converted into biogas by different microorganisms in several steps thereby reducing the volume of the raw material. Usually the process is divided into four consecutive steps (Figure 4-1): hydrolysis, acidogenic, acetogenic and methanogenic phase (Kaltwasser 1980; KTBL 2007; Mudrack & Kunst 2003; Sasse 1998; SBGF *et al.* 2004).

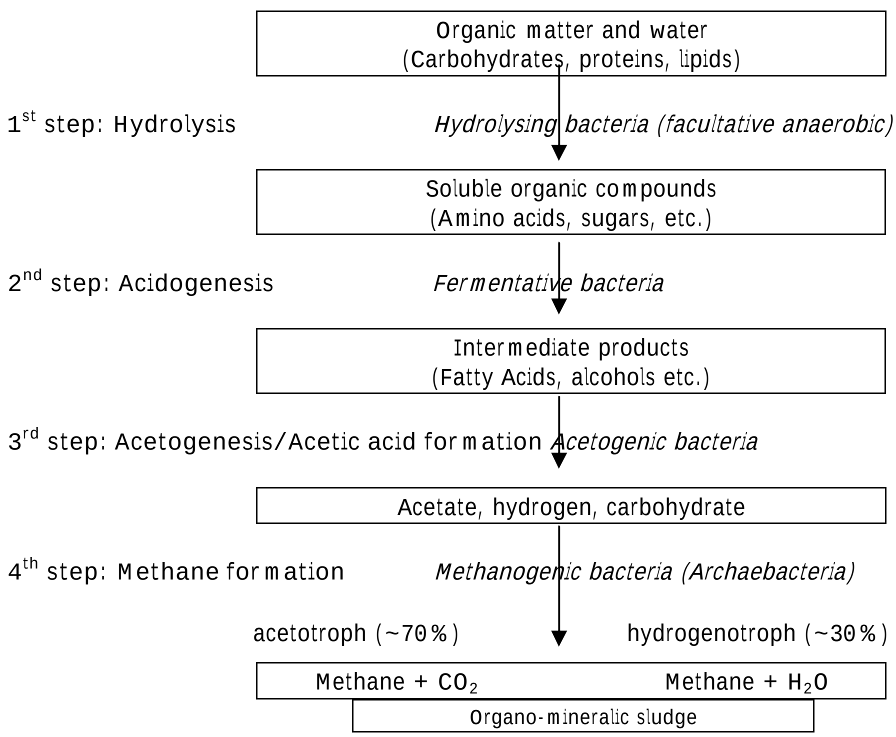

Figure 4-1: The four degradation steps to biogas formation

The simplest system is a single batch reactor where the four degradation steps take place in one single vessel until the process comes to an end. The most common systems are one-step reactors with continuous or semi-continuous feeding (Nyns 1999).
More advanced are two-step reactors where the first acid-forming step basically removes suspended solids and starts degradation, the second step will provide methane (Kaltwasser 1980; Li 1987; Zeemann *et al.* 2001). Advantages and disadvantages of one- and two-step reactors are summoned (Wandrey & Aivasidis 1983).

The main compound of the produced biogas is CH_4 (55-70%) which is considered a valuable fuel. The gas is non-toxic, non-smelling and lighter than air (0.56 of air density).
CO_2 which is mildly toxic (OES: 5,000 ppm) is heavier than air and present in biogas with around 30-45%.
Furthermore, H_2S, well known for its smell like rotten eggs, is emitted at around 1-2%. It is toxic (OES: 10 ppm) and it has to be removed for further processing of biogas (Kaltwasser 1980; SBGF *et al.* 2004). Besides, traces of nitrogen (NH_3, N_2), hydrogen (H_2) and carbon monoxide (CO) are found.
Biogas may ignite at concentrations of about 5-20% in the air (Kaltwasser 1980; SBGF *et al.* 2004), as methane is explosive at 5-15% CH_4 in the air (KTBL 2007). The produced gas may be used directly for cooking or light, it may be transformed in further steps to electricity or fuel (SBGF *et al.* 2004).

Energy gain can be estimated by 5-7 kWh m^{-3}_{STP} biogas according to its methane content (Angelidaki *et al.* 2003) or ~ 40 kWh t^{-1} slurry, and up to 2000 kWh t^{-1} fat (KTBL 2007). Besides energy, biogas sludge is produced.
The sludge quantities are much smaller than in the aerobic process due to the fact that only 5-15% of organic carbon is transferred to biomass compared to 50-60% in the aerobic process (Libhaber 2006). The sludge contains transformed organic substances (e.g. carbo-hydrates of lower molecular weight) and non-transformed substances (e.g. lignin, metals). Due to the gaseous carbon losses, the C/N ratio in the sludge is smaller compared to the initial material.

Depending on the temperature, different groups of bacteria are involved and the processes differ slightly:

- Thermophilic digestion takes place at 50-60°C (Ahring 2003b) and shows highest methane production, fast throughput and high pathogen reduction. It requires higher technology, greater energy input and more operation and maintenance than mesophilic digestion as the process is less stable (KTBL 2007).
- Mesophilic digestion is taking place at 20-40°C (Kaltwasser 1980). The optimum temperature range is given by different authors with 25-35°C (Wellinger 1999), 30-40°C (Ahring 2003b), 35°C (Marchaim 1992), 32-42°C (KTBL 2007). The process tends to be more robust and tolerant than thermophilic processes but gas production is smaller. Hygienisation - if required - has to be a separate process step.
- Digestion is possible under psychrophilic conditions, too but at low process rates (Angelidaki *et al.* 2003).

Compared to aerobic processes, the anaerobic process is more susceptible to instabilities caused e.g. by overloading, inhibiting substances or even inadequate temperatures (Gavala *et al.* 2003). Regarding the process stability, the most important factors are temperature, pH, substrate composition and toxins (Angelidaki *et al.* 2003). Besides pH, buffer capacity is an important factor (Hecht 2009).

4.1.2 Common Techniques and History

First descriptions on anaerobic wastewater treatment were found in the Indus area about 6000 years ago (Seyfried 1999). Methane bacteria which are responsible for the anaerobic process belong to the oldest living organisms that already existed before the oxygen atmosphere of the earth was created.

The fact that anaerobic processes are caused by microscopic organisms was discovered in the 17th century (Dunbar 1912, reprint 1998). Plenciz published the first germ theory in 1762. Volta put methane digestion on scientific footing in 1776 (Ahring 2003a; Marchaim 1992). The process has later been studied by Linné, Pasteur, Bechamp, Popoff and others (Jäkel 2003; Lerner & Lerner 2003).

A first application in wastewater treatment can be traced back to 1860 when the "Fosses Mouras", pits within a sewer, were introduced in France (Dunbar 1912, reprint 1998; Seyfried 1999), taking advantage of the anaerobic process. Since 1881 the technology has been gradually evolved (Kalogo & Verstraete 2001; Seyfried 1999). The best known is the Imhoff tank or "Emscher Becken" developed in Germany as a primary treatment for wastewater. The wastewater of more than 12 Mio people in Germany was treated by Imhoff tanks in the beginning of the 20th century (El-Gohary 2001). Anaerobic treatment of sewage was brought forward by the development of the Anaerobic filter (AF) and the Upflow Anaerobic Sludge Blanket reactor (UASB) in the early 70´s by Lettinga (El-Gohary 2001; Seyfried 1999).

Nowadays the largest number of anaerobic plants are built for sludge stabilisation in waste water treatment (Ahring 2003b). Anaerobic technology is also used as the 1st step in wastewater treatment for removal of solids. It found special interest for the treatment of industrial wastewater with high organic loads of more than 5,000 mg l^{-1} COD (Ahring 2003a, 2003b; Mudrack & Kunst 2003). Contrary to biogas plants aiming for energy production, the focus of anaerobic wastewater treatment systems is on COD reduction and fast performance - gas may not even be gained. These reactors usually belong to "high-rate" reactors with an HRT < 5 d (Pind *et al.* 2003).
An overview on anaerobic waste water treatment systems and their potential application is given in Table 4-1.

Nowadays in Germany, around 4,300 GWh are gained by wastewater treatment plants. This is nearly 20% of the total amount of 22,370 GWh gained by biogas; the other sources being landfills (6,670 GWh) and others, e.g. farms (11,400 GWh). In the EU the ratio is similar: 11,050 GWh by wastewater treatment compared to 62,200 GWh in total (36,250 by landfill, and 14,900 by others) (Wikipedia 2006).

Table 4-1: Anaerobic wastewater treatment systems

Type	Suitable kind of wastewater	R.	Kind of treatment	Main Source
Septic tank	Domestic wastewater	LR	Sedimentation, sludge stabilisation	(Sasse 1998)
Imhoff tank	Domestic wastewater	LR	Sedimentation, sludge stabilisation	(Sasse 1998)
Baffled tank, Rotating tank	Pre-settled wastewater of narrow COD/BOD ratio, strong industrial wastewater	LR	Anaerobic degradation of suspended and dissolved solids	(Sasse 1998)
Anaerobic pond / anaerobic lagoon	Strong and medium industrial wastewater	LR	Sedimentation, degradation	(Sasse 1998)
Continuously stirred tank reactor (CSTR)	Wastewater with high suspended matter	LR		(Pind *et al.* 2003)
Anaerobic filter (AF)	Wastewaters or sludges of low dry matter content, pre-settled wastewater	HR	Degradation, carrier material as an adhesive surface especially for methane producers	(Sasse 1998)
Fluidised beds (FB) Expanded beds (EB)	Wastewater with low dry matter content	HR	Degradation by microorganisms adhered to small particles (fine grained medium), into contact with the substrate by a strong upflow	(Mudrack & Kunst 2003)
Upflow anaerobic sludge blanket (UASB)	High and medium wastewater with low concentration of suspended solids, good capability for granulation	HR	Sedimentation and degradation by microorganisms in aggregates (thus remaining in the reactor despite a strong inflow)	(Mudrack & Kunst 2003; SBGF *et al.* 2004)

HR= High Rate, LR= Low Rate

Regarding small scale biogas digesters, China was one of the first countries to apply biogas digesters in large quantities for households (Feachem *et al.* 1983; Li 1987). Although digesters have been working already since the 1920´s, the major developments took place since the early 1970´s (Hawkes 1986) when the Chinese government stipulated the diffusion for energy gain and progress in agricultural production. Special agencies were established for building, management, and training.

In the 1980´s several million digesters were installed mainly in rural areas (An 2002; Hawkes 1986; Marchaim 1992; Wellinger 1999) but quite some were closed down due to leakage problems (Marchaim 1992).

Another biogas "pioneer country" is India where plants were experimentally introduced in the 1930's (Lawbuary 2000). India owns the 2nd largest amount of installed biogas digesters (An 2002).

The main types of small household scale biogas digesters are listed below (An *et al.* 1997; Li 1987; Marchaim 1992; Pressea 2000).

Fixed-dome (Figure 4-2) and plastic-tube digesters (Figure 4-3) are more popular in Vietnam compared to Floating dome.

Table 4-2: Anaerobic biogas systems for treatment of domestic wastewater used in Vietnam

Digester Type	Construction material	Gas storage	Details by
Fixed-dome, "Chinese digester" (FD)	Concrete and brick	Main digester ("water pressure methane digester")	(Marchaim 1992) (Hawkes 1986)
Plastic bag, Plastic-tube digester (PT)	Plastic foil (formerly PVC, now polyethylene)	External gas storage bag, plastic	(An *et al.* 1997; Kehlbach 2006)
Floating dome, Floating gasholder, "Indian design"	Steel, plastic	Floating dome	(Marchaim 1992)

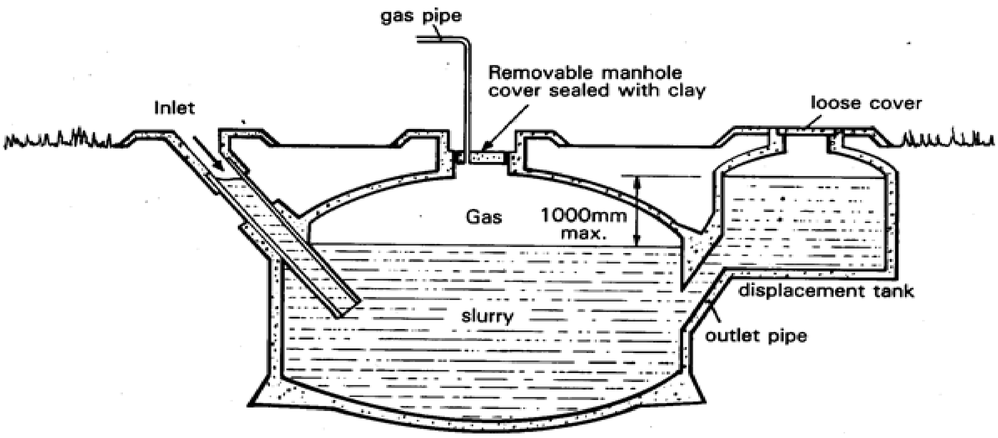

Figure 4-2: Fixed-dome reactor (Marchaim 1992)

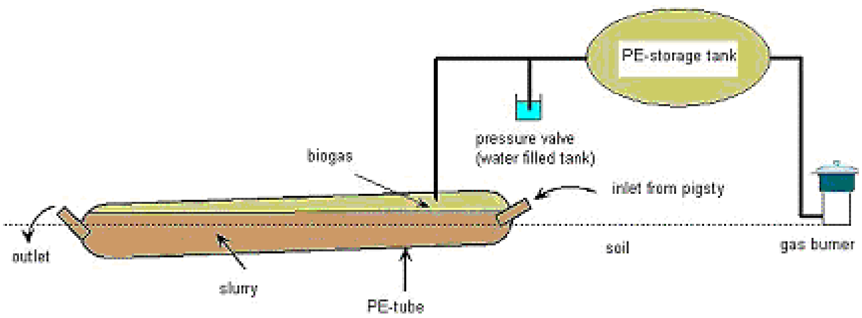

Figure 4-3: Plastic-tube reactor (Kehlbach 2006, modified)

In 1981, the United Nations Conference on "New and Renewable Sources of Energy" has designed bio-methanation as one of three priority energy sources for integrated rural development (United Nations 1981). However, besides alternative energy supply it is also and sometimes mainly a biotechnology to treat wastewater (Compagnion & Nyns 1986). Various international non-governmental organizations (NGO) have been implementing biogas digesters in Asia (Table 4-3). Besides, national governments ran programmes to enhance biogas technology, e.g. in the Philippines, Thailand, Nepal. In several countries, e.g. Cambodia, Bangladesh, Colombia, Ethiopia, Tanzania, especially the simple polyethylene tubular digester was promoted (An 2002).

Table 4-3: Organizations implementing biogas systems in Vietnam and other Asian countries

Organisation	Origin	Country implemented	Project details	Source
Borda	Germany	Vietnam, India, Indonesia		(Sasse 1998)
GTZ-GATE	Germany	Vietnam, Thailand, Nepal	Thailand from 1975, in the Mekong Delta from 1993-96	(GTZ-GATE 1999; Quynh *et al.* 1996)
Brot für die Welt	Germany	Vietnam		(Luu 1995)
Terre des Hommes	CH, G	Vietnam		(Luu 1995)
SNV	Nether-lands	Vietnam, Nepal, Bangladesh, Cambodia	> 220,000 households (1.35 mio people) in 5 Asian countries	(SNV 2008)
SIDA-Sarec / FAO-TCP	Sweden	Vietnam, India	Rural Energy Study Vietnam, Improved Biogas Systems Project (IBSP) in India	(Chinh *et al.* 2002; Ellegard 2000)
JIRCAS	Japan	Vietnam and others	Mekong Delta II project	(Yamada 2004)

4.1.3 Development and Practice in Vietnam

In Vietnam anaerobic waste water treatment takes place in septic tanks and small scale biogas digesters.

Septic tanks with one or two chambers are found mainly in new houses in urban areas as the actual legislation demands its construction. In the rural Mekong Delta, less than 10% of the households own a septic tank as two Vietnamese studies – the Vietnamese National Health Study and the Vietnam Household Living Standard Survey - revealed (Soussan *et al.* 2005). Usually the tanks are installed beneath the bathroom, below the floor tiles and do not receive regular maintenance (Kermer 2006; Nemcova & Wust 2008). They are emptied only when blocked - manually in most cases. First tank lorries were seen in Can Tho from 2006 on. The overflow of the tank is discharged via pipes to the river, or just into an open canal near by. Quite some of these septic tanks are in contact with the groundwater (Loi 2005). According to Sinh *et al.* (1996) their treatment effectiveness is rather low. In general septic tanks are able to reduce 25-50% COD (Sasse 1998). Due to the better treatment efficiency and potential energy gain, this study focuses on biogas digesters.

Biogas digesters are installed in peri-urban and rural areas and are treating animal and human waste. The technology was introduced to Vietnam in 1960 and since 1975 it has been one of the country's priority renewable energy research programmes (Pressea 2000). A Renewable Energy Action Plan (REAP) was prepared with assistance of the World Bank (Bogach *et al.* 2001) to support an acceleration of renewable energy production.
In Vietnam the main primary energy is biomass. In private households 80-90% of fuel is from biomass, wood and charcoal mainly (Pressea 2000).
Electrification reaches 90% of the communes in Vietnam. Nevertheless, in quite some provinces in the Mekong Delta the rate of power-consuming households is low: e.g. 41% for Tra Vinh, and below 30% in Bac Lieu and Ca Mau (General Statistics Office 2003).

According to Vietnamese reports (An 2002; Khang & Tuan 2002), more than 20,000 PT digesters have been constructed in Vietnam within the 10 last years. Vacvina and the DOSTE have installed several thousand Fixed-dome and Plastic-tube digesters from 1995-2000. Digester volumes varied from 1 to 15 m^3, and up to 200 m^3. Main implementation areas were the Red River Delta and the Mekong Delta (Pressea 2000). For the Mekong Delta, the Agricultural Extension Service of Can Tho Province reports a number of 10,000 sold plastic-tubes since 1997 (Ha Anh Dung 2004).

Recently, in 2003, the Dutch NGO SNV started to finance several thousands of household-based biogas plants in Vietnam for treatment of animal and human waste. Involving both public and private actors in the promotion and construction of domestic biogas, the programme has already reached over 37,000 plants in 25 provinces, including some in the Mekong Delta in the end of 2007. By 2010, the programme aims to have a total of over 140,000 fixed-dome plants installed in over 50 provinces (SNV Vietnam 2008a). The ultimate objective of the Vietnam Biogas Programme is to establish an economically viable, market-oriented biogas system. This goal fits with Vietnam's national policy and strategies in the areas of energy, rural development, water and sanitation.

This fixed-dome digester (FD), shown in Figure 4-4 is the dominating type in the research area in Long Tuyen (15 compared to 4 plastic-tube digesters, Wieneke 2005). This is due to joint activities of CTU and several NGOs, e.g. Brot für die Welt in Tan Phu Dong, Terre des Hommes in Long Hoa Commune (Luu 1995). The Renewable Energy Research and Development Centre of CTU, being in contact with the Gtz/GATE programme of Thailand had already constructed 261 plants by 1998 (Quynh 2004). In other areas, fixed-dome- reactors are less common. They are preferred due to less land requirement in suburbs of big cities (Chinh *et al.* 2002) but because of their price of ~300 € they are affordable only by wealthy farmers. Besides, installation and operation was found to be more difficult (An 2002).


Figure 4-4: Fixed-dome reactors in Vietnam during construction

The plastic-tube reactors (PT) are quite attractive as the material cost is only about 30 €. In the examined area, the dominating digester type was this PT reactor with 73% (Wieneke 2005). They consist of several layers of PE-foil, hold an in- and outlet for the liquid and another outlet for the gas. The gas is usually collected in a separate storage tank (Figure 4-5, left). Despite their bad reputation of limited durability, a study on 200 digesters revealed that more than half of the digesters survived 6 and more years (Khang & Tuan 2002).


Figure 4-5: Plastic-tube reactor in the Mekong Delta during construction (outflow into the pond) and gas storage bag underneath a shelter

In the examined project areas around Can Tho, 1.6% of the households in An Binh, 0.7% in Long Tuyen and 0.8% in Hoa An were using a biogas plant (Wieneke 2005). The small digesters of 1-15 m^3 were usually fed with animal manure. 21% of the users added their human sanitary waste. Use of other than these substrates was not reported (Wieneke 2005). The effluent of the fermenter was led into a fishpond, irrigation canal or into a runoff ditch. Usually biogas treatment was integrated in a farmer's "VAC" system where biogas (B) is added to a system combining crops (Vuon), fish (Ao), and pigs (Chuong), to "VACB" (Luu 1995). Owners make use of the biogas, mainly for cooking but also for lamps (Figure 4-6).

Figure 4-6: Biogas use for cooking and lightning in Vietnam

Owners of biogas plants as well as non-users expressed a high acceptance for biogas digesters and the connected advantages (Wieneke 2005). They especially mentioned as positive the advantages for cooking (saving of firewood, no smoke), saving of mineral fertiliser, less pollution of public canals and cleaner working conditions.

4.1.4 Main Findings and Research Questions

Regarding anaerobic technology and its usage in the Mekong Delta the following facts can be summarised:

- Anaerobic digestion is applied in waste water treatment to reduce carbon (BOD, COD) and solids. Nutrients remain in the liquid or sludge for further treatment or use. Energy can be gained of the produced methane.
- Biogas technology has been developed during the last century and improved a lot during the last decades.
- First attempts with biogas technology have been made in Vietnam and in the Mekong Delta. Among various digester types, simple, small scale, and affordable reactor types are available for potential use in the Mekong Delta region. So far the technology receives a high acceptance.
- Waste water treatment in the Mekong Delta is marginal. Besides a number of septic tanks in urban areas and extensive fishponds, biogas digesters represent one of the few existing installations.

Based on the review, the following research questions were asked:

1. Is the actual treatment by biogas digesters a suitable option for the discharged household wastewater?
2. Is the treatment efficiency of concrete fixed-dome biogas digesters different to the plastic-tube digesters?
3. Is the composition of the biogas substrates suitable as fertiliser for the existing soils and cropping systems?

4.2. Material and Methods

4.2.1 Field examinations

First, to get an overview on the existing anaerobic digestion systems in the Mekong Delta, a small field study was conducted. Experts of CTU selected 15 representative farming households with biogas digester in the provinces Can Tho, An Giang and Tra Vinh. These households were visited, inspected and owners were interrogated in interviews.

Based on these findings, two field trials were conducted to study the management and variability of wastewater treated in the two types of biogas plants:

1. Monitoring of two farming households in Can Tho Province for one year (May 2003 - April 2004), samples taken once per month to study the temporal variability
2. Screening campaign at twenty farming households in Long Tuyen, An Binh, and My Khanh in February/March 2006 to study the spatial variability

The monitored farm in Long Tuyen was equipped with a fixed-dome digester (FD-LT), the one in An Binh with a Plastic-tube digester (PT-AB). Both were situated at small canals and were using surface water. Additionally both farms received groundwater from own wells (depth ~100 m, diameter 0.6 m). The farms were performing an "integrated farming system", combining pigs, fishpond, garden and biogas ("VACB"). Both farm sizes were around 1 ha which is the average household size in the Mekong Delta (Wieneke 2005) . Management was examined by interviews, own inspections and a "management diary". Monthly measurements of the water were conducted: untreated wastewater prior to biogas treatment, treated outflow of the biogas fermenter, outflow of the succeeding pond and several additional sampling points in the household.

Screening was done at typical farming households around Can Tho (An Binh, Long Tuyen and My Khanh) with biogas digesters. These had been selected together with specialists from CTU and members of the people's committee. 10 farms with FD-digester (FD-10) and 10 with PT digesters (PT-10) were examined. The farms were performing the "integrated farming system" VACB on around 1 ha area. The farms were usually situated at a canal. 20 Interviews were carried out, water volumes were measured and water samples were taken and analysed. The water amount was quantified with a water meter at the inflow. In the screening the outflow was quantified by using buckets (40 l).

4.2.1.1 Collaboration and Additional Data

The monitoring of biogas digesters was carried out with the support of members of Can Tho University, especially T.K. Tien, Ms. Van, V.T.Y. Phi, L.H.Viet. Introduction and supervison of microbiologial analysis was given by A.Rechenburg from Bonn University.
The field data and samples of the screeing were collected and processed by J.Nuber, T.K. Tien, and Ms. Van, supported by further members of Can Tho University.

Due to adjoining research within the SANSED project (www.sansed.uni-bonn.de), additional data and information was available:

- Sociological data (Wieneke 2005)
- Hygienic-microbiological data (Rechenburg *et al.* 2008; Rechenburg *et al.* 2007)
- Hydrogeological and surface water data (Nuber 2008)
- Data on biogas yields (Plöchl & Nuber 2008)
- Biogas plant design (Plöchl *et al.* 2008)

4.2.1.2 Chemical and Microbiological Analysis Methods

Several chemical and microbiological parameters were analysed: field parameters as far as possible directly at the site, further parameters immediately in the laboratory at Can Tho University. A number of substances was analysed in Germany at Bonn University.

Table 4-4: Methods and instruments of analysed parameters

Parameter	Unit	Method	Instrument	Laborat.
PH		DIN 38404	MultiLine 340i, WTW	On site
Electric Conductivity (EC)	mS cm^{-1}	DIN EN 27888	MultiLine 340i, WTW	On site
Dissolved Oxygen (DO)	mg l^{-1}	DIN EN 25814	MultiLine 340i, WTW	On site
Temperature	°C		MultiLine 340i, WTW	On site
Redox potential	MV		WTW, pH 90	On site
Turbidity	NTU	EN 27027 : 1994	Model 966 Portable, Orbeco-Hellige.	CoT, CTU
BOB_5	mg O_2 l^{-1}	EN 1899-1 : 1998	Bottles, temperated storage	CoT, CTU
COD	mg O_2 l^{-1}		K-dicromate methode	CoT, CTU
Dry Matter	%	DIN 38 414 Teil 2	Oven Fa. Memmert, Germany	CoT, CTU
Organic Dry Matter (oDM)	%	DIN 38 414 Teil 2	Typ EF11/88, Lenton	CoT, CTU
Suspended Solids (SS)	mg l^{-1}	DIN 28 409 Teil 2	Fa. Memmert, Germany	CoT, CTU
Total Nitrogen (N_{tot}), Ammonium-N	mg l^{-1}	DIN EN 25 663 DIN EN 25 663	Kjeldahltherm Type KB 20, Vapodest 20, Gerhard, Germany	CoT, CTU
Ammonium-N (NH_4^+-N)	mg l^{-1}		QUANTOFIX N-Volumeter for faeces	CoT, CTU
Nitrate-N (NO_3^--N)	mg l^{-1}		Lange Cuvette test: LCK 339	CoT, CTU
Total Phosphorus (P_{tot})	mg l^{-1}		Molybdenum blue methode, some by Cuvette test: LCK 315	INRES
Ortho-Phosphate	mg l^{-1}		Lange Cuvette test: LCK 049	INRES
Potassium	mg l^{-1}		Lange Cuvette test: LCK 328	INRES
Heavy Metals	mg l^{-1}		Determined in the filtrate	INRES
E. coli	CFU 100 ml^{-1}	ISO 9308 Part 1 and 2	Fluorocult® Laurylsulfat-Bouillon, UV-flouroescence & Chromocult® Coliform agar (Merck)	CoT, CTU
Total coliforms	CFU 100 ml^{-1}	ISO 9308-2	MPN in Fluorocult® Laurylsulfat-Bouillon (Merck)	CoT, CTU
Faecal Streptococci	CFU 100 ml^{-1}	EU Bathing water directive 76/160/EWG	Kanamycine Esculin Azide Agar (OXOID)	CoT, CTU
Salmonella sp.	+/-	DIN EN ISO 19250, 2003-Draft	Rappaport-Vassiliadis broth (Merck), confirmation on Salmonella-Shigella agar (Merck)	CoT, CTU
Helminth eggs	number	Bailinger method recom. by WHO	Microscopically examined after density centrifugation	CoT, CTU

4.2.1.3 Statistical Evaluation

For statistical evaluation of the data, several tests were conducted. Selection of tests was done according to Kreyszig (1979) and RRZN (2005). Statistical analyses were processed by the programme SPSS (version 14.0).

- One-way ANOVA was used for analysis of variance e.g. comparison between FD and PT digesters.
- Correlation analysis was carried out for evaluating if parameters are correlated, i.e. connected to each other, e.g. different inflow parameters, in- and outflow samples (correlation of paired samples).

- Student's *t*-Test was applied for testing a hypothesis if two groups are significantly different; e.g. if the water quality improvement (quotient between inflow and outflow) was dependent on the type of digester; i.e. test whether the average of the difference is significantly different than μ_0.
  Dependent *t*-Test for paired samples when there is only one sample that has been tested twice (repeated measures) or when there are two samples that have been matched or "paired", e.g. inflow and outflow or inflow and quotient in- to outflow
- Factor analysis was done by principal component analysis (considering the total variance in the data); and common factor analysis (considering the common variance), e.g. factors explaining the different digester behaviour.

4.3. Results and Discussion

In the following chapter, results on wastewater quality, performance of the system and fertiliser quality are evaluated. Thereby the results of both studies (temporal and spatial) are combined. The evaluation comprises a mass balance, too.

4.3.1 Wastewater Characteristics

4.3.1.1 Waste Water Types

In the Mekong Delta, 80% of the population is living in farming households in rural and peri-urban areas, and most of them own livestock (see chapter 2).

Different wastewater types are produced:

- The largest amount of wastewater in rural areas originated from the animal husbandries. In the examined farms, the amount varied from 0.4 to 5.0 m^3 per day and farm. The manure was discharged directly with the cleaning water into fishponds and canals (~ 200 l pig^{-1} d^{-1}). Direct use of excrements e.g. as farm yard manure was not very common, according to own observations and literature (Watanabe 2000; Wieneke 2005).
- Household black water (toilet water) was discharged the same way: around 2/3 of the households in the Mekong Delta have been using fishpond toilets (see chapter 2.1.1. (Soussan *et al.* 2005; Wieneke 2005)). Another 10-20% were without any latrine at all (Soussan *et al.* 2005). For the majority of households not more than 2-5 l d^{-1} (not collectable) black-water per person are produced. Only a small number of households with flushing toilets will deliver more, ~10 l $pers^{-1}$ d^{-1} black water (see chapter 8). Compared to the water from animal husbandries, this wastewater is more concentrated. The amount of nutrients produced by humans, however, is about the same as the amount produced by animals in the Mekong Delta.
- Grey water, containing less nutrients, was disposed separately into open ditches or was running into the surface water, too. The amount varied depending on the availability of water and the water supply (e.g. community supply, electrical pump, hand pump). Measurements at CTU dormitory where tap water was supplied resulted in 60-100 l grey water $pers^{-1}$ d^{-1} (see chapter 8).
- Rainwater was often collected in rural areas and stored for drinking water purposes – different to discharge in urban areas. In the examined areas around Can Tho this was done by nearly 60% of the households (Wieneke 2005).

In Table 4-5 an overview is given on the quality of wastewater collected at the inflow of biogas plants of farming households in the surroundings of Can Tho city.

Table 4-5: Consistency of waste water at screened farming households

Parameter	Unit	Mean (n=20)	Coefficent of variation
Amount	$m^3\ d^{-1}$	1.80	0.62
Temperature	°C	28.7	0.03
ph		7.5	0.05
EC	$mS\ cm^{-1}$	1.7	0.38
Dissolved oxygen	$mg\ l^{-1}$	0.3	2.25
Redox potential	MV	-260	-0.28
Suspended Solids	$g\ l^{-1}$	4.9	0.60
Turbidity	NTU	370	0.34
BOD_5	$mg\ O_2\ l^{-1}$	4,400	0.41
COD	$mg\ O_2\ l^{-1}$	7,730	0.51
N_{tot}	$mg\ l^{-1}$	320	0.45
NH_4-N	$mg\ l^{-1}$	73	0.56
P_{tot}	$mg\ l^{-1}$	250	0.70
E. Coli	MPN 100 ml^{-1}	2.9×10^8	
Total coliform	MPN 100 ml^{-1}	5.0×10^8	

The concentration of the wastewater varies between the different households and also during time at the same household. Table 4-6 shows the spatial and temporal range: the lowest and highest concentrations of BOD and COD at the 20 different households screened once (spatial) and the two households monitored for one year (temporal). Compared to European municipal wastewaters, these concentrations are quite high. Typical concentrations in sewage wastewater are BOD ~100 mg $O_2\ l^{-1}$, and COD ~300 mg $O_2\ l^{-1}$ (Gujer 2002; Mudrack & Kunst 2003).

Table 4-6: Range of biological oxygen demand (BOD_5) and chemical oxygen demand (COD) in wastewater of examined farming households

	Parameter	Unit	Min	Max
Spatial (n=20)	BOD_5	$mg\ O_2\ l^{-1}$	800	6,800
	COD	$mg\ O_2\ l^{-1}$	1,200	14,000
	Ratio COD/ BOD_5		1.20	3.2
Temporal (n=24)	BOD_5	$mg\ O_2\ l^{-1}$	1,000	3,500
	COD	$mg\ O_2\ l^{-1}$	2,500	12,500
	Ratio COD/ BOD_5		2.3	9

The wastewater collected at the farming households is suitable for anaerobic treatment which can be applied for wastewater with a BOD > 1,000 mg $O_2\ l^{-1}$ (Sasse 1998) or a COD > 1,500 mg l^{-1} (Mudrack & Kunst 2003). Anaerobic treatment is especially favourably when COD concentrations are higher than 5,000 mg $O_2\ l^{-1}$ (Mudrack & Kunst 2003).

The COD/BOD ratios varied from 1.2 to 3.2 with a mean of 1.7 in the screening. The ratios in the monitoring were only slightly higher, with one exceptional high ratio of 9. The measured ratios of COD to BOD of 2 to 4 are usually seen in routine municipal sewage wastes (Dugan 1994, revised 1999). These values are indicating a fast degradation process, e.g. sugar has 1.5. Wastewater with more stable organic compounds shows a lot higher ratio (Gujer 2002). Ratios of COD to BOD of 4 to 6 are usually indicative of industrial type of wastes (Dugan 1994, revised 1999).

4.3.1.2 *Composition of Water Sources*

All examined farms were using surface water from canals near by, the majority had access to groundwater, too.

Surface water was examined in the monitoring. The quality was quite similar at the farms FD-LT and PT-AB (Table 4-7). Both canals were about 5 m wide. Depth varied from 0.5 to 1.5 m, influenced by seasonal and tidal changes (flow changed direction). The rainy season also affected the concentrations, e.g. lower EC, DO, BOD, COD from June to December.

Table 4-7: Typical composition of surface water at the farms FD-LT and PT-AB in Long Tuyen and Anh Binh

Parameter	Unit	Long Tuyen (n=14)	Anh Binh (n=14)
Temperature	°C	28.2	28.1
pH		6.7	6.7
EC	µS cm^{-1}	196 (225/167)*	193 (222/163)*
DO	mg l^{-1}	2.5	2.9
Redox potential**	MV	107	152
Turbidity**	NTU	46	46
BOD_5	mg O_2 l^{1}	6.8 (9.3/4.7)*	2.8 (3.2/2.3)*
COD	mg O_2 l^{-1}	15.4 (20.1/8.5)*	15.3 (25.7/6.6)*
N_{tot}	mg $^{-1}$	2.7	1.3
NH_4^+-N	mg $^{-1}$	0.2	0.2

* seasonal means (dry season/rainy season), ** n=7

Contrary to fairly similar results for surface water, groundwater results differed significantly regarding the EC (Table 4-8).

Table 4-8: Typical composition of groundwater at the farms FD-LT and PT-AB

Parameter	Unit	Mean FD-LT (n= 14)	Mean PT-AB (n= 14)
Temperature	°C	27.9	27.9
pH		6.6	6.6
EC	µS cm^{-1}	710	1360
DO	mg l^{-1}	2.1	2.1
Redox potential	mV	-75	-73
Turbidity	NTU	28	11
BOD_5	mg O_2 l^{-1}	0.5	0.8
COD	mg O_2 l^{-1}	2.2	2.2
N_{tot}	mg l^{-1}	1.4	1.0
NH_4^+-N	mg l^{-1}	0.8	0.8

These differences are due to the different geological formation in the 2nd groundwater layer and due to salt intrusions (Nuber *et al.* 2009). Due to the high salt content in AB, the use of groundwater is limited to household purposes like dish washing or laundry and animal husbandry. Drinking water is gained from surface water after treatment (sedimentation, ceramic filter). In LT groundwater is used for all purposes. Nevertheless, surface water has been used for washing of vegetables, laundry and others additionally. Groundwater was used at both farms for cleaning the pig sty (Figure 4-7).

Figure 4-7: Pig sty at monitored farms in Long Tuyen and An Binh

4.3.2 Management of Biogas Digesters and Process Parameters

4.3.2.1 Management

The size of the digesters examined in the monitoring experiment was 8 m^3. The digesters examined in the screening had volumes of either 6 or 8 m^3, calculated mean was 6.8 m^3 for both, fixed-dome and plastic-tube digesters (Table 4-9).
All biogas digesters were fed with wastewater from the pig sty, a few received a minor amount of human black water. Wastewater ran via pipes or open canals (constructed by tiles or concrete) into the biogas plant.

The reactors were charged semi-continuously, 2 - 4 times a day according to the cleaning of the pig sties. The total water volume varied from 0.4 – 3.1 $m^3\ d^{-1}$, with one exception showing 5.0 m^3.

The amount of pigs per farm varied from 5 to 48 fattening pigs. Most farms owned sows and piglets additionally. Typical weights were: sow 200 kg, fattening pig 70 - 100 kg, piglet 15 - 40 kg. The amount of pigs was calculated in Livestock Units (LU) and ranged from 0.8 to 7.7, with a mean of 2.9 LU. The mean LU was 3.2 for the FD-10 and 2.6 for the PT-10.
The pigs were fed mainly by rice bran, broken rice, and commercial food; with an amount of 1.5 - 2 kg per pig and day.
The main characteristics of the examined biogas reactors are listed in Table 4-9.

Table 4-9: Characteristics of the examined biogas systems in the screening

n=20	Unit	Min	Max	Mean
Size	m^3	6	8	6.8
Livestock	LU	1.0	7.7	2.9
Input wastewater	$m^3\ d^{-1}$	0.4	5.0	1.8
Input per digester size	$l\ m^{-3}\ d^{-1}$	50	840	280
HRT	d	1.2	10.9	5.8

4.3.2.2 *Hydraulic Retention Time*

The Hydraulic Retention Time (HRT) was calculated according to: HRT = $V_{reactor}$ / V_{inflow}

The resulting HRT varied from less than 1.2 days to 16.5 days with a mean of 5.3 days. The mean differed between the FD (4.6 d) and the PT-digesters (6.3 d), but the difference was not significant (One-way ANOVA).

Usually digestion time should be longer than the growth rates of relevant bacteria, which is ~10 days for a number of methanogenic species (Nyns 1999). In biogas systems for energy production digestion is done until 65-75% of the gas production is reached. This happens after around 10-20 days in a mesophilic system with good conditions (Kaltwasser 1980). In systems with high suspended solid content, long sludge retention times are necessary to provide a sufficient hydrolysis and methanogenesis (Zeemann *et al.* 2001). To decrease the smell, long retention times (30 d) are recommended, too (KTBL 2007).
In fast processing reactors used for wastewater treatment (UASB, AF, Fluidised/expanded bed reactor) HRT are smaller. Due to the high volumes to be processed, reasonable reactor sizes are more important than gas yields that could be achieved with longer HRT.

Despite the relatively low HRT, the systems are working and are producing reasonable gas yields with high methane content. CH_4 contents were mostly above 70% and average methane yields 250 and 310 l_N kg^{-1} oDM . Comparable batch tests resulted in 214 l_N kg^{-1} oDM for pig slurry and 311 l_N kg^{-1} oDM for pig faeces (Plöchl & Nuber 2008). Average yields for pig slurry are given with 240 l_N kg^{-1} oDM (KTBL 2007). In all farms, the amounts were enough for the household´s cooking demand.

Contrary to more advanced technology, no stirring takes place in these simple reactors. This way the sludge may settle down and remain in the digester for a longer time. This increases the removal of suspended solids and leads to higher degradation of organic substance.

4.3.2.3 *Loading Rate*

The loading rate expresses the amount of organic material fed into the digester.

In wastewater treatment, the loading rate of a digester is usually given as COD, e.g. "kg COD m^{-3}. Sometimes BOD or TOC are used, too. For calculations in water engineering, commonly a BOD_5 of 60 g $PE^{-1}d^{-1}$ and a COD of 120 g $PE^{-1}d^{-1}$ are used for calculations (Bahlo & Wach 1992; Gujer 2002; Mudrack & Kunst 2003; Wissing 1995). COD per LU is rarely measured.

In agricultural engineering, loading rates are assessed in organic dry matter (oDM) per m^3_{STP} and day, or as volatile solids (VS).
For pig slurry, the oDM is ~80% of the dry matter content (KTBL 2007). Supposing a slurry production of 5 m^3 $LU^{-1}a^{-1}$ with 5% DM, an amount of 200 kg oDM $LU^{-1}y^{-1}$ can be used for estimations (KTBL 2007).

For better assessment, the general use of VS as the characterising parameter is suggested (Pind *et al.* 2003) as there is no general transfer coefficient for converting oDM/VS to COD. The factor depends on the kind of substrate and so there are only material-specific coefficients. Literature comparing the two parameters is scarce. Henze *et al.* (2000) found factors of 1.5-1.7.
For the wastewater analysed in the screening, the coefficient found was 1.69 (CV 0.28) being in accordance with the above mentioned coefficients (Henze *et al.* 2000):

COD [g O_2 kg^{-1} wastewater] = 1.69 * oDM [g kg^{-1} wastewater].

In Figure 4-8 the correlation of COD and oDM is shown which is fitting quite well for the wastewater of the screening.

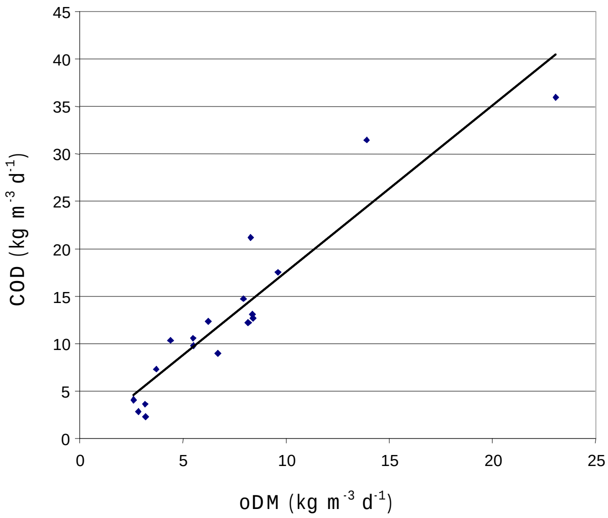

Figure 4-8: Correlation of COD and oDM values of wastewater for biogas plants, r^2 = 0.941

The data showed also a high correlation between BOD, COD, dry matter, organic dry matter and suspended solids (Table 4-10).

Table 4-10: Correlation between characterising influx parameters (kg d^{-1}), α < 0.001

	SS	DM	oDM	BOD
DM	0.928	--	--	--
oDM	0.924	0.997	--	--
BOD_5	0.739	0.847	0.828	--
COD	0.955	0.941	0.941	0.866

In Table 4-11 the characteristic parameters related to loading rates are listed and compared to values recommended in literature.

Table 4-11: Characteristic loading rates of biogas systems

n=20		Mean FD-10	Mean PT-10	Mean all	(Bank 1993)	(KTBL 2007)	(IRC 1986)	(Kaltwasser 1980)	(Wellinger 1999)
Suspended solids	g l^{-1}	5.6	4.4	5.0			6%		
DM	kg DM m_N^{-3} d^{-1}	2.0	1.1	1.6					
oDM	kg oDM m_N^{-3} d^{-1}	1.5	0.8	1.2		0.3-2.7*		3**	3-3.5**
COD	kg COD m_N^{-3} d^{-1}	2.5	1.5	2.0	3-60		< 4		
COD/DM	kg COD kg^{-1} DM d^{-1}	1.1	1. 1	1.1	0.1-2		< 1		

* crop-slurry mixture, ** pig manure

The daily organic load, oDM per m^3 (mean 1.2, CV 0.78) is small compared to German biogas systems for energy gain but it is still in the possible range of 0.3-2.7 kg oDM $m_N^{-3}d^{-1}$ (KTBL 2007). For energy crop-slurry mixtures an optimum of 2.5 kg oDM $m^{-3}d^{-1}$ is recommended (KTBL 2007), for pig excrements an amount of 3 kg oDM $m^{-3}d^{-1}$ (Kaltwasser 1980) or 3.0 - 3.5 kg VS m^{-3} d^{-1} (Wellinger 1999) are suggested. In another study with the same kind of PT digesters in Southeast-Asia, an optimum loading rate was reached at 2 kg DM $m^{-3}d^{-1}$ (An & Preston 1999).

The amount of livestock (LU) and the COD in the waste water were correlated (r^2= 0.825, α= 0.005). This is shown in Figure 4-9. Interestingly, the ratio of COD per LU was higher in farms with a FD-digester compared to those with a PT-digester (data not shown). The same applied to the amounts of organic dry matter per LU. In FD farming households organic dry matter was 3.8 kg LU^{-1} d^{-1} compared to 2.5 kg LU^{-1} d^{-1} (i.e. 67%) in PT households.

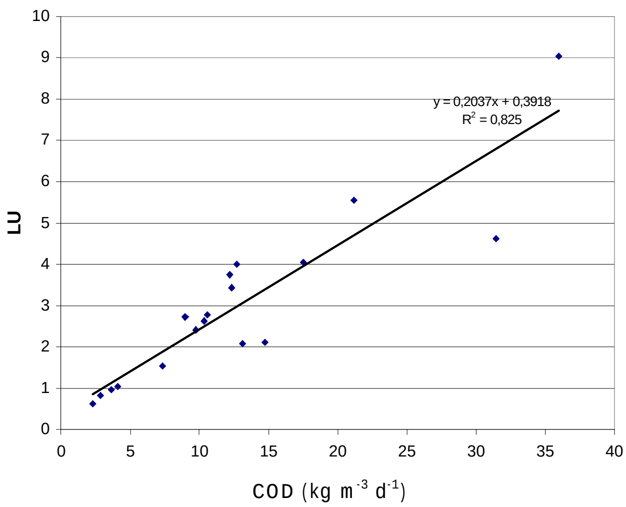

Figure 4-9: COD in wastewater per Livestock Unit (LU)

This phenomenon was visible for P and N, too. Waste water of PT-10 households only had 54% of P and 85% of N per LU compared to FD-10 wastewater.

This might be due to different feeding – households owning an expensive FD digester might have higher quality fodder than owners of less expensive PT digesters.

Although the data of COD, BOD and LU were correlated, to predict the BOD or COD in wastewater by the amount of LU needs a better database.
Calculating loads by the human average load per person (70 kg) will underestimate the COD as values in animal manure are higher. Due to different nutrition and physiology, C/N in human excrements is lower than in pig manure (Kaltwasser 1980).
Estimations with the German rule of thumbs on DM and oDM (KTBL 2005) were found to be applicable for Vietnamese conditions, too, although they might overestimate the amount slightly (Table 4-12).

Table 4-12: Estimated and measured dry matter contents in waste water

	Unit	Mean FD-10	Mean PT-10	Source
Amount of livestock	LU	3.2	2.6	(own counting)
Expected dry matter	kg d^{-1}	13	11	(KTBL 2005)
Measured dry matter	kg d^{-1}	13	8	(own measurements)

4.3.2.4 Process Parameters

Process stability is most important for an effective operation of the plants. Imbalances often affect the methanogenic bacteria, lead to accumulation of acids, decrease of pH and may even cause breakdowns. Typical parameters used for process control are discussed in the following, more details are discussed in literature (Pind *et al.* 2003; Wellinger 1999).

The temperatures in the Mekong Delta are high enough all year to allow digestion without insulation. The groundwater used for cleaning the pig sties had a temperature of around 28°C (mean in LT: 27.6°C, SD: 1.3), with a seasonal minimum in December-January. The temperatures in the reactors in FD-LT and PT-AB already reached 28°C - 29°C in the early morning (Table 4-13). The temperatures in the screening (FD-10 and PT-10) were around 1°C higher as the screening took place from March until May, at the end of the dry season.

These temperatures were slightly below the optimum temperature range for the mesophilic process. Mesophilic processes are taking place down to 20°C but the process is slowed down below 30°C and lower yields are to be expected (Angelidaki *et al.* 2003; Nyns 1999).

In the examined biogas systems, the pH was in the range of optimal functioning (Table 4-13) although the specific optimum of anaerobic bacteria differs: The optimum pH for methanogenic and acetogenic bacteria is at pH 7; methanogens grow very slowly below pH 6.6. For acidogenic bacteria optimum is at around pH 6 (Angelidaki *et al.* 2003).
For most sampling dates, a pH decrease between inflow and outflow (around 0.5 units) had been found. The in- and outflows in the screening were correlated (correlation of paired samples, $r^2 = 0.527$, $\alpha=0.017$). Interpreting the pH is complex as pH is affected by many factors, especially by organic acids, carbon dioxide or ammonia. Depending on the buffer capacity of a substrate, pH will remain stable despite an already affected process (Hecht 2009; KTBL 2007). Process control should therefore not rely on pH only but include measurements of alkalinity, too (Angelidaki *et al.* 2003; Hecht 2009).

Table 4-13: Process parameters in different digester types, mean of different plants (FD-10, PT-10) and mean of different sampling dates (FD-LT and PF-AB) compared to literature data

		FD-10 n=10	FD-LT n=10	PT-10 n=10	PT-AB n=10	(Bank 1993)	(KTBL 2007)	(Mudrack & Kunst 2003)	(Angelidaki *et al.* 2003)	(IRC 1986; Nyns 1999)
Temp.	°C	29.2	28.1	29.0	27.8	35-40	32-42		25-40	30-37
pH		6.9	6.9	7.0	7.0	~ 7	6.7-7.5*	6.8 – 7.2	6-8.5	7-8
DO	$mg\ l^{-1}$	0.1	0.1	0.1	0.3		< 0.4			
Redox potential	mV	-332	-315	-330	-270		< -250			-265
EC	$mS\ cm^{-1}$	3.2	3.8	3.6	3.4					

Crop-slurry mixture

Dissolved oxygen and Redox potential were low enough to allow survival of methanogenic bacteria and to permit methane formation (KTBL 2007). The values were slightly lower in the fixed-dome compared to PT-tube. The difference was not significant in the ANOVA.

An increase of EC from inflow to outflow was noticed in both, the monitoring and screening. In the screening, the increase was 1.9 $mS\ cm^{-1}$ (FD-10) and 1.7 $mS\ cm^{-1}$ (PT-10); the mean quotients outflow/inflow were 2.2 (FD-10) and 2.3 (PT-10).
In the monitoring, increase was higher in the FD digester (Figure 4-10): the quotient outflow/inflow was 2.9 (FD-LT) compared to 1.7 (PT-AB).

Relevant processes for the increase are the formation of new molecules – organic material is broken up, thereby producing organic acids (VFA). Another effect is the formation of ions, e.g. NH_4^+, out of organic material and urea.

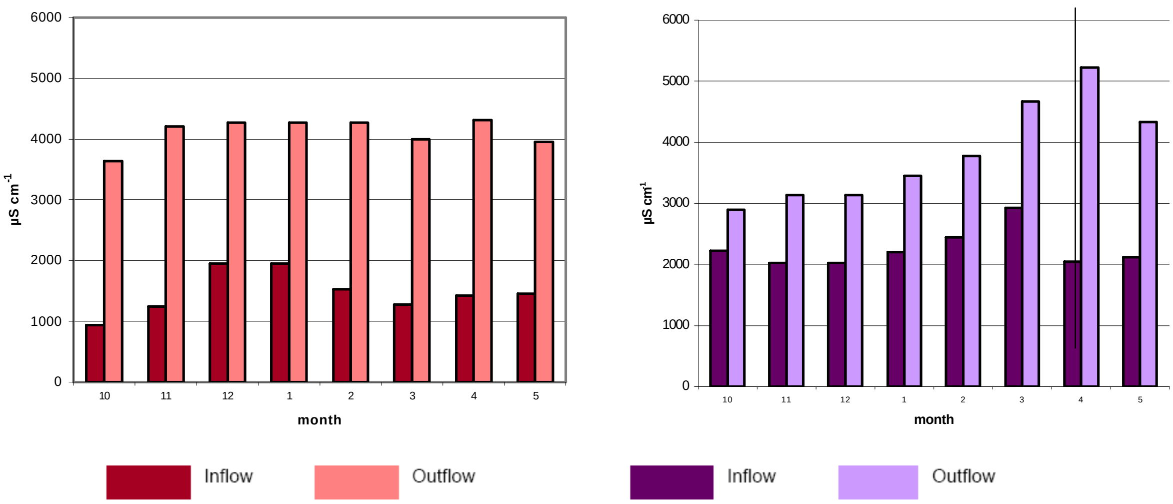

Figure 4-10: Electric conductivity in inflow and outflow of digesters FD-LT (left figure) and PT-AB (right figure)

The digestion process usually ran fairly stable in the examined systems; farmers rarely complained about breakdowns. This is mainly due to the material and the low loading rates. Pig slurry has a high buffer capacity (Angelidaki *et al.* 2003; Nyns 1999). It also contains the necessary micro-nutrients per kg COD degradation (Bank 1993; KTBL 2007; Mudrack & Kunst 2003). Due to the low loading rates the systems are not at their limit and the danger of acidic breakdowns is minimised. In general, process control has not been a major concern in farm-scale bio-digester (Wellinger 1999).

Nevertheless, high levels of protein in swine manure and even more in slaughterhouse waste may inhibit methanogenesis (Ahring 2003b; Wellinger 1999). This is due to nitrogen - when transformed to NH_3 - and sulphide. Different tolerable levels of free ammonia are reported, as low as 55 mg N l^{-1} and adaptable up to 800 mg N l^{-1} (Angelidaki *et al.* 2003). The toxic effect is stronger the higher the temperature and pH. This may even lead to difficulties in thermophilic digestion of swine manure (Ahring 2003b). Temperatures in the examined digesters, however, are much smaller.

Another reason for occasional breakdowns may be antibiotics used for swine treatment or other chemicals in the wastewater. Salts, heavy metals, detergents, antibiotics, oils and lipids are known for inhibiting the digestion process (Mudrack & Kunst 2003).
Two-step reactors are expected to be superior when waste contains a high concentration of inhibiting substances (Ahring 2003b).

4.3.3 Wastewater Treatment Efficiency

4.3.3.1 Suspended Solids and Turbidity

To change dirty, muddy wastewater into clean, clear water, the removal of contained cloudy matter and solids is necessary.

By treatment in biogas plants suspended solids were reduced. Taking the mean of all screened digesters, the concentrations decreased from 4.8 to 2.3 g l^{-1}. The effluent of the FD-systems contained 31%, the PT systems 60% of the inflow concentration. However, the difference was not significant (multivariate test of quotients).

Turbidity was reduced in the screened digesters from 370 to 192 NTU (n=20). FD digesters still contained 60%, PT digesters 40% of their inflow values.
In the monitoring, turbidity values were higher. The mean inflow values were 560 (FD-LT) and 558 NTU (PT-AB). They were reduced to 323 (58%) and 371 (67%), respectively.

4.3.3.2 Biological and Chemical Oxygen Demand

Organic substances have to be removed in wastewater treatment. This process needs oxygen, so the demand of oxygen can be used as a means for the amount of organic material in wastewater. The biological oxgen demand represents the fast degradable fraction that can be used by microorganisms for their existence (e.g. BOD_5 within 5 days), the chemical demand is the total amount.

Figure 4-11 shows the changes in BOD_5 and COD concentrations from inflow to outflow in the monitored digesters.

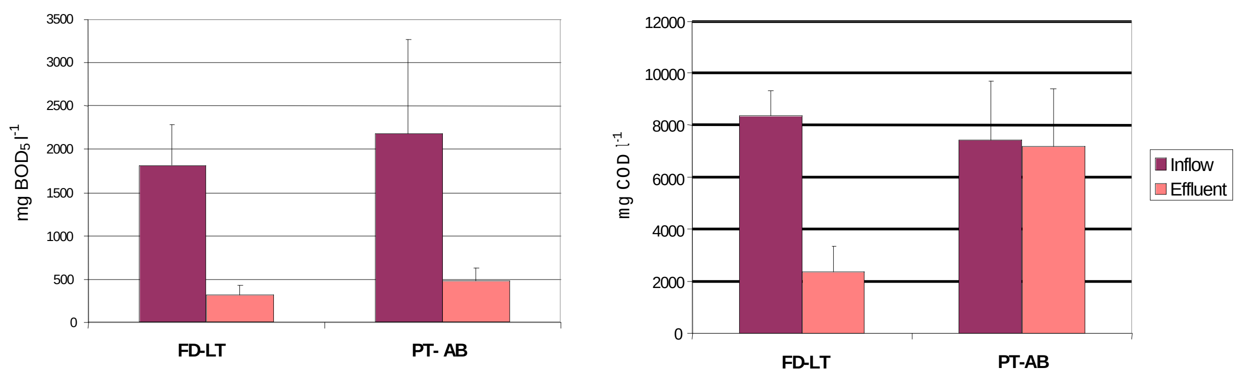

Figure 4-11: Mean BOD_5 and COD concentrations in inflow and effluent of monitored digesters: fixed-dome in Long Tuyen (FD-LT) and plastic-tube in An Binh (PT-AB) (n=12, bars indicate standard deviation)

BOD_5 concentrations were reduced similarly by 83% in FD-LT and 78% in PT-AB in average (Table 4-14).
In the FD-LT digester, the COD reduction was 75%, close to the BOD_5 reduction (Table 4-14). In PT-AB, however, the COD concentrations in in- and outflow did not differ significantly when taking the mean of one year (Figure 4-11). Besides seasonal changes, e.g. increasing size of growing piglets and sales of livestock, formerly accumulated sludge may be released due to an increasing sediment layer. High turbidity in the outflow of PT-AB confirms this hypothesis. If long retained sludge was released this material would contain a higher amount of non-degradable or less-degradable substances compared to the inflow. This result - high COD and a reduced BOD_5 - was found for PT-AB.

In the screened digesters, BOD_5 effluent concentrations were half the inflow values (51% reduction). Half of the examined digesters reached a BOD_5 reduction higher than 70%. Another third of the digesters reached reduction rates of 30-60%. As a tendency, concrete FD-biogas plants tended to perform better: outflow values were 37% (FD-10) and 67% (PT-10) of the inflow value. But the differences were not significant in the ANOVA.
For the COD, factor analysis revealed a difference between the COD reduction quotients of fixed-dome and plastic-tube digesters ($r^2 = 0.253$, at $\alpha = 0.033$).

Table 4-14: Mean reduction of BOD_5 and COD concentrations in FD and PT digesters in percent

		FD-10	FD-LT	PT-10	PT-AB	(Bank 1993)	(IRC 1986)
BOD_5 degradation	%	63	83	33	78		83
COD degradation	%	64	75	29	5	50-90	53

Comparable degradation rates were reviewed (Khang & Tuan 2002). In four Vietnamese studies on PT digesters, COD reduction was 60-70% at slightly lower inflow concentrations.

The results also match with the general expectation of 50-90% COD reduction in the anaerobic digestion process (Bank 1993). Processes with higher loading rates (15-20 kg COD m^{-3} d^{-1}), may reach reductions of 80% within one day or even half a day hydraulic retention time (Wandrey & Aivasidis 1983).
Reductions of 15 kg COD m^{-3} d^{-1} may be reached in UASB or AF reactors, up to 30 kg COD m^{-3} d^{-1} in Fluidised bed reactors (Mudrack & Kunst 2003).

According to the Environmental Regulations of Vietnam (1995), the tolerable pollution caused by animal wastes is 100-400 mg COD l^{-1} (Khang & Tuan 2002). The examined effluents were usually below these limits.

4.3.3.3 Microbiology

For hygienic-microbiological concerns, the indicator microorganisms *E. coli*, total coliforms, *Salmonella*e and helminth eggs were determined. These are generally acknowledged (Blumenthal *et al.* 2000a; WHO 2006b), offer a good overview of the effectiveness and make different treatments comparable.

However, newly it is recommended that in anaerobic environments faecal streptococci (enterococci) should be examined additionally as an indicator bacteria (Ahring 2003b). These bacteria are present in manure and other materials but more resistant than coliforms to high temperatures and anaerobic conditions. Most other pathogens will be inactivated before the enterococci in anaerobic environments (Ahring 2003b).

E. coli and total coliforms were reduced around 1 $\log_{10}$ unit (90%) and more when passing the digester. However, the reduction was higher in the concrete FD digesters, e.g. for *E. coli* the reduction efficency was 98.7% in the FD-LT compared to 80% the PT-AB (Figure 4-12).

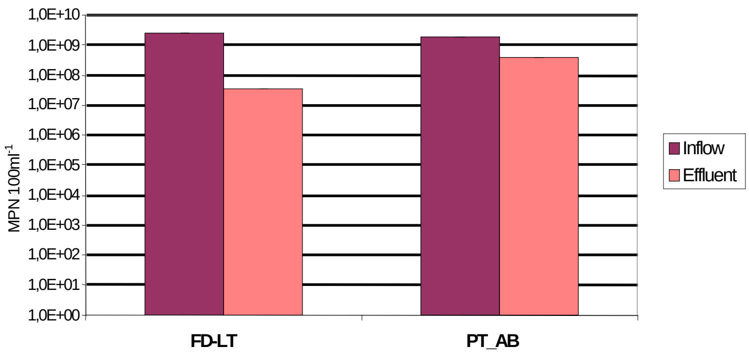

Figure 4-12: Mean *E. coli* concentrations in inflow and effluent of monitored digesters: fixed-dome in Long Tuyen (FD-LT) and plastic-tube in An Binh (PT-AB)

In the screening, bacterial reduction was higher in the fixed-dome digesters, too (Table 4-15). The difference between the digester types was more obvious for helminth eggs that were found regularly at low concentrations in the effluent of the plastic-tube digester but not in the effluent of the fixed-dome digester (Table 4-15). This might be due to improved sedimentation at the fixed-dome digesters.
In the monitoring, more than 50% of the registered helminth eggs were reduced, an average of 300 helminth eggs per litre decreased to 120 eggs l^{-1} in the efflux (Rechenburg *et al.* 2005). The species identified most often was *Clonorchis sinensis*, the oriental liver fluke (Rechenburg 2008).

Table 4-15: Median reduction of hygienic parameters in fixed-dome (FD-LT, FD-10) and plastic-tube (PT-AB, PT-10) digesters (Rechenburg *et al.* 2005; Rechenburg *et al.* 2008)

	Unit	Inflow FD-10	Effluent FD-10	Log_{10} reduction	Inflow PT-10	Effluent PT-10	Log_{10} reduction
E. coli	MPN 100 ml^{-1}	3.3×10^8	1.4×10^7	1.4	2.1×10^8	2.0×10^7	1.0
Total coliform	MPN 100 ml^{-1}	3.3×10^8	1.9×10^7	1.2	2.7×10^8	2.3×10^7	1.1
Helminth eggs	number l^{-1}	45,000	0	5.4	140,000	75,000	0.3

Salmonella sp. were found in the effluent regularly, even though this organism was sometimes not present at the inflow. As development of pathogens during storage was rarely reported (Rechenburg *et al.* 2008; Strauch 1996), this appearance is most likely due to the time lag between in- and outflow. Usually, microorganisms die when removed from their natural habitat - their survival time depends on the species´ resistance and the difference of the new media compared to their regular environment. For *Salmonella* sp. the survival time in pig slurry was found to be around 40 days (Strauch 1996) so these might be able to maintain in the anaerobic digester for several days.
In a study on the occurrence of *Salmonella* sp. in the Mekong Delta, different species had been isolated in 16% of the pig excrements (n=50) whereas the prevalence in surface waters was higher: 43% in river samples and 38% in pond samples (n=55) (Phan *et al.* 2002).

An overview on survival times of pathogens in different media is given by Feachem *et al.* (1983). In case of short retention times, the effluent of a biogas plant might not contain significantly less pathogens than the raw sludge (Feachem *et al.* 1983).

It is generally agreed that storage time leads to reduction and death of pathogens, with increasing death rates at higher temperatures. E.g. sanitation in a thermophilic process (52°C for 10 h or 55°C for 6 h) equals the heating step of 70°C for 1 h (Angelidaki *et al.*

2003). This is usually regarded as sufficient for later application in agriculture (required in the European decree # 1774/2002 (EG 2002)).

For a number of microorganisms, a toxic antibiotic effect, caused by methanogenic bacteria has been observed (Kaltwasser 1980). This effect has been verified for Paratyphus B, Enteritis Breslau and Enteritis Gärtner. As well, *Vibrio cholerae* and Hepatitis A virus seem to be very sensitive against anaerobic processes (Kaltwasser 1980).

In summary, treatment in the biogas reactors improved the hygienic status but the quality of the wastewater is still not sufficient for recommended use in agriculture (ANZECC 2000; WHO 2006b), as shown in Table 4-16.

Due to the fact that irrigation water is used for food crops which are eaten - some might even be eaten uncooked - the WHO demands the same high health protection as used for drinking water: the tolerable burden, expressed in DALY should be $\leq 10^{-6}$ for wastewater used in agriculture (WHO 2006a).
The concentrations are also above the Vietnamese Standard Regulation that accepts $5x10^3$ to $5x10^4$ MPN ml^{-1} for *E. coli* in animal wastes (Khang & Tuan 2002).

Table 4-16: Trigger values for faecal coliforms in irrigation waters used for food and non-food crops (ANZECC 2000)

Intended use	Level of faecal coliforms
Raw food crops in direct contact with irrigation water (e.g. via sprays, irrigation of salad, vegetables)	<10 CFU 100 ml^{-1}
Raw food crops not in direct contact with irrigation water (edible product separated from contact with water, e.g. by peel, use of trickle irrigation) or crops sold to consumers cooked or processed	<1,000 CFU 100 ml^{-1}
Pasture and fodder for dairy animals (without withholding period)	<100 CFU 100 ml^{-1}
Pasture and fodder for dairy animals (with withholding period of 5 days)	<1,000 CFU 100 ml^{-1}
Pasture and fodder (for grazing animals except pigs and dairy animals, i.e. cattle, sheep and goats)	<1,000 CFU 100 ml^{-1}
Silviculture, turf, cotton, etc. (restricted public access)	<10,000 CFU 100 ml^{-1}

For eliminating pathogens more effectively, additional heating (as required e.g. in Germany) or thermophilic processes may be helpful. Both is not likely to happen in the near future due to the economic situation in the Mekong Delta. As low-cost alternatives, longer retention times and additional treatment steps, e.g. in ponds or constructed wetlands are suggested to improve the hygienic status and reduce the bacteria (*E. coli*, coliforms) ~ 3 log_{10} units (Mara 1997; Mara & Johnson 2007).

4.3.3.4 Summary and Recommendations for Process and Wastewater

The installed anaerobic digesters showed rather stable process parameters despite a negligible process control management. Fortunately, digestion of pig slurry at low capacities provide fairly safe conditions (Wellinger 1999).

The biogas systems contribute to an improvement of water quality. Suspended solids, BOD, COD and pathogens are reduced during passage. Furthermore, bad odour is removed. The results indicate a slightly better performance of concrete fixed-dome compared to plastic-tube digesters although the database was not large enough to prove this statistically.

Advantages of wastewater treatment by an anaerobic system are low sludge production, low land demand, positive energy balance, low emissions (Klingler 1998). Furthermore, the cost for installation is not too expensive.

In spite of the outlined improvement, the performed treatment was usually not enough to meet Vietnamese legal requirements, the FAO-standards for irrigation water (Pescod 1992) or WHO recommendations (WHO 2006b), mainly due to high concentrations of indicator organisms.

To improve the process, the following may be applied:

- Increasing the HRT, by less water volume or by a larger, or - even better to increase sedimentation - a second additional digester
- Improve the reactor configuration
    - A two-step reactor (with a small hydrolysis) would not only increase digestion but also ban bypasses and should lead to a higher pathogen reduction (KTBL 2007; Wandrey & Aivasidis 1983). This might be an even more appropriate option when a more solid co-substrate with different biodegradability e.g. rice straw, or strongly acidifying, e.g. coffee pulp is charged, too (Nyns 1999). Two-step reactors are usually recommended for wet wastes from the food industry or in case of high sulphate concentrations (Mudrack & Kunst 2003) and at low temperatures (Zeemann *et al.* 2001). pH control is less difficult in this reactor type (Li 1987).
    - UASB reactors are highly recommended in tropical conditions especially for treatment of black-water or total domestic wastewater (Zeemann *et al.* 2001). They allow high loading rates (10-12 kg COD per m^3 and day), short HRT (6-12 h), good performance and favourable BOD removal (Horan 2001; Nyns 1999). It is not suitable for material with a high solid material content (Wellinger 1999).
    - Higher temperatures increase the efficiency. As even small temperature differences from day to night are reducing the degradation of BOD, COD and pathogenic microorganisms, insulating may contribute to keeping the temperatures at the optimum range of mesophilic digestion. The investment-revenue of this means has to be studied.
- For reaching better hygienisation, heating is a common treatment, i.e. demanded in the regulations of EU and US-EPA. Heating could be realised by using the energy of the produced CH_4, although this would reduce the revenue and therefore not reach the farmer´s acceptance.
- The outflow may undergo a post-treatment, usually aerobic e.g. in a soil filter or pond system. The combination with wastewater ponds is suggested by several authors (Horan 2001; Marchaim 1992), and anaerobic digestion may be the first step in a pond cascade (Mara 1997).
  Another suggestion of Ahring (2003b) to recover nutrients by membrane treatment after anaerobic digestion may be useful elsewhere; it does not seem an appropriate solution for the Mekong Delta.
- Prior to the digester, a sedimentation basin, or a screen could be installed, although this seems least recommendable. In order to gain the energy of the organic substrate, avoid emission of greenhouse gases, and to minimise the odour disturbance, other options are to be preferred.

4.3.4 Balance and Fluxes

4.3.4.1 Pig Sty Balance

A rough estimation was made regarding the pig sty output; the ratios between the amount of pigs, the amount of produced waste, its dry matter content, BOD and COD were evaluated.

On-site the number of sows, fattening pigs and piglets was counted. For further calculations all were transformed to Livestock Units (LU) using the factors 0.33, 0.18 and 0.04 (KTBL 2005).

Per LU, a mean amount of 4.3 kg dry matter, and 3.2 kg oDM was produced per day. The mean amount of suspended solids was 3.4 kg LU^{-1} d^{-1}. The mean BOD_5 was 3.0 kg LU^{-1} d^{-1} and the COD 4.8 kg LU^{-1} d^{-1}. Per LU a daily amount of 210 g N was discharged.

These parameters (DM, oDM, BOD_5, COD, N, NH_4^+) were correlated to LU (Table 4-17). The mean amount of P was 190 g LU^{-1}, but P was not correlated to LU. This might reflect different feeding customs with differences in protein diet.

Nevertheless, P was correlated to BOD_5, COD, dry matter and organic dry matter and to N and NH_4^+.

E. coli was correlated to coliforms and helminths. They did not correlate to any other parameter.

Table 4-17: Waste water parameters correlating to LU (Pearson's correlation)

Significance $\alpha < 0.01$	Mean	r^2	Significance $\alpha < 0.05$	Mean	r^2
Inflow volume	750 l	0.563	Suspended solids	3.4 kg	0.475
BOD_5	3.0 kg	0.596	Dry matter content	4.3 kg	0.463
COD	4.8 kg	0.628	Organic dry matter	3.2 kg	0.444
N_{tot}	210 g	0.606			
NH_4^+-N	45 g	0.684			

Compared to prior calculations on nitrogen flows in Can Tho Province, Watanabe (2000, 2002) had resumed a release of 65 g N d^{-1} and pig. "Pig" was not clearly defined but equals most probably one sow. With 200 g N d^{-1} LU^{-1} it matches the above measured results, 210 N d^{-1} LU^{-1} quite well.
Another study estimated much smaller fluxes in their calculations on farming systems in southern Vietnam (Hedlund & An 2000). They estimate 90 g N d^{-1} for 6 fattening pigs (~ 1 LU) by assuming that one third of the nitrogen in fodder will be excreted by the animals. Due to Watanabe (2000, 2002) 60 % of the fodder-N is excreted.

4.3.4.2 Water Balance for Biogas Digesters

For balancing the fluxes, the water volumes of inflow and outflow were determined. When measuring, the amount of outflow was usually lower than the inflow: The outflowing volume was 75 % at the FD and 85 % at the PT digesters. Including two FD-digesters where no outflow at all left the fermenter, it is only 58 % for the FD digesters.

Most probable reasons for the different volume measurements are:

1. Part of the difference is due to the measurement inaccuracies: the inflow pipe was usually below surface and hardly reachable. A water meter for measuring the

inflowing water had to be fixed at the water pipe just before the pig sty. This way, water spilled in the sty, did not reach the fermenter and consequently, could not be measured in the outflow. Nevertheless, this error was regarded minor compared to total volumes of several hundred litres but may reach up to 5-10%.

2. In the "water pressure"-FD reactors variable gas pressures occur. This is causing different water levels in the tank which will lead to temporal differences in the output volumes. In the study, volume measurements were limited to one hour time after loading. Later dripping water was not collected.
3. Leakage of FD-digesters via concrete, cement and brick is mentioned by Li (Li 1987): due to high pressure in the tank, fixed-dome digesters are supposed to suffer from leakage and seepage frequently. Regarding PT-digesters, the foil may allow diffusion of gases. This result was gained in Germany with a similar pilot plant (Kraus 2008), the digester in Vietnam were not examined.

Observation of the biogas digesters did not provide any indication for leakages at the surface – the control of the embedded parts was not possible.

For further assessment, it is assumed that there had not been any leakages at the digester. The inaccuracies of spilled water are small and similar at all pig-sties and therefore not regarded further in the balance. It is further assumed that the missing volume will be dripping out later during the day. Therefore, the calculations are based on the inflow volume, only.

4.3.4.3 Nutrient Fluxes in Biogas Digesters

By passing the reactor, chemical and microbial processes take place in the wastewater, and from the inflowing water stream, components are partly separated into the gas phase and into the sediment.

In a simplified mass balance (Figure 4-13) the following main flows are expected:

- Organic dry matter is transformed to gaseous methane and carbon dioxide to a large extent.
- Water and nutrients are conserved in the liquid and solid phase but may change their chemical form when passing the digester.

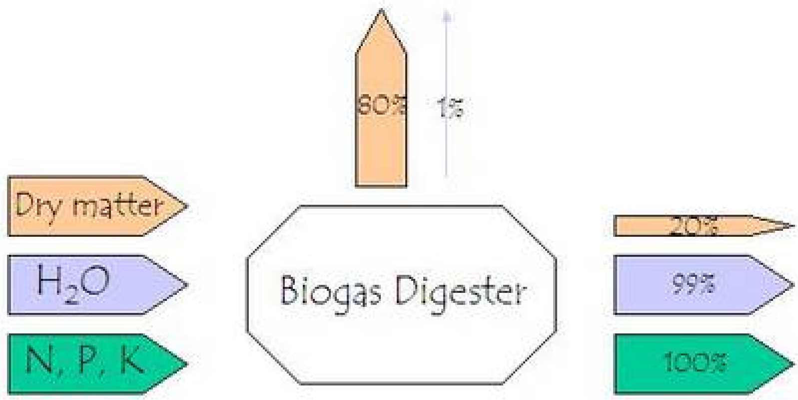

Figure 4-13: Expected mass flows in a biogas plant (percentages according UTEC 2006)

Flux balances are able to match situations of reactors in a steady state. This is due within long periods of constant management. In the short run, process parameters are changing,

inflows may vary and representative sampling is difficult (Pind *et al.* 2003).
Besides, in an anaerobic system, neither the amount nor the concentration of sediment can be evaluated easily. To evaluate, the process is going to be interrupted.

Therefore, an exact balance of fluxes can not be expected for a farmer´s system. The following chapter will nevertheless give details, basic calculations and bring the attention to processes that future studies should focus on.

For nitrogen the mean input was 550 g N d^{-1}, mean output 720 g N d^{-1}. Calculated per LU it results in 211 g N LU^{-1} d^{-1} the mean outflow 272 g N LU^{-1} d^{-1}. Nitrogen influx and outflux were correlated by correlation of paired samples ($r^2=0.797$, $\alpha<0.001$).
Although other authors also observed that the effluent and also the sludge contain a higher percentage of nitrogen and phosphorus than the same quantity of raw organic material fed inside the digester (Falk 2005), an increase of nitrogen mass within the process is not possible. So the difference might be due to release of formerly accumulated material, especially from the PT-digesters – which is an explanation for the COD results as well.

Nitrogen inflow and outflow were quite similar in the samples from FD-10 digesters (227 g to 234 g N LU^{-1} d^{-1}) but different at the PT-10 digesters (194 to 314 g N LU^{-1} d^{-1}).

Different to nitrogen, the total amount of phosphorus decreased from inflow to outflow: 190 g P LU^{-1} d^{-1} in the inflow and 135 g LU^{-1} d^{-1} in the outflow (mean of 20 examined digesters).

Correlation of P inflow and outflow was lower than for N ($r^2=0.527$ at $\alpha = 0.025$).

Similar to nitrogen, the P outflow of the PT digester was higher than the outflow of the FD digesters. For the PT digesters a small increase was balanced (130 to 167 g LU^{-1} d^{-1}), for the FD digesters a decrease occurred (241 to 110 g LU^{-1} d^{-1}).

This decrease indicates better sedimentation and precipitation conditions at FD digesters. The higher outflow of phosphorus at the PT digesters would fit to the theory that formerly accumulated sludge is released in the outflow. A hypothesis which is supported by the results on suspended solids which were reduced 69% in the FD-10 digesters and only 40% in the PT-10 digesters.

4.3.5 Potential Use as Fertiliser

Generally, a fertiliser supplies nutrients, organic substrate, alkalinity or plant supporting substances (chapter 3.1). Within the digestion process two substrates are generated that could potentially be used as a fertiliser: the liquid effluent and the muddy sludge.

4.3.5.1 Effluent

Nutrients

The effluent is a liquid substrate which is available daily, as it is leaving the digester regularly after feeding. It has low dry matter content (0.2% suspended solids) and contains the soluble part of the nutrients.

Nitrogen is an important element for plant growth. Depending on the kind of crop and the level of intensification, 60-200 kg ha^{-1} and even 300 kg ha^{-1} a^{-1} for vegetable are applied. For aquaculture nitrogen has to be supplied, too.

In the biogas digesters, the total nitrogen concentration varied from 120 mg l^{-1} to 810 mg l^{-1} in the effluent of 20 screened biogas plants. The mean was 380 mg l^{-1}; it was found to be slightly higher in the PT effluents compared to the FD plants (420 mg l^{-1} to 340 mg l^{-1}). The means were somewhat lower in the two monitored farms, e.g. at the FD-LT it was 203 mg N l^{-1}.

The nitrogen concentration had not been influenced by the amount of livestock or the amount of water used.

The main nitrogen fraction in the effluent is NH_4^+-N; counting for around 70% of the total nitrogen. The nitrogen to ammonium concentrations were correlated in the data set of the 20 biogas plants. In the FD-10, NH_4 counted for 75% of the nitrogen, in PT-10 digesters for 65%, respectively. The mean NH_4^+-concentration was 240 and 320 mg N l^{-1}.

The concentrations correspond to other data on plastic biogas digesters in the Mekong Delta where the effluents contained 95 to 497 mg NH_4^+-N l^{-1} (mean 327 mg l^{-1}) (Phan *et al.* 2003) and 37 to 467 mg NH_4^+-N l^{-1} (Suzuki 2002).

In studies in Cambodia (Thy *et al.* 2003a) an increase of NH_4^+-N was reported, resulting in 46-65% ammonia-N (percent of N_{tot}). This proportion had not been influenced by retention time (10, 20, 30 days) or loading rate.
For European biogas systems, the proportion of NH_4 is given with 65% of N_{tot} for digested pig manure-corn mix (KTBL 2007).
Slightly lower but still similar results have been obtained with dairy manure, too (Marchaim 1992): Jewell et al. (1976) found that the ammonia nitrogen increased from 38 to 45% of the N_{tot} during digestion. Similarly, Hart (1963) found an increase of from 24 to 49%.

This transformation may be an advantage as ammonium can be taken up by the plants directly whereas organic nitrogen needs to be mineralised by soil bacteria before uptake. It may be a disadvantage when conditions induce nitrogen (NH_3) losses (e.g. at high N-concentrations, application technique, weather and soil conditions). However, at the determined neutral or slightly acid pH and the fast infiltration speed ammonia losses can be kept low.

Despite the favourable NH_4^+/N_{tot} ratio, it must be noted that the nitrogen concentration in biogas effluent was less than 0.05%. This is quite low compared to mineral fertiliser or even slurry where ~1% N is expected (KTBL 2005).
Nevertheless, per year around 260 kg N were leaving each digester in the effluent. Per LU this is around 100 kg N.

Phosphorus is necessary for plant nutrition. It may be the most important parameter in marginal soils or when phosphorus is fixed in the soil e.g. due to low pH. The specific importance of phosphorus for SE-Asia was pointed out (Becker 2004; Zapata & Roy 2004).

P_{tot} was determined in the effluent of the 20 screened systems. The mean concentration was 180 mg l^{-1}, values ranging from 24 to 336 mg l^{-1} (CV 0.5). The lowest concentration determined (24 mg l^{-1}) might not be representative as it had been measured in an effluent with comparable low suspended solids and very low dry matter content.

Obviously the management and especially water use at the farms differs considerably – the phosphorus concentration was not influenced by the amount of livestock (r^2=0.02) or the amount of water used (r^2=0.02).

An interesting difference could be noticed between the effluents of the FD-10 and PT-10 digesters: the phosphorus concentration in the PT-effluent was close to its inflow concentration (194 mg P l^{-1}), whereas the FD-effluent contained only half the inflow

concentration (149 and 307 mg P l^{-1}).
Among various processes that are taking place in the digester, sedimentation and precipitation are possibly the important ones for this finding. FD digesters also reduced suspended solids more effectively, so in the outflow of FD digesters, there are less organic particles and consequently less P_{org} in the effluent.

This leads to the conclusion that in FD-digesters, more P is stored in the sludge. Biogas sludge of FD digesters will therefore contain a higher amount of P compared to PT digesters. The total amount of phosphorus will nevertheless be the same.

On average, 350 g P d^{-1} are released with the effluent. This way, 122 kg P a^{-1} are available in the average effluent of a farm or 46 kg P a^{-1} LU^{-1}.

Heavy metals

Several heavy metals belong to the group of micro nutrients that are necessary or helpful for plant growth (Finck 1991b). In high concentrations they are toxic to plants, and to organism of higher trophic levels.

- Zinc is needed for activating enzymes in the plant. The effluent contained concentrations up to 60 µg l^{-1}.
- Copper is needed in enzymes, for photosynthesis and chlorophyll formation. The mean copper concentration was 55 µg l^{-1} and the maximum value 160 µg l^{-1}.
- Nickel is contributing to N_2-fixation. The highest nickel concentration measured in any of the samples was 40 µg l^{-1}.
- Cadmium is not regarded essential for plants. It was below detection limit in almost all samples. It could be detected on one sampling date; concentrations were just above the detection limit.
- Chromium is not regarded essential for plants but necessary for humans and animals. It was below detection limit in most samples, too. It could be detected in a few samples with a maximum of 1.82 µg l^{-1}.

For the risk assessment, not only the concentration – the acute toxic effect – has to be regarded. For long term effects, the yearly input and the potential accumulation in the soil is important. Therefore, different trigger values for short and long term application of irrigation water were set up (ANZECC 2000), shown in Table 4-18.

Table 4-18: Maximum heavy metal concentrations in effluents compared to long- and short-term trigger values (LTV, STV) for irrigation water (ANZECC 2000)

Element	Maximum effluent concentration in mg l^{-1}	Long-term value (irrigation 100 a) in mg l^{-1}	Short-term value (irrigation 20 a) in mg l^{-1}
Cadmium	n.d.	0.01	0.05
Chromium	0.002	0.1	1
Copper	0.160	0.2	5
Nickel	0.040	0.2	2
Zinc	0.060	2	5

n.d. not detectable

The accumulation and the availability of heavy metals also depend on the soil type or grain size, respectively. This is reflected e.g. in the German Federal Soil Protection Ordinance (BBodSchV 2004) where the tolerable charge for clay or loamy soils is higher compared to sandy soils.

By calculating the "input per nutrient equivalent" – i.e. the amount of heavy metals that is added with the organic substrate when applying a standard amount of nutrients per ha – a more specific comparison of different substrates is possible (Hammer & Clemens 2007; Rieß 2003).

Salts

High salt contents in soil or irrigation water may harm crops, may lead to soil degradation and pollution of groundwater. Depending on the site specific conditions (plant tolerance, soil conditions, climate, management, water supply) tolerances will differ.

Salt content can be determined by electrical conductivity as a sum parameter. Not all ions are affecting the plants the same way, so for detailed analysis the specific ions have to be analysed, e.g. chloride, sodium, bicarbonate (ANZECC 2000). However, according to Cardon *et al.* (2008) effects may arise at 1.2 mS cm^{-1} in soil, and water with EC >1,500 µS cm^{-1} should not be used for irrigation purposes (Lenntech 2006).

According to this classification shown (Table 4-19), the effluent of the examined biogas digesters which had EC values from 1.7 to 6.4 mS cm^{-1} (mean 3.4 mS cm^{-1}) is classified as water with high or very high salt content.
These high salt concentrations are partially due to the groundwater used, which has an initial value of 710 µS cm^{-1} (FD-LT) and 1360 µS cm^{-1} (PE-AB) already.

If use can not be avoided, good management and control is essential, e.g. leaching, artificial drainage (Cardon *et al.* 2008). Furthermore, the salt tolerance of the plants has to be regarded (ANZECC 2000).

Table 4-19: Electrical conductivity (EC) of biogas effluent and groundwater compared to the salt content classification of irrigation water by total dissolved solids (TDS) (Lenntech 2006)

	Measured values EC (µS cm^{-1})	Classification EC (µS cm^{-1})	Classification TDS (ppm or mg l^{-1})	Classification salt content
Groundwater LT	710	<750	<500	Low
Groundwater AB	1,360	750-1,500	0-1,000	Medium
Biogas effluent min.	1,700	1,500-3,000	1,000-2,000	High
Biogas effluent max.	6,400	>3.000	>2,000	Very high

In the Mekong Delta biogas effluent may be diluted with pond or canal water with lower salt concentration of around 200 µS cm^{-1}. Depending on the effluent concentrations, dilution rates should be at least 1:1 up to 1:10.
These numbers may be used as a first rule of thumb, for a detailed consideration fluxes have to be regarded, i.e. the applied yearly amount, the uptake by plants, evaporation, percolation, etc. which is beyond the topic of this work.
When effluent is discharged into ponds and used for enhancing aquaculture, dilution occurs automatically.

Pathogens

To minimise the risk of infection when using waste water for irrigation in agriculture, the WHO has developed a DALY, a health based target that target can be achieved by a combination of waste water treatment and other health protection measures, e.g. an estimated 3-4 log unit pathogen reduction by natural die-off in the field and by domestic washing (WHO 2006a, 2006b). Depending on the intended use, different targets have to be reached which require different reduction rates. Table 4-20 shows that for unrestricted irrigation and cultivation of vegetables higher standards are necessary than for drip irrigation where the water is places below the crop.

Table 4-20: Health based targets for treated waste water use in agriculture ((WHO 2006a)

Intended use	DALY per pers. and year	$\log_{10}$ pathogen reduction needed	No. of helmith eggs per litre
Unrestricted irrigation	$\leq 10^{-6}$		
Lettuce / Onion		6 / 7	≤ 1
Restricted irrigation	$\leq 10^{-6}$		
Highly mechanised / labour intensive		3 / 4	≤ 1
Localised (drip) irrigation	$\leq 10^{-6}$		
High / low growing crops		2 / 4	No recomm./ ≤ 1

Consequences for Agricultural Production

The use of effluent in aquaculture seems to be the most common practice in the Mekong Delta; it was done at the monitored farms, too. Few farmers applied effluent to their fields (Watanabe 2000), sometimes effluent reached orchards or rice fields via irrigation canals.

The use of biogas effluent has been reported for different cultures and species in the region and the neighbouring Cambodia (Table 4-21). In these studies biodigester effluent was usually found to be superior to raw manure which might be due to a better availability due the higher ratio of NH_4^+ to N_{org}.

Table 4-21: Experienced use of biogas effluent: cultures, species and source

Culture	Species	Source
Fish culture	Tilapia (*Tilapia* sp.)	(Thy *et al.* 2003b)
	Tilapia (*Tilapia* sp.), Silver carp (*Hypophthalmichthys molitrix)*, Bighead carp (*Hypophthalmichthys nobilis)*, Silver barb, Mrigal	(Sophin & Preston 2002)
	Tilapia (*Tilapia* sp.), Carp (*Cyprinidae),* Gourami (*Osphronemidae)*	(Own observations at numerous farms in CT, An Giang, Tra Vinh)
Algae production	Diverse algae	
Water plants	Duckweed (*Lemna* sp.)	(Chau 1998b; Rodríguez & Preston 1996)
	Water hyacinth (*Eichhornia crassipes)*	(Moorhead *et al.* 1990)
	Water spinach (*Ipomoea aquatica var. reptans*)	(Hiep & Preston 2006; Sophea & Preston 2001)
Plant cultivation	Cassava (*Manihot esculenta*)	(Chau 1998a)
	)rchard, e.g. Banana (*Musa* sp.), ongan (*Dimocarpus longan*)	(own observation)

In several studies biogas effluent proved to be a better fertiliser than raw manure:

- Fish yield was higher in ponds that had received effluent, compared to mineral fertiliser or manure (Sophin & Preston 2002).
- In duckweed (*Lemna* sp.) higher crude protein concentrations were found when grown with digester effluent. 35-40% crude protein makes duckweed a valuable supplement for pigs and poultry (Rodríguez & Preston 1996).
- Higher yields could be achieved when applied to cassava (Chau 1998a, 1998b).

Other studies found similar effects between effluent and mineral fertiliser (Hiep & Preston 2006; Sophea & Preston 2001) or manure (Hiep & Preston 2006).

Recommendations for application

The results on heavy metals in biogas effluent do not demand further actions. The microbiological results, however, restrict application (Table 4-15, Table 4-16, Table 4-20) to crops.
Furthermore, the measuered salt concentrations were quite high for direct application to crops. The used groundwater already contains high salt concentrations. To improve the situation, the effluent may be diluted with canal or river water.
It has to be kept in mind that application via irrigation canals will distribute nutrients not uniformly in the fields. If the necessary logistics are available, pumping is a possibility for better distribution.

The direct use of effluent in aquaculture can not be recommended as the microbiological results revealed high bacterial concentrations. Microorganisms may be taken up by the fish. Anyhow, fish from wastewater ponds should not be consumed raw.

4.3.5.2 Sludge

When operating a reactor, sediment accumulates in the bottom. This sludge is generally regarded as a valuable fertiliser (Ortenblad 2005; Winblad & Simpson-Hébert 2004).

It has a high nutrient content and – compared to farm yard manure – reduced carbon content. If digested sufficiently, anaerobic sludge is without smell (Kaltwasser 1980).

The biogas sludge is not released regularily, it is removed only after some years when the digester has filled up. At the concrete fixed-dome-reactor, in addition the “overflow sludge” in the second tank has to be removed every few months.

Of the 22 interviewed households, several had not emptied their tank for 5 and more years. In other digesters, accumulation had been less than one year. Figure 4-14 illustrates the “emptying praxis”.

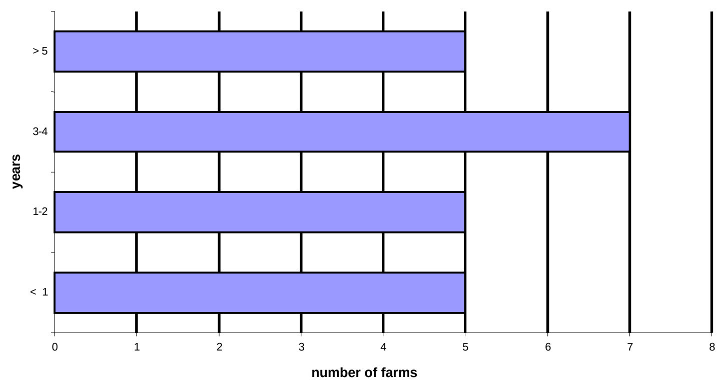

Figure 4-14: Storage time of biogas sludge in digesters (n=22)

Wieneke (2005) could not identify a common regular practice of emptying but more than 90% of the interviewed households were using the material. Two third (67%) of the interviewed farmers were using the sludge material for own cultivations, e.g. to fertilise their fruit trees or flowers. Among them, 10% were mixing the biogas sludge with ash or mineral fertiliser. An additional 12% sold or gave it away as fertiliser. The others have been emptying it into the fishpond (14%) or into the public canal (4%).

At the examined farms AB and LT, biogas sludge was used as a fertiliser for orchard trees: during the time of the study, a new Banana (*Musa* sp.) field was established in AB. Existing Longan trees (*Dimocarpus longan)* in LT received the sludge.

Direct use of biogas sludge minimises nitrogen and carbon losses, but bears a higher risk of pathogen transmission.

The applied sludge had a nitrogen concentration of 20 g N kg^{-1} dried sludge (LT, 17.10.03). With an 80% water content, 250 kg of sludge supplied 1 kg N.

This water content (~ 80-90%) makes transportation difficult; huge amounts of substrate are necessary to supply the fertiliser demand. Marchaim (1992) summarised necessary masses for the main substrates (Table 4-22).

Table 4-22: Estimated quantities of manures or fertilisers needed to supply 1 kg nitrogen (Marchaim 1992)

Fertiliser type	Demand (kg)
Anaerobically digested cattle dung sludge (wet)	676
Cattle dung (fresh)	345
Anaerobically digested cattle dung sludge (dried to 10% of wet weight)	80
Ammonium superphosphate	33
Ammonium phosphate	9
Urea	2

The low nutrient concentration of substrates from waste water is a major impediment for application of sludge and slurry. A reduction of water and volume can be achieved by different further treatments, e.g. by composting ("hot rotting") or by vermicomposting ("cold rotting"). These potential further treatments for biogas sludge will be examined in the next chapters.

4.3.6 Economic Aspects

Due to the high importance of economics for further implementation, the main aspects are mentioned, although an economic evaluation was not the focus of this study.

4.3.6.1 Business Economics

Anaerobic treatment not only improves water quality but – in some prospects even more important – produces energy. Per m^3 biogas around 4,500-5,500 kcal m^{-3} heat energy are produced (KTBL 2005). Calculating a 60% efficiency of specific burners for methane 1 m^3 i.e. 2,500 kcal m^{-3} would heat up 250 l H_2O 10°C, respectively 25 l from 0 to 100°C. A typical household in SE Asia produces 250 to 300 m^3 biogas per year in an unheated digester of 6-10 m^3 (Hawkes 1986).

According to Chinh (2002) farming households spend around 750,000 VND a^{-1} for fuel. SNV calculates € 10 per month saving on fuel. Accordingly, the break even point will be reached within 2.5 years (SNV Vietnam 2008a). SNV also calculates saving of workload of 1 – 1.5 hours per day per household. Li (Li 1987) estimates a bit less, 2.5 days/month. This time may be used for other jobs.
Several authors describe higher yields when applying biogas slurry (Marchaim 1992; SNV 2008). However, the disadvantage is a higher application effort. Less cost for commercial fertiliser and a possibility of "buffering" market prices are an advantage.

4.3.6.2 Further Effects and National Economics

Vietnam´s economy has a high energy demand which can hardly be satisfied at the moment. Supposing a continuing economic growth, Vietnam needs to increase its energy sources. Biomass accounts for 60-65% of the national primary energy consumption, in private households it is 80-90%, mainly firewood and agricultural residues (Pressea 2000).

Be wastewater treatment, the construction and use of biogas digesters also adresses deforestation which is a severe problem in various areas of Vietnam.

Methane has a strong greenhouse effect and contributes ~ 20% to global warming. The effect of CO_2 is less but due to the large amounts produced it is responsible for ~ 60% (SBGF *et al.* 2004). These green house gas emissions can be reduced by using biogas as - compared to other fuels - it shows positive emission data (Klingler 1998).
SNV calculates that per household biogas plant and year, 5 ton CO_2 can be saved (SNV Vietnam 2008a). The methane emissions, 6 Tg a^{-1} in Asia (of 28 Tg a^{-1} worldwide) could be compensated by biogas digesters to around 3 Tg a^{-1} thereby nearly half the amount (Klingler 1998).

The reduced emission of one plant builds up five tradeable emission rights per year with a price of € 5 – 10 per ton CO_2 (SNV Vietnam 2008a).

A demand for biogas plants will create new enterprises for biogas plant construction, employment for labourers, and increasing the gross national product (SNV Vietnam 2008a).

4.4. Conclusions

Wastewater of farming households - being the main type of settlement - originates from pig sties mainly. Due to management practice, animal waste is diluted notably.
Human sanitary waste is another source of wastewater. In rural areas the main path for discharge is directly via fishponds, although this practice is reducing.
In rural areas, organic solid waste is often used for feeding animals and therefore only contributing a minor part to wastewater pollution.

Regarding the operation of local biogas digesters as a wastewater treatment system, the following can be stated:

- The existing digesters are working rather stably with pig slurry. Biogas is produced continuously and in most cases sufficiently for family cooking.
- By passing the fermenter, wastewater quality is improved by reducing BOD_5, COD, turbidity as well as pathogens. Odour is reduced.
- As a single treatment it is not sufficient to match Vietnamese surface water thresholds.
- Treatment efficiency was not optimised at the examined sites. Both can be increased by longer HRT (leading to higher C- and pathogen removal). Additional treatment steps are necessary, e.g. pond, wetland or constructed wetland afterwards.

Regarding the differences between concrete fixed-dome and plastic-tube reactors the following conclusions are drawn:

- The cost of the systems is different (FD digesters 10 times higher), the expected time of operation as well.
- The C-reduction was higher in the fixed-dome reactor compared to the plastic-tube digester due to the longer HRT but may also be influenced by the different flow properties in the digesters.

Both types of biogas digesters can be recommended as one step in wastewater treatment (but are not sufficient as a potential single treatment). Positive side effects are the reception of cooking gas and fast available nutrients in fertiliser.

Implementing other digester types (e.g. a two-step reactor) may improve slightly but not considerably the digestion of pig manure and human excrements. On the other hand, if other material is added, e.g. rice straw, this might be a more recommendable solution.

Anaerobic digestion contributes to reduce the global warming if compared to other treatment systems like anaerobic ponds where the CH_4 and CO_2 is not collected usually.

As a result of their studies in the Mekong Delta, Watanabe & Nagumo (2002) specified that rice straw and animal waste could be used more effectively although recycling of farm by products in the Mekong Delta was already quite effective.
To face increasing intensification several authors advise to pay attention to the nutrient fluxes to keep the sustainability (Kranert & Hillebrecht 2000; Suzuki 2002).

The use of animal and human excrements for biogas production and as a fertiliser is a step towards sustainability.

The resulting products of biogas plants - effluent and sludge - contain plant nutrients and can be used as a fertiliser. However, due to high concentrations of hygienic relevant microorganisms in the effluent, it is recommended to use it only after further treatment, e.g. in a maturation pond or in a wetland.

High water content and low nutrient concentration of effluent emphasises use in aquaculture to enhance primary production. The lower carbon content compared to the material prior to digestion makes the effluent even more attractive for pond systems due to the reduced danger of oxygen depletion.

However, the lower C/N ratio (carbon loss during digestion), the increased plant-available mineral nitrogen, and contained micro nutrients allow its use as a fertiliser for plants.

Sludge which may also contain significant numbers of pathogens, should be applied directly only at sites where the risk to consumers is marginal, e.g. at ornamental plants or tree nurseries. This also applies to fruit trees. Use for other food crops may be possible after treatment by composting or vermicomposting, which is discussed in the next chapters.

5. Waste Water Ponds

5.1. Introduction

Waste water ponds are a complex system where a lot of processes influencing one another take place. Ponds can achieve a complete wastewater treatment – if done by a "pond-cascade" (Mara 1997):

- Wastewater stabilisation pond
- Anaerobic pond
- Facultative pond
- Maturation pond.

5.1.1 Process

For wastewater improvement several processes are responsible:

- sedimentation of solids and anaerobic digestion (in stabilisation ponds)
- aerobic chemical and microbial transformation (in maturation ponds)
- nutrient uptake by algae, by microorganisms, plants and fish.

Organic load, measured as BOD and COD is removed due to the transformation of C-compounds to CH_4, CO_2, or biomass.

Nitrogen (N_{org}) will mainly be hydrolysed to ammonium, taken up by algae and move up to further trophic levels (20% of the N in algae will be deposited as non-biodegradable algal cell at the ground) (Mara 1997). Depending on the environmental conditions (pH, O_2), gaseous emissions (NH_3, N_2, N_2O, and others) will reduce nitrogen concentrations in the water.

Phosphorus will be taken up into algal biomass. It might be precipitated as inorganic phosphorus and settle into the sediment (Mara 1997). According to Mara (1997) BOD and P removal are correlated: if the BOD removal is 90%, the P removal will be ~45%.

Removal of pathogens takes place due to sedimentation and die-off (storage time and temperature), high pH (>9), and photo-oxidative reactions (Mara 1997).

When digester slurry is used in ponds, the nutrients stimulate the growth of both phytoplankton (algae) and zooplankton (*Daphnia* sp. and Crustaceae) which the fish harvest (Marchaim 1992).

5.1.2 History and Common Techniques

Man has exploited the self purification process of natural water bodies polluted with organic matter for many centuries (Pearson 1987). Ponds are the cheapest form of treatment, both to construct and operate. Maintenance is very simple and high efficient (Mara 1997). In ponds, about 10 times as much as in irrigation fields (sewage fields) can be processed (Dunbar 1912, reprint 1998).

Pond systems for waste water treatment are common around the world, not only in developing but also in developed countries. One of the biggest systems is treating the waste water of Melbourne.

Raising fish in ponds supplemented with human and animal excreta has a long tradition in China and other Asian counties (Feachem *et al.* 1983; Mara 1997; Marchaim 1992). It has been practiced in ancient Egypt and was widely used in European monasteries in the middle

age (Feachem *et al.* 1983). Fishponds have been used in Germany in the 19th and 20th century for treatment of waste water after sedimentation, thereby supplying 2,000-3,000 PE per hectare (Dunbar 1912, reprint 1998).
The largest area of wastewater-fed fishponds, ~3000 ha located east of Calcutta, started around 100 years ago by the local fishermen. Some effluent is used for crop irrigation and some to cultivate of rice (Mara 1997).

A wide range of fish species has been cultivated in aquaculture ponds receiving human waste (Pescod 1992). Especially carps, e.g. common carp (*Cyprinus carpio)*, or Indian major carp), and tilapia (*Tilapia* sp.) grow rapidly (Feachem *et al.* 1983).

In southern China where cultivation of fish in ponds is common, digester slurry has been used as a feed supplement in recent years. This increased fish production and decreased costs for feed (Marchaim 1992). The treatment in a digester already reduces the pathogen load.

However, these practices – using wastewater, excreta or biogas sludge directly for raising fish comprises health risks. In a waste water pond system consisting of several cells, fish growth is recommended only in maturation ponds (Feachem *et al.* 1983).

5.1.3 Practice in the Mekong Delta

Excreta-fed fishponds are common in the rural area of the Mekong Delta. Estimated 60-70% of the households are using fishpond toilets (Soussan *et al.* 2005; Wieneke 2005; Wohlsager *et al.* 2008). Besides, swine slurry is fed to the fishponds frequently. Poultry cages above the fishponds are found regularily (Hanh *et al.* 2007; Man 1991)

Most of the households owning a biogas digester are collecting the biogas effluent in a fishpond (Wieneke 2005). Sometimes the effluent is discharged to an irrigation canal, too.

The main fish raised in fishponds are tilapia, gouramis and carps.

5.1.4 Research Questions

When looking at the recent practice in the Mekong Delta - ponds as recipients of waste water from biogas digesters - the following questions were asked:

- What improvement is achieved by the existing ponds for the relevant waste water parameters? Does the management need improvement?
- Is the composition of the produced outflow the pond sludge suitable for agriculture?

5.2. Material and Methods

Ponds were examined in both farms of the monitoring, in LT and AB. At both farms, the biogas effluent was running into a fishpond. In LT one pond of 600 m^2 and in AB two fishponds with a size of 700 and 400 m^2, received the effluent. The ponds had a depth of around 1 m, each and received additionally canal water when necessary. Ponds were emptied once a year after fish harvest to take out sludge.

Every time, the biogas digesters were examined, the succeeding pond was checked, too. Samples were taken in the pond, close to the outlet. Field parameters were measured on site, further analysis of samples carried out in the laboratory, according to the biogas samples (see chapter 4).

The pond samples were gained and processed with the support of T.K. Tien, Ms. Van, V.T.Y. Phi, L.H.Viet from Can Tho University, and with supervison of A.Rechenburg from Bonn University for microbiologial analysis.

5.3. Results and Discussion

5.3.1 Field Parameters

Table 5-1 shows the field parameters measured in the ponds during the monitoring.

Table 5-1: Field parameters in ponds receiving biogas effluent in the monitoried farms in An Binh and Long Tuyen (FD-LT, PT-AB)

n=12	Temp °C	EC µS cm^{-1}	pH	DO mg l^{-1}	Turbidity	Redox mV, µV
P-AB1	28.3	375	7.2	4.6	43.0	78.7
P-AB2	28.3	347	6.9	3.7	43.5	88.6
P-LT	28.9	208	6.7	2.3	54.2	63.3

Temperature was around 28-29°C and varied only slightly during the year, with lowest values in December and highest around April.

Electric conductivity was very constant throughout the whole year of monitoring. EC was significantly lower in the pond in LT compared to the one in AB.

The pH values were close to neutral pH, and ranged from 6.7 to 7.2. The pH in the first pond was usually higher than in the second. This pH range is suitable for raising fish (Table 5-2); an optimum range is given from 6.5 - 8.5. The pH should not drop below 4.8 - 5.0 and pH levels higher than 9.2 put an end to salmonoids, higher than 10.8 also to cyprinids.

Dissolved oxygen (DO) was 4.6 mg l^{-1} and 3.7 mg l^{-1} in the ponds in AB and 2.3 in LT mg l^{-1} (mean). DO varies with the photosynthetic activity almost directly; thus being lowest at night and rising till the afternoon (Crites *et al.* 2006). The samples were usually taken at 7:30 to 8:00 a.m.
The sensitivity of fish to low DO levels varies with species, life stage (eggs, larvae, adults) and life process (feeding, growth, reproduction). A minimum constant DO concentration of 5 mg l^{-1} is considered satisfactory, although an absolute minimum consistent with the presence of fish is probably less than 1 mg l^{-1} (Alabaster & Lloyd 1980). Bliefert (1997) mentions 4 mg O_2 l^{-1} as “fish critical value”. According to Svobodova *et al.* (1993), salmonoids´ optimum is 8 - 10 mg l^{-1}, below 3 mg l^{-1} they suffer of suffocation. For cyprinids the optimum levels are given with 6 - 8; suffocation starts below 1.5 - 2 (Svobodova *et al.* 1993). Fish cultured in waste-fed ponds appear to be able to tolerate very low DO concentrations, for at least short periods of time. Air-breathing fish are the most tolerant, less tolerant are (by increasing order) tilapia, carps, canal catfish and trout.

The recommended values of the Australian and New Zealand Environment and Conservation Council (ANZECC 2000) are listed in Table 5-2.

Table 5-2: Physico-chemical guidelines for the protection of aquaculture species (ANZECC 2000)

Measured parameter (mg l^{-1})	Freshwater production	Saltwater production
Dissolved oxygen	>5	>5
pH	5.0 – 9.0	6.0 - 9.0
Salinity (total dissolved solids)	<3000	33 000 – 37 000 (3000–35 000 Brackish)
Suspended solids	<40	<10 (<75 Brackish)
Temperature	<2.0°C change over 1 hour4	<2.0°C change over 1 hour

Compared to a direct discharge from the pig sty, the biogas effluent contains less carbon. Therefore the risk of oxygen depletion in the pond is reduced. Nevertheless, there might still be a risk if ammonia (NH_3) concentrations get too high.

5.3.2 Biological and Chemical Oxygen Demand

Degradation of organic substances takes place by consumption of oxgen. The biological oxgen demand represents the fast degradable fraction used by microorganisms for their existence, the chemical demand is the total amount of oxygen necessary to oxidise all substances present.

The COD and BOD_5 concentrations were measured in the ponds where biogas effluent from the plastic tube digester in An Binh (PT-AB) and the fixed dome digester in Long Tuyen (FD-LT) was discharged.
The results achieved during the monitoring are shown in Table 5-3. The concentrations were about 1% of the COD and about 2.5% of BOD concentration in the biogas effluent.

Table 5-3: Mean COD and BOD concentrations measured in ponds in An Binh (P-AB1 and P-AB2) and Long Tuyen (P-LT) during monitoring (n=12)

n=12	COD mg l^{-1}	BOD_5 mg l^{-1}
P-AB1	38	11.0
P-AB2	31	10.9
P-LT	25.5	13.0

These values are in the range of treated municipal waste water. Surface water has usually still lower concentrations, e.g. <6 mg l^{-1} BOD for surface water of good to medium quality (Bahlo & Wach 1992; Bliefert 1997).
However, no major threats are expected by these COD and BOD concentrations if this water is discharged to the canal or any other surface water. In the tropical climate residual carbon molecules will be degraded quite quickly. According to the physico-chemical guidelines of ANZECC (Table 5-4), the water is suitable for freshwater aquacuture.

Table 5-4: Physico-chemical guidelines for the protection of aquaculture species (ANZECC 2000)

Measured parameter (mg l^{-1})	Freshwater production	Saltwater production
Biochemical oxygen demand (BOD_5)	<151	n.d.
Chemical oxygen demand (COD)	<401	n.d.

n.d. not determined

5.3.3 Microbiology

In the ponds the effluent concentrations of *E. coli* and coliform bacteria were reduced further. However, the incoming surface water was always strongly contaminated microbiologically, with values of 1.000 – 10.000 *E. coli* 100ml^{-1}.

The number of helminth eggs discharged with the effluent also gets reduced in the pond. Here especially sedimentation to the bottom sludge is important (Rechenburg *et al.* 2005). Although the intestinal pathogens of humans and animals are not the normal flora of fish, there may be a passive intake of those. A number of helminthic pathogens can use fish as intermediate hosts. *Clonorchis sinensis* (Chinese liver fluke), *Opistorchis viverrini* or *O.felineus* (Cat liver flukes) is associated with excreta-fed fishponds.

Clonorchis sinensis is one of the major sources of porcine and human helminth infections as fish ponds are an ideal habitat and the necessary hosts of water snails, fish and mammals are available *(Thien et al. 2007)*. Vietnam is named as one of the countries where the prevalence in the population can reach 60% (Feachem *et al.* 1983) due to direct supply of excreta without previous settling of worm eggs.
Other helminths having fish as intermediate host, like *Diphyllobothrium latum* (fish tapeworm) are infecting only river fish or *Heteropyhs heterophyes, Metagonimus yokogawai.* They are of no major public health importance (Feachem *et al.* 1983)

The study of Thien *et al.* (2007) showed further that fish raised in the VAC system in the Mekong Delta are often infected with trematodes.

Fishermen might also be infected with *Shistosoma* sp. when the intermediate host are present in the pond.

The removal of pathogens in ponds is due to natural die-off, predation, sedimentation, and absorption. It is very effective in multiple cell pond systems for worm eggs, bacteria and virus. At normal pond systems with 20 retention time, three cells have little risk of parasitic infection from pond effluents (Crites *et al.* 2006).

5.3.4 Nitrogen

The total nitrogen concentration (Kj-N+NO_3) in the pond samples was 3.7 and 3.5 mg l^{-1} in AB and 3.3 mg l^{-1} in LT, about half of it being organic nitrogen. The ammonium concentration in the samples was usually around 1 mg l^{-1} (see table 8). It was 2 mg l^{-1} only twice in AB pond 1 and once in LT.

Table 5-5: Nitrogen concentrations measured in ponds during monitoring at farms in An Binh (P-AB1, P-AB2) and Long Tuyen (P-LT)

n=12	NH_4 mg l^{-1}	NO_3 mg l^{-1}	N_{tot} mg l^{-1}
P-AB1	1.1	0.6	3.1
P-AB2	0.8	0.4	3.1
P-LT	0.8	0.7	2.6

Unionized ammonia (NH_3) is toxic to fish in the concentration range 0.2 - 2.0 mg l^{-1} (Alabaster & Lloyd 1980). However, the tolerance of different species of fish varies, with tilapia species being least affected by high ammonia levels. Bartone et al. (1985) found in his studies in Lima, Peru that satisfactory growth and survival of tilapia was possible in fish ponds fed with tertiary effluent when the average total ammonia concentration was less than 2 mg l^{-1} and the average unionised ammonia concentration was less than 0.5 mg N l^{-1}, with the latter only exceeding 2 mg N l^{-1} for short periods.

5.3.5 Reuse of Pond Water and Sludge for Agriculture and Aquaculture

5.3.5.1 Reuse of Water

At the examined farms, ponds were used for growing fish. From time to time grown algae were removed from the water surface and used as fodder. In this study, no details on fish yield or nutrient transfer rates were evaluated.
In a study in Cambodia, it was found that all of the five fish species examined (tilapia, silver carp, bighead carp, silver barb and mrigal) grew faster in ponds fertilised with biogas effluent than with manure (Sophin & Preston 2002). The degree of response was highest for silver carp, then bighead carp and tilapia and least for mrigal. The net fish yield was 55%

greater in ponds fertilized with bio-digester effluent rather than with fresh manure. The improvement with effluent compared with chemical fertiliser was 27%.

Alviar & *al.* (1980) investigated the growth of fish in an integrated farming scheme in the Philippines. The average yield of tilapia was 25 kg m^{-2} every two months resulting in 19 t $ha^{-1}a^{-1}$.

Growing biomass is an interesting and safer alternative to growing fish in sewage ponds. Algae might be harvested directly and be used as protein rich fodder (Feachem *et al.* 1983; Phuc 2000). Duckweed *(Lemna* sp.*)*, or water spinach *(Ipomea aquatica)* are used as forage for pig, poultry, duck and fish (Phuc 2000).
Barnard mentions the growth of algae for producing fuel. More than 10,000 pond systems in the US are treating domestic and industrial waste (Barnard 2007).

Pond water may be used for irrigation purposes too. Due to low nutrient concentrations found in the examined ponds, the fertilising effect will be marginal.

5.3.5.2 Reuse of Pond Sludge

At the examined farm, pond sludge was used for establishing a new banana orchard next to the pond. This growth was apparently.

As pond sludge contains sedimented microorganisms, careful use of the material is recommended. Application for orchard trees is regarded as quite safe although the farmer has to be aware of and take basic hygienic means.

Composting may be a further treatment for pond bottom sludge. However, an attempt of composting the sludge with rice straw – similar to composting pig manure - had not been successful. Obviously, the organic matter content was too low, leading to a dry-out and agglomeration to a hard chunk.

5.4. Conclusions

Ponds are a cheap form of wastewater treatment, both to construct and operate; maintenance is very simple and high efficient (Mara 1997). However, for proper treatment, a relative large area is needed and several succeeding ponds are necessary.

The combined biogas plant–pond system combines the advantages: in contrast to entire pond systems, gaseous losses can be avoided and instead used as biogas.

The combined system leads to considerable water improvement by reducing carbon load and microbiological parameters.

However, the first ponds should be restricted to growth of algae or biomass and used as fodder or green manure. Fish breeding may be done in subsequent ponds to avoid transmission of pathogens.

6. Composting of Biogas Sludge

6.1. Introduction

Digested biogas sludge contains valuable nutrients that may be reused in agriculture. However, when transporting the sludge or slurry to the fields difficulties arise due to its high water content. Even if it is applied wet in the field, tilling is difficult. Besides, hygienic parameters may not recommend unrestricted use (see chapter 4.3.3.3 and 4.3.5).

One solution to overcome these difficulties may be composting the slurry (Marchaim 1992).

To find out about the feasibility and the main issues related to composting of biogas sludge in the Mekong Delta, this study was carried out. The following chapter will summarise important fundamentals of composting, explain the results of the experiment and propose treatment options.

6.1.1 Process

Composting -from Latin "compositus", combined- is the biological decomposition of organic substrates to carbon dioxide, water and biomass under aerobic conditions, usually encompassing mesophilic and thermophilic phases caused by self-produced heat. Via intermediate products, a stable humus, an earth-like substrate is produced that is free from offensive odours and can be used as an organic fertiliser or soil-conditioner (Haug 1980; Riddech *et al.* 2002). Within the process, viable pathogens and weed seeds are destroyed (Idelmann 2005).

The process develops in a characteristic way. The different phases and their characteristics resumed by different authors are summarised in Table 6-1 (Amlinger *et al.* 2005; Anh 2002; Bidlingmaier 2000; Jenkins 2005).

Table 6-1: Characteristics of different process phases in composting

Phase	Starting phase	"Hot-rotting" phase	Cooling phase	Curing phase
Status	Organic raw material	Destruction	Main rottening phase	Maturation
Heat development	Mesophilic 40 - 52°C	thermophilic > 55° - >70°C	30° - 55°C	Cooling down < 45°C
Moisture	> 55%	55 – 70% wet mass	45 – 55/60% wet mass	35 - 50% wet mass
Organic substance	Plant residues and manure; high molecular substances; sugar, fats, proteins, cellulose, lignin	Mineralisation of low molecular substances, destruction of high molecular substances	Destruction of long aliphatic polymers (hemicelluloses, cellulose); beginning of lignin destruction, formation of ligno-proteins and humic substances	Lignin destruction; stabilisation; formation of humic substances, clay-humic-complexes
Nutrients	Initial material is characterised by C/N-ratio, water, air-pore-volume, and nutrient content	N-losses (NH_3); NH_4^+ formation; formation of H_2S	N-stabilisation: Assimilation in bacteria and fungi; increase of NO_3, removal of NH_4^+	Organic binding of N to humic substances, decrease of C/N ratio
Energy	Biochemic, molecular	New organisms and heat	New organisms and heat	
Organisms	Mesophilic bacteria	Bacteria mixed flora, thermophilic organisms	Bacteria and fungi; actinomycetes and first animals	New animals (e.g. springtails, earth-worms), fungal flora

Although composting is an age-old technique and a long-standing process, little is known about the microorganisms involved and their role in the composting process (Riddech *et al.* 2002; Sundh & Rönn 2002; Tiquia 2002).

In general, the decomposition is performed primarily by facultative and obligate aerobic bacteria, actinomycetes, yeasts and fungi. In the cooler initial and ending phases they are supported by a number of larger organisms, such as springtails, ants, nematodes and oligochaete worms (Anh 2002; Bidlingmaier 2000; Sulzberger 1989).

The heterotrophic organisms are using the energy released by the breakdown of organic molecules (carbohydrates, proteins etc.) and the carbon compounds for their cell building. The example of glucose may explain the process (Bidlingmaier 2000; Haug 1980):

$$C_6H_{12}O_6 + 6\ O_2 \dashrightarrow 6\ CO_2 + 6\ H_2O \qquad \Delta G = -2870\ kJ\ mol^{-1}$$

The energy (ΔG, Gibb´s free energy) is partly released as heat, another part is used by organisms (Bidlingmaier 2000). The total energy gain for degradation of glucose is the same in aerobic and anaerobic degradation, except that in the anaerobic process it is separated into two steps: building of CO_2 and CH_4 the first step (– 132 kJ mol^{-1}), and CH_4 combustion in the 2nd step (- 2,671 kJ mol^{-1}) (Amlinger *et al.* 2005).

Aerobic microbes decompose waste at a faster rate than their anaerobic brethren. The amount of microorganisms produced is higher, 63 g per 100 g oDM in the aerobic compared to 9 g biomass in the anaerobic process. Even lignin can be degraded in the aerobic process although at lower rates than proteins, oil/fat and carbohydrates (Bidlingmaier 2000).

Besides the material itself, several other factors are determining the rate of breakdown in the aerobic process (Table 6-2).

Table 6-2: Main factors influencing the composting process (Amlinger *et al.* 2005; Bidlingmaier 2000; Haug 1980)

Factor	Description
Temperature	Heat is a by-product of decomposition and accelerates the process. Heat production is mainly due to microorganisms degrading organic matter. The optimum temperature for fast decomposition is between 30-55°C, below that decomposition will slow down, but not cease. Temperature is the main factor for process control.
Oxygen	Oxygen is required for respiration by all aerobic organisms within the pile. Adequate ventilation increases the rate of decomposition; it removes excess moisture, CO_2 and heat. Therefore a sufficient air pore volume has to be assured, 0.8-2 g O_2 kg OM^{-1} h^{-1}.
Water Content	An efficient composter needs to have a moisture content of approximately 50% (45-65%). If it is too dry (<25%), activity of microorganisms and decomposition will slow down considerably, while overly wet piles can trigger anaerobic conditions and begin to smell.
Surface Area	The more surface area is exposed for microorganisms to attack, the faster the decomposition. This can be maximised by shredding and chipping waste into small pieces.
Carbon-Nitrogen Ratio	For fast composting, C/N ratio should be 20-40. This ratio can be achieved by mixing materials with different C/N ratios. Proper mixing can be assured e.g. by alternating layers.

Usually a combination of different substrates is best to achieve a good process and a good final product. For this purpose, biodegradable[1], compostable[2] or compost-compatible[3] as a

[1] Biodegradable material can be broken down completely into carbon dioxide, water and biomass. It may take a very long time (e.g. hardwood in an arid area), but it ultimately breaks down.

complementary material may be used. By combining, structure and in consequence aeration can be improved. C/N ratio can be adapted to an optimal range. It might also be favourable to increase the pH or reduce the salt content of the final product. A list of favourable combinations is given by Anh (2002). For slurry he recommends straw or straw-manures as best additives. Besides getting rid of the waste, an important aim of the composting process is to create a useful, applicable product. The main factors related to compost quality are listed in Table 6-3.

Table 6-3: Main factors for the quality of final compost (Amlinger *et al.* 2005; Haug 1980)

Factor	Description
PH and alkalinity	PH is most important for the availability of plant nutrients. In general a neutral pH is favourable. Composts with high alkalinity will contribute to buffer the system against pH changes; although little is known about the buffer capacity of composts. Compost with pH lower than pH 5 indicates that it is not mature yet and may contain phyto-toxic compounds.
C/N ratio	Best results are obtained at C/N ratios of 1:20 to 1:25. Composts with a high C/N ratio are limited by N supply and may immobilise N, whereas composts with at a low C/N ratio are limited by carbon and may lead to ammonia toxicity.
Cation Exchange Capacity (CEC)	The higher the CEC, the more exchangeable cations can be hold by the compost. CEC tends to increase as maturity and humic substances increase. It depends on the pH.
Salinity	The tolerable salinity depends on the kind of application. Salinity is usually higher in composts of manure wastes than from yard waste.
Heavy metals	High concentrations of heavy metals are toxic for plants.
Stability	Low microbial activity indicates the termination of the composting process, observed by low oxygen consumption and/or low CO2 production.
Pathogens	To avoid transmission of diseases has to be avoided, the compost should be free of viable pathogens.
Weed seeds	Transmission of weed seeds is not wanted. For process control tomato seeds are added.

6.1.2 History and Common Techniques

Composting has a long tradition in Asia, e.g. in China, Korea and Japan where animal and human excrements (“night soil”) have been reused in the fields after treatment (King 1911, reprint 1984; McGarry & Stainforth 1978; Shiming 2002).
The use of compost was reported along the Himalayas already 4000 years ago. The “Healty Hunzas” reused all vegetable leftovers, their own excrement, dung and urine from their barns for composting (Jenkins 2005).

[2] "Compostable" is a subset of "biodegradable": compostable material biodegrades substantially under composting conditions. The size of the material is a factor in determining compostability because it affects the rate of degradation.
[3] A compost-compatible material does not have a deleterious effect on the compost (e.g. it is not a biocide). Compost-compatible materials are generally inert and are present in compost at relatively low levels, e.g. sand particles and inert particles of plastic.

However, when regarding the habits of some bird species, composting is probably much older and not man-made. Megapodes of various regions of Australasia (belonging to the family Megapodiidae, order Galliformes) are famous for building nests in the form of huge compost heaps containing leaf litter, in which they incubate their eggs. The fowl-sized birds, also called "mound-builders", "incubator birds", work constantly to maintain the correct, almost exact, incubation temperatures, by adding and removing leaves from the compost pile (Lovette 2008).

In Europe first reports on composting practice are dated in the roman era, given by Columella and Plinius (Locher & Kornwachs 2006). Further reports of Cato (234-149 BC), Varro (16-47 BC) and Vergilius (70-19 BC) contain descriptions of agricultural practice (Buch 1986). During the middle age, human and animal faeces was used in the fields and as medicine (Pieper 1987).
From the middle of the 19th century more attention was paid to composting due to increasing population and waste. At that time, additionally to animal manure, toilet waste was added to the fields. It was even brought from the cities to the rural area in bins ("Tonnensystem") before water-based sanitation was introduced (Berger & Lorenz-Ladener 2008; Dunbar 1912, reprint 1998; Seyfried 1999). Until the end of the 19th century the "Humustheorie" was common belief before being displaced by the "Mineralstofftheorie" by Sprengel, Thaer and Liebig (Diepenbrock *et al.* 2005).
Composting of farm yard wastes kept being promoted by a number of scientists in Europe (Reuter 1954; Steiner 1924) or the USA (Lyon & Buckman 1922) regardless the increasing popularity of mineral fertiliser in the 20th century.
This progress also decreased reuse of compost in China, where in the end of the 1950´s humanure was still contributing one third of the fertiliser used in the country (Jenkins 2005).

Composting is nowadays usually connected to bio-waste treatment which is a common part of the waste management in Europe, as well in several states of the US and Canada. Main materials composted are organic household waste, industrial organic waste, and sewage sludge.

Newly, composting toilets are gaining popularity, not only in developing countries but also in the western world (e.g. forthcoming 3rd Dry Toilet Conference in Tampere in August 2009). Due to the existing sanitation systems, the installation in the western countries is usually restricted to remote areas e.g. in Northern Europe, or to allotment areas (Naudascher 2001). However, especially in areas where no (water based) sanitation systems are installed yet, composting toilets may contribute to improved sanitation (Berger & Lorenz-Ladener 2008; EcoSanRes 2008a; Werner 2003; Werner *et al.* 2007) and the produced compost to soil fertility.

Two main types of composting have to be distinguished nowadays: home composting and industrial composting. Essentially the same biological processes are involved in both scales, however, besides the modified technologies, the aims are slightly different – in industrial composting, the aim is to get rid of the waste and safe costs for waste disposal; home composting is often done to get adequate supply (of fertiliser) even in remote areas or to get an additional income by selling (Mansur 2004).
Most commercial and industrial composting operations are run by "active" methods that include additional air supply e.g. by turning. The environmental conditions or composition of the material etc. are optimised. Heat development is controlled and in consequence pasteurisation in all parts of the compost is ensured. Backyard composting on the other hand, is less intensive, mostly passive, and due to varying type of material supplied, may not always provide best conditions for heat development.

A description of the main traditional, on-farm composting methods is given by Misra & Roy (2003):

- Indian Bangalore Method (pit and heap composting)
- Chinese Rural composting (pit and hot temperature method)
- Windrow composting, by piling organic waste in long rows (windrows).

Due to Chinese tradition, the best combination of raw materials was found to be "four-combined-into-one": (1) human excrement and urine, (2) domestic animal and poultry faeces (mainly mule, horse, donkey and a small amount of ox faeces), (3) rubbish (brushwood and grass ashes, weeds and leaves), (4) soil (McGarry & Stainforth 1978).

The main industrial methods are (Bidlingmaier 2000; Haug 1980):

- Windrow composting, generally turned
- Tunnel or aerated static pile (ASP) composting which can be outdoor operations, under roof, or totally enclosed; with passive and active methods. Static piles are not turned, but placed on perforated piping, improving air circulation. It works well for wet materials and a wide variety of feedstocks.
- In-vessel composting, in vertical or horizontal closed metal or concrete reactors, e.g. agitated bed, tumbling bed, rotating drum.

In Table 6-4 the characteristics of basic composting methods are compared.

Table 6-4: Basic composting methods (Agriculture and Rural Department Alberta 2007; Bidlingmaier 2000; Haug 1980)

	Bin	Passive Windrow	Active Windrow	Aerated Static Pile	In-Vessel Channel
General	High-technology, medium quality	High-technology, quality problems	Most common on farms	Effective for farm and municipal use	Large for commercial application
Labour	Medium labour required	Low labour required	Increases with aeration frequency and poor planning	System design and planning important monitoring needed	Consistent level of flow to be cost efficient
Site	Limited land but composting structure	Requires large land areas	Can require large land areas	Less land required, given faster rates and effective pile volumes	Very limited land due to rapid and con-tinuous operations
Active Period	Range: 2 - 6 months	Range: 6 - 24 months	Range: 21 - 40 days	Range: 21 - 40 days	Range: 21 - 35 days
Height/Width	Dependent on bin design/ variable width	1 - 4 m / 3 - 7 m	1 - 2.8 m / 3 - 6 m	3 - 4.5 m / variable	Dependent on bay design/variable
Aeration System	Natural convection and mechanical turning	Natural convection only	Mechanical turning and natural convection	Forced airflow through the pile	Extensive mechanical turning and aeration
Process Control	Initial mix or layering and one turning	Initial mix only	Initial mix and turning	Initial mix, aeration, temperature and/or time control	Initial mix, aeration, temperature and/or time control, turning
Odour	Odour can occur, but generally during turning	Odour will occur, the larger the windrow the greater the odours	Turning can create odours during initial weeks from surface area of windrow	Odour can occur, control possible, e.g. pile insulation and filters on air systems	Odour can occur, often due to equipment failure or system design limitations

A special issue is the composting of wet materials, like biogas sludge. Due to the high moisture content (> 85-90%) and low structure stability, special arrangements for sufficient oxygen supply have to be established.

For composting of wet material three main approaches have been used (Haug 1980):

- Addition of ready compost
- Addition of organic amendments , e.g. sawdust, straw, rice hulls or bulk agents e.g. wood chips
- Drying before composting.

Additionally, "liquid composting" is possible on industrial scale by continuos air supply (Brocks 1990). Liquid composting has been managed e.g. in treatment plants for blackwater in Sweden (Kvicksund, Norrtälje) and Finnland (Åland) as an alternative to anaerobic treatment (Vinneras 2004), demanding lower investment but higher energy supply. There are also liquid composting plants for slurry, e.g. in the German Eifel area (Doll 1999; Ernst & Dügebacke 2000).

An important issue when reusing waste material are health related aspects. The danger of disease transmission has to be evaluated clearly (Blumenthal & Peasey 2002; Feachem *et al.* 1983; Jenkins 2005; Pieper 1987) and adequate means to minimise these have to be taken.

Regulations to assure the compost quality have been established by the European Union and the countries in Europe. Nowadays, a remarkable number of laws and guidelines are regulating the production and application of compost, e.g. in Germany. Different quality classes are distinguished, depending on the recommended use (Bidlingmaier 2000; Bundesgütegemeinschaft Kompost 1998; Gottschall *et al.* 1995).

Other parts of the world lag behind (Brinton 2000), despite some regulations in the US[4] or Canada[5] (Agriculture and Rural Department Alberta 2007). There are applicable recommendations given by the WHO (WHO 1989, 1996; 2006b) and FAO (Pescod 1992).

6.1.3 Development and Practice in Vietnam

According to the National Soil and Fertiliser Institute in Hanoi, "Vietnamese farmer use farm-yard manure to fertilize their field, this tradition is widely applicable, especially in the north of the country. In average, the rate applicable is almost 6 to 25 tons ha^{-1} a^{-1} depending on the crop and the region. This amount reaches a total equivalent of 210,000 t N, 105,000 t P_2O_5 and 230,000 t of K_2O. […]. Biological fertilisers, foliar fertilisers are also produced in Vietnam but it is not easy to control the quality as well as quantity of this product" (Son & Ha 2006).

Due to calculations of Anh (2002), around 60 million tons of organic fertilisers were produced in Vietnam, the major part being animal and straw manure.
Besides animal manure, human excrements are reused (Anh 2002; Humphries *et al.* 1997; Pieper 1987). Composting toilets or double-vault latrines are common in various areas of Vietnam (Soussan *et al.* 2005) although there are no reports of composting privys (toilets) being used on a wide scale until the 1950s, when the Democratic Republic of Vietnam initiated a five-year plan of rural hygiene and a large number of anaerobic composting toilets were built (Jenkins 2005). These toilets, known as the Vietnamese Double Vault, consisted of two above ground water-tight tanks, or vaults, for the collection of humanure (Jenkins 2005). However, acceptance and use seems to be strongly dependent on ethnic customs,

[4] "Standards for the Use or Disposal of Sewage Sludge", Regulation no. 503, 1992
[5] Quality guidelines on compost, 1994, http://www.compost.org/compostqualitydoc.pdf

and the influences of other cultures - Chinese in the North, Indian (Cham, Khmer) in the South.

Reuse of municipal bio-waste was not found to be significant in Vietnam, despite some activities in the major cities Hanoi and HCMC. Similar to other South-east Asian countries, like Thailand and the Philippines, the existing limited supply of compost products is beset with problems of uncertain quality, high prices and an inefficient retailing organization, according to Van Buuren & Dieu (2007).

In Hanoi, the main composting plant is Cau Dien in Tu Liem District. Hanoi generates on average 3,000 tons of waste a day. Only 20% of it is recycled, the vast majority of refuse is dumped in landfill sites, according to Le Van Duc, deputy head of the municipal Department of Transport and Public Works (Hoa 2007). The construction of another processing plant with a capacity of 250.000 tons waste per year (66,300 t a^{-1} compost) in Nam Son has been planned (Thuy 2004).

In Ho Chi Minh City as well, the capacities for composting municipal waste are not sufficient. The city has capacities for around 12% of the collected material, 88% are disposed at a landfill (Anh 2002). The generation of municipal solid waste has reached 6000 metric tons per day, of which 5000 tonnes are delivered to landfill. Waste generation is increasing at 6% annually. About 150,000 tons per year of compost are produced from mixed solid waste (VanBuuren & Dieu 2007). The authorities in HCMC are trying to implement their first biowaste-reuse chain, based on source separation, as the greater part (79%) is organic waste (Hädrich & Bidlingmaier 2007). Due to high contents of heavy metals and organic pollutants, composting of mixed municipal waste was abandoned in favour of separated organic waste in Europe (Bidlingmaier 2000).

Can Tho does not manage a composting site - like many other smaller cities. Although the main part of the collected municipal waste is of organic origin, the material is disposed on the landfill, together with other waste.

Small scale composting at farm level takes place in the North and in mountainous regions of Central Vietnam. In the Mekong Delta composting was rarely seen. Interviews of Wieneke (2005) confirmed this, as 79% of the interviewed households did not use compost, and only half of the households were providing organic waste (excrements) for aquaculture.

Nevertheless, even in the south of Vietnam there seems to be a (small) demand for composts as compost can be purchased at the local market (own observations):

- Commercial, dry "bio-fertiliser" packed in small units of 500 g, ingredients not clearly mentioned
- Dried cow dung, sold open, packed as required
- Biogas sludge "compost" (overflow sludge dried in an earth pit), sold directly at the local farm, for 1,000 VND kg^{-1}
- Sugarcane mud.

Anh (2002) supposed that the shortage of proper organic compost leads to use of mineral fertiliser. In his opinion, a clear focus on the use of organic biomass in Vietnam is lacking.

According to the National Soil and Fertiliser Research Institute of Vietnam (Bo 2008), scientific approaches based on integrated plant nutrition management are necessary to the define the role of fertilisers in Vietnam. In his opinion it is the insufficient knowledge that is leading to the overuse of chemical fertiliser.

6.1.4 Summary and Research Questions

By composting, organic waste can be transformed to a more pleasant substrate that can be used as a soil conditioner and may be traded. It may also be a method to treat sludge or slurry that is generated e.g. by water treatment systems or biogas plants. In Table 6-5 the main benefits and obstacles of composting are listed.

Table 6-5: Benefits and disadvantages of manure compost

Benefits	Disadvantages
Reduces mass and volume -> improves transportability	Loss of nitrogen (ammonia-N)
Reduces odour	Time and labour and some experience involved
Destroys pathogens and kills weed seeds	Cost of equipment (initial and operating)
Increases water retention of soil	Land requirement for composting
Slow release of nutrients from compost	Marketing required for sale
Stabilises the volatile nitrogen into large particles, reduces losses and pollution	
Saleable product	

Summarising the situation in Vietnam, composting and the application of organic waste in cropping is practiced to some extent at household level but is not a common practice in the Mekong Delta. In general, organic waste treatment does not receive high appreciation in this area. Ready-made organic fertiliser however is sold on the market at rather high prices.

Although composting of wet material is more complex, there are experiences in different fields and different parts of the world that may be combined and applied to local conditions in the Mekong Delta.

To find out about the efficiency of the process (heating, nutrient changes, hygienisation) and the plant response to these substrates, composting experiments were set up with biogas sludge with and without amendment of rice straw. For comparison, pig excrements were investigated, too.

The main questions related to this topic were:

- Is composting a suitable post-treatment for biogas sludge? Is straw a suitable amendment?
- Which is the return of composting regarding nutrient transfer (N-loss) and pathogen reduction?
- Is mechanical turning improving the process?

6.2. Material and Methods

6.2.1 Set-up

Different substrates, listed in Table 6-6, were composted for two months, during the "sunny season" from February to March. The experiment was carried out at an open space area at the Department of Agriculture at Can Tho University (Figure 6-1).

Figure 6-1: Experimental set-up of composting bins

Biogas sludge without amendment (T1) was chosen as it represents the simplest treatment on biogas sludge and the only one seen at farms in the Mekong Delta. In treatment T2 and T3 biogas sludge was supplied with different amounts of rice straw –a widely available material - to improve structure and aeration (Table 6-6).
The biogas variants were compared to pig manure (T4) and pig manure with straw (T5).

Table 6-6: List of treatments and initial amount of material used

Name	Material composition	Amount (kg)
T1	Biogas sludge without amendment	90
T2	Biogas sludge with rice straw	90+2
T3	Biogas sludge with rice straw	90+4
T4	Pig excrements without amendment	45
T5	Pig excrements with rice straw	45+2

Biogas sludge and rice straw had been collected at neighbouring farms, pig manure from the pig sties at the Department of Animal Science at CTU. Rice straw was cut into pieces of around 10 cm, and put into straw baskets together with the sludge. No initial inoculation with bacterial strains or old compost was done. The addition of rice straw increased the total volume around one third although only a minor amount of mass had been added.

Each treatment consisted of three replications that were started the same way. After two weeks one replicate (called "active") was turned over, two replications were left undisturbed ("passive"). The baskets were placed above a plastic jar to collect percolation water.

6.2.2 Measurements

The experiment was carried out by Sebastian Hedel when conducting his diploma thesis in Can Tho in 2004. His data were processed further.

In the following table (Table 6-7) methods and instruments are listed that were used for determining the required parameters.

Table 6-7: Methods and Instruments used for analysis

Parameter	Method description	Instrument	Place
Organoleptic evaluation	Daily	Eye, nose, ear	On-site CTU
Temperature	3 different depths, 9 points, up to 3x a day	Thermometer, 55 cm	On-site CTU
Moisture	2x a week	Soil moisture sensor Profile Probe (PR1), Delta-T Devices	On-site CTU
Weight		Balance	On-site CTU
Maturity index	AT_4 respiration test (Bundesministerium 2001)	Dewar vessel, respirometer	Laboratory CTU
Plant response	Cress test (details see below)	Germination jar and cress seeds	Open space at laboratory CTU
Nitrogen	Kjeldahl	Gerhard Kjeldahl Therm	
Phosphorus	Photometric		INRES Bonn
Heavy metals	AAS		INRES Bonn
Total coliforms	MPN-methode, ISO 9308-2	MPN in Fluorocult® Laurylsulfat-Bouillon (Merck)	CoT, at CTU
E. coli	MPN methode, ISO 9308 Part 1 and 2	Fluorocult® Laurylsulfat-Bouillon & Chromocult® Coliform agar (Merck)	CoT, at CTU
Salmonella sp.	DIN EN ISO 19250, 2003-Draft	Rappaport-Vassiliadis broth, *Salmonella*-Shigella agar (Merck)	
Helmith eggs	Counting, Bailinger method	Microscope	CoT, at CTU

The respiration test was carried out according to AT_4 (Bundesministerium 2001). Each compost sample was analysed with one replication. The compost samples were adjusted to the required optimal water content with the "fist test". 20 g of each compost were placed into special bottles with pressure sensors and were stored in a incubator at defined temperatures of 32-33°C. At pressures lower than -110 hPa, the system was aerated for 10 min.

A cress test was carried out by putting compost into jars of 100 cm^2 size, two replication of each type. 1 g of cress seeds *(Lepidium sativum)* was distributed and watered daily. After 5 days, germination was evaluated, after 10 days growth and root system in comparison to standard compost from the botanical garden Bonn.

6.3. Results and Discussion

In the following chapter, results related to the composting process itself as well as the consequences for later use as a fertiliser are described.

6.3.1 Management and Process Parameters

6.3.1.1 Observations

The initial smell of pig manure and the slight smell of anaerobic sludge had disappeared after two months, and the material had reached an earth like structure.

In the pig manure compost (T4), fly larvae of the black soldier fly (*Hermetia illucens*) developed. This finding has been described by other authors (Phan *et al.* 2003).
These larvae are common scavengers in compost heaps. They are used in manure management, for house fly control and reduction in manure volume. The mature larvae and prepupae are a feed supplement (Sheppard *et al.* 1994).

6.3.1.2 Temperature

The temperature development is most important and is used for process control. High temperatures will lead to fast decomposition and assure pathogen die-off. The expected temperature course consists of several phases. In all variants, except for T1, thermal activity followed more or less the typical rhythm (as explained in Table 6-1).

Table 6-8 shows the maximum temperatures reached in the different active and passive variants.

Table 6-8: Maximum temperatures of different substrates by active and passive composting

	Treatment	Max. Temperature active pile [°C]	Max. Temperature passive pile [°C]
T1	Biogas sludge without any amendment	33	31
T2	Biogas sludge with rice straw	33	31
T3	Biogas sludge with rice straw	40	40
T4	Pig excrements without amendment	57	47
T5	Pig excrements with rice straw	68	70

Taking temperature increase as an indicator for the successful treatment, the process worked well for pig excrements (T4 and especially T5). Even in the small given volume with a rather low insulation effect high temperatures occurred. It was successful for biogas sludge with straw (T2, T3) despite lower maximum temperatures. However, the highest "thermal effectiveness" takes place between 40-50°C (Bidlingmaier 2000).

The composting process did not show any temperature increase in biogas sludge without amendment (T1). Due to the fact that fast available carbon was already transformed within digestion process only larger carbon molecules are left, the activity of microorganisms is hampered and thus heating is low. Highest temperatures were reached in T5 (Figure 6-2), pig excrements with straw: 69°C were kept for several days in the centre of the pile.

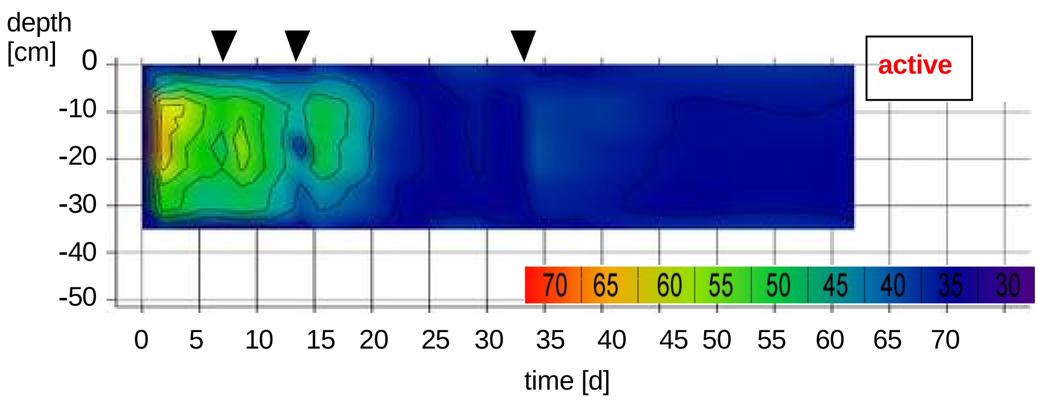

Figure 6-2: Temperature development during composting of T5 (pig manure + rice straw); arrow marks the date of turning

In the biogas-straw mixture (T3), temperatures increased to a maximum of 40°C within the first days. This temperature was kept for several days. Active treatment seemed to keep the high temperature ~2 days longer in its central part but for the whole pile the difference was hardly noticeable (Figure 6-3).

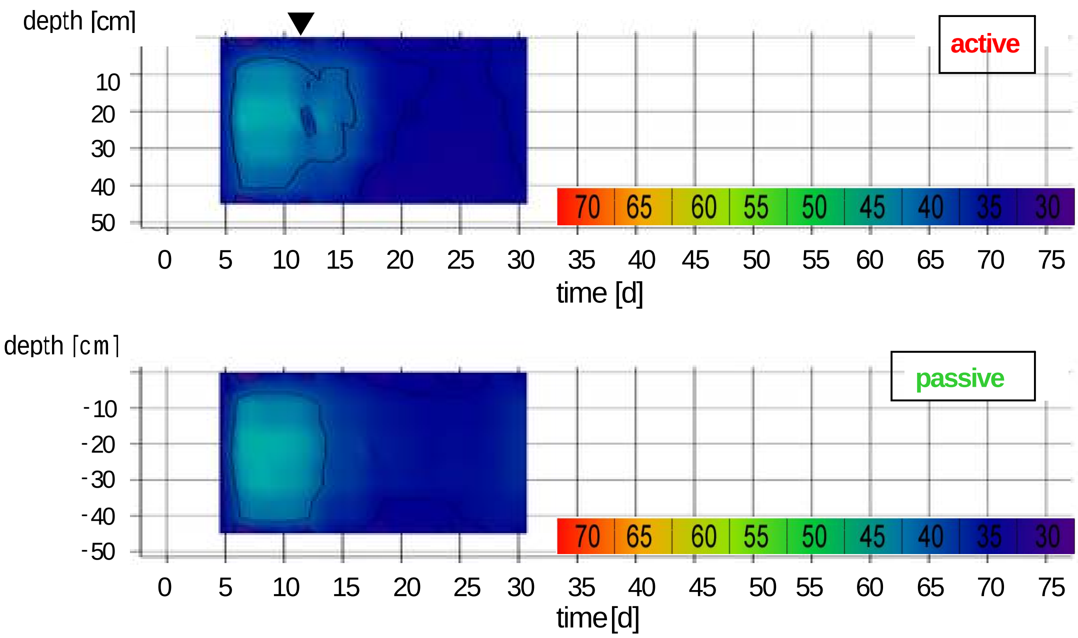

Figure 6-3: Temperature development during composting of T3-biogas sludge + rice straw, changes by time and depth; arrow marks the date of turning

In treatment T1 – biogas sludge without amendment (figure not shown) – temperature increase was hardly recognisable. A small temperature increase to 33°C was found in the centre of the pile. After turning, the temperatures increased to 34°C in the centre core of the active treatment. In the passive treatment, temperatures did not exceed 32°C.
This temperature increase was very little, most probably due to the high moisture content (Table 6-9) which did not allow enough aeration. C/N ratio was high, too (Table 6-14).

The differences between active and passive treatment were marginal for T1 to T3. For T4 the active treatment lead to a higher temperature increase whereas in T5 passive treatment was more effective. Active turning takes out moisture and may have increased aeration in T4 where structure was poorer than in T5. In T5, where the process worked well already before turning, this action did not lead to any further improvement but may have hampered the activity of fungi and actinomycetes by destroying their mycelia (Anh 2002).

6.3.1.3 Moisture

Decomposition of organic matter depends on the presence of moisture to support microbial activity, to allow nutrients to reach the organisms. On the other hand, it must not be too wet as sufficient oxygen needs to be supplied to the microorganisms.

During the experiment, moisture decreased. Final moisture contents of active treatment were higher or equal to passive treatment in the end (Table 6-9).

Table 6-9: Moisture content of different substrates in the beginning and end of treatment process

	Treatment	Initial [%]	Final active [%]	Final passive [%]
T1	Biogas sludge without amendment	75.5	53.5	50.5
T2	Biogas sludge with rice straw	73.5	50.0	45.0
T3	Biogas sludge with rice straw	70.0	56.0	56.5
T4	Pig excrements without amendment	71.5	48.0	45.0
T5	Pig excrements with rice straw	66.5	48.5	39.0

Generally, moisture contents of 40-60% are suitable (Anh 2002). However, the intended moisture values depend on the structure of the material; e.g. fibrous and bulky material can take up high amounts of water and still keep their structure and porosity. Therefore moisture content may be 75-85% for straw and rice hulls, and 55-65% for manures. For digested or raw sludge 55-60% are recommended (Haug 1980). The air pore volume should be 35% to 55% (Bidlingmaier 2000).

More detailed measurements showed that piles with straw manure did not have very moist areas in the bottom of the piles but usually ranged in the medium moisture content. The passive pile had a very moist area for a short time, the active pile showed more or less the same moisture content in all areas.

6.3.1.4 Mass Balance

The heights in the baskets decreased from the beginning to the end, resulting in a final volume of approximately two thirds of the initial.
Within two months, the total mass decreased to 20-30% of its initial amount (Figure 6-4). At the beginning, dry matter contributed around 20% to the total mass (T1: 24%, T2: 21%), after two months, it was around 50% (Figure 6-4).

The fast mass decrease that can be seen in the first days is due to water loss.
Dry matter decreased within the composting process to 50-70% of the initial dry mass.

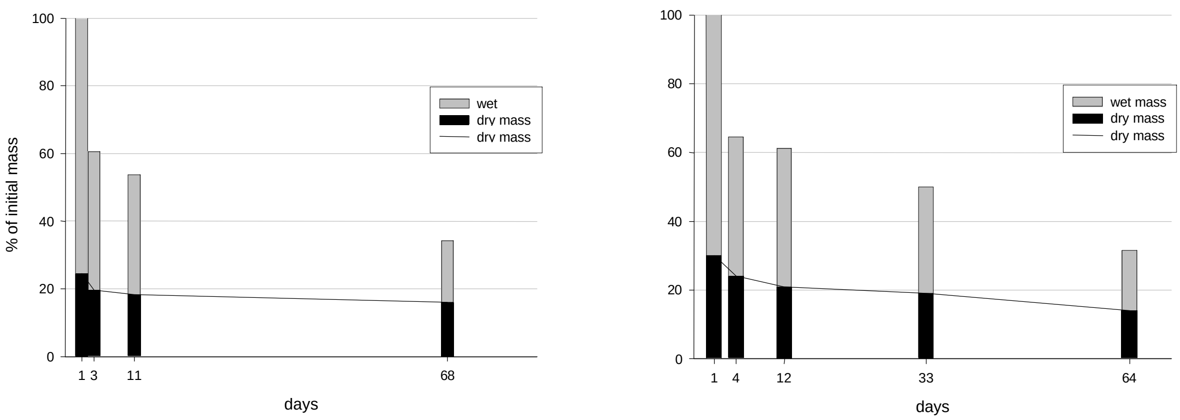

Figure 6-4: Mass development of biogas sludge (T1, left) and biogas sludge with rice straw (T3, right graph) during composting

In the treatments with rice straw, the active treatment showed a higher mass loss, for biogas sludge (T1) passive was higher Table 6-10). These differences are most probably due to the increased air supply which improved living conditions for microorganisms in the active piles (Anh 2002; Bidlingmaier 2000).

Table 6-10: Dry matter of different substrates in the beginning and end of treatment process

	Treatment	Initial* (kg DM)	Final active (kg DM)	Final passive (kg DM)	Final active (% of init.)	Final passive (% of init.)
T1	Biogas sludge without amendment	22.0	15.6	14.2	70.9	64.5
T3	Biogas sludge with rice straw	22.2	13.0	13.2	58.6	59.5
T5	Pig excrements with rice straw	15.8	9.7	12.0	61.4	75.9

*Initial masses see Table 6-6

6.3.2 Fertiliser Quality

6.3.2.1 Carbon and Nitrogen

Carbon contents at the beginning and the end of the composting process are listed in Table 6-11. The data show that carbon was reduced to about half of the initial content in the variants with straw (T3, T5). Less carbon was gone in the biogas sludge treatment (T1).

Table 6-11: Carbon content in different substrates in the beginning and the end of composting

		Initial* (kg C)	Initial (% C)	Final active (kg C)	Final passive (kg C)	Final active (% C)	Final passive (% C)
T1	Biogas sludge	9.0	41	5.6	5.1	36	36
T3	Biogas sludge with rice straw	8.9	40	4.4	4.7	34	35
T5	Pig excrements with rice straw	5.8	37	2.8	3.9	29	33

* Initial total mass see Table 6-6

Carbon is consumed during composting by microorganisms to build up biomass and for respiration. Minor consumption of carbon in T1 therefore indicates lower microbial activity. This finding corresponds to the results of heat development (see chapter 6.3.1) where T1 showed lowest temperature increase.

Nitrogen contents of the composting materials are listed in Table 6-12. The initial N-concentrations of the applied biogas sludge were rather low but still in the range mentioned by other authors (Becker *et al.* 2005).
During composting the N mass did not change significantly.

Table 6-12: Nitrogen content in different substrates in the beginning and the end of composting

	Material	Initial* (kg N)	Initial (% N)	Final active / passive (kg N)	Final active/passive (% N)
T1	Biogas sludge without	0.2	0.8	0.2 / 0.2	1.3 / 1.2
T3	Biogas sludge with rice straw	0.2	0.9	0.2 / 0.2	1.5 / 1.2
T5	Pig excrements with rice straw	0.3	2.1	0.3 / 0.3	2.8 / 2.6

* Initial masses see Table 6-6#

N concentration however, increased from 0.8% in the biogas sludge (treatment T1) to 1.3% in the active and 1.2% in the passive treatment. In the biogas-sludge-straw mixture (T3) the share increased similarly from 0.9% to 1.5% (active) and 1.2% (passive pile). In the pig manure-straw compost an increase from 2.1% to 2.8% (active) and 2.6% (passive) could be recognised.

Usually higher nitrogen losses are to be expected during composting. Table 6-13 shows mean values of nitrogen "surviving" various processes.

Table 6-13: Residual nitrogen after composting and other processes (Marchaim 1992)

Nitrogen Effectiveness	Index (%)
Manure piled for 30 days	50
Manure piled for 14 days	55
Manure piled 2 days before spreading and ploughing in	80
Dried digester plant effluent spread and ploughed in	85
Effluent from digester, introduced immediately into irrigation water	100
Manure spread and ploughed in immediately	100

A suitable C/N ratio is important for the process performance and also has a strong impact on the plant suitability. Table 6-14 presents the C/N ratios of the three treatments. The biogas sludge (T1) and biogas-straw mixture (T3) showed high initial C/N ratios of 50:1 and 45:1 whereas the one of pig manure-straw (T5) is remarkably lower (17:1).

Table 6-14: C/N ratio in different substrates in the beginning and the end of composting

		Initial C/N	Final active	Final passive
T1	Biogas sludge	50:1	25:1	28:1
T3	Biogas sludge with rice straw	45:1	28:1	30:1
T5	Pig excrements with rice straw	17:1	10:1	13:1

Thinking of the production of biogas sludge - CH_4 and CO_2 formation during anaerobic digestion thus removal of carbon as gas - one would expect (at first) that the C/N ratio increases in the substrate as nitrogen is conserved in the digestion process. The data, however, show the opposite effect: the C/N ratio of the substrate before digestion (pig manure) is a lot lower than the one of the substrate after digestion (biogas sludge).

This high C/N ratio may be due to the geometry of the biogas fermenters: in the fermenter, most solids get settled in the bottom whereas soluble components, e.g. NH_4-N are taken out with the liquid effluent. As specified in chapter 4.3.4.3, the nitrogen mass in the effluent of the biogas plants (mainly NH_4^+-N) usually met the inflow values. Thus, a large amount of N is washed out and only a minor part of the N (the hardly degradable N_{org} fraction) remains in the sediment, the biogas sludge.

Compared to recommended values, the measured C/N ratios are either too high or too low for an optimal composting process. Ideally C/N ratio should be in the range of 25 to 35 (Amlinger *et al.* 2005; Haug 1980). At C/N ratios wider than 40:1 composting is inhibited or retarded; below the range, composting is likely accompanied by high nitrogen losses. This may already start from below 30:1 (Haug 1980), most other sources mention lower values, e.g. below 10:1 (UBA 1999).
Composting was retarded in T1, and to a minor extent in T3, too (see temperature development, chapter 6.3.1). Nevertheless, C/N ratio of all substrates was reduced considerably.

To accelerate and improve the process, other substrates may be added to reach the recommended C/N ratio of ~30:1 (Table 6-2). Suitable co-substrates that may be added for reducing the C/N ratio are liquid slurry, urine, human excrements or poultry manure; for increasing the ratio, leaves, wood chips or paper waste are suitable additives (Amlinger *et al.* 2005; Anh 2002; Jenkins 2005).

6.3.2.2 Hygienic Parameters

To evaluate the hygienic concerns, the common indicator organisms were examined: total coliforms, *E. coli*, *Salmonella* sp. and worm (helminth) eggs.

In treatment T1, coliform bacteria and *E. coli* were reduced 5 log-units, 99,999 % within two months: within the first 17 days their number was reduced from 3.1 x 10^6 CFU g^{-1} to 1.3 x 10^4 CFU g^{-1} and within the next 48 days to 16 CFU g^{-1} (Figure 6-5, left).

The reduction in the treatments T2 and T3 (biogas sludge with rice straw) was faster but not higher in total. In T3, a reduction from 3.1 x 10^6 CFU g^{-1} to 5.4 x 10^3 CFU g^{-1} was achieved already within the first 17 days for coliform bacteria and *E. coli*. During the following 48 days, the concentration was reduced one more log_{10}-unit to 180 CFU g^{-1} (Figure 6-5, right).

In T2, total coliforms and *E. coli* were reduced to $3x10^2$ and 60 CFU g^{-1} respectively, within 82 days (data not shown).

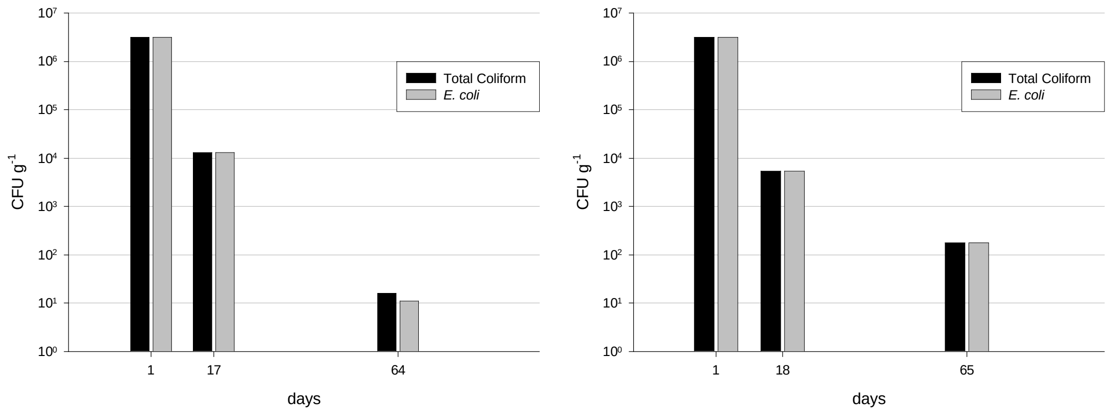

Figure 6-5: Concentration of total coliforms and *E. coli* in biogas sludge: T1 (left) and T3 (right) during composting, mean of three replicates

When composting pig manure (T4, T5), reduction rates of *E.coli* and total coliform bacteria were smaller (Figure 6-6).

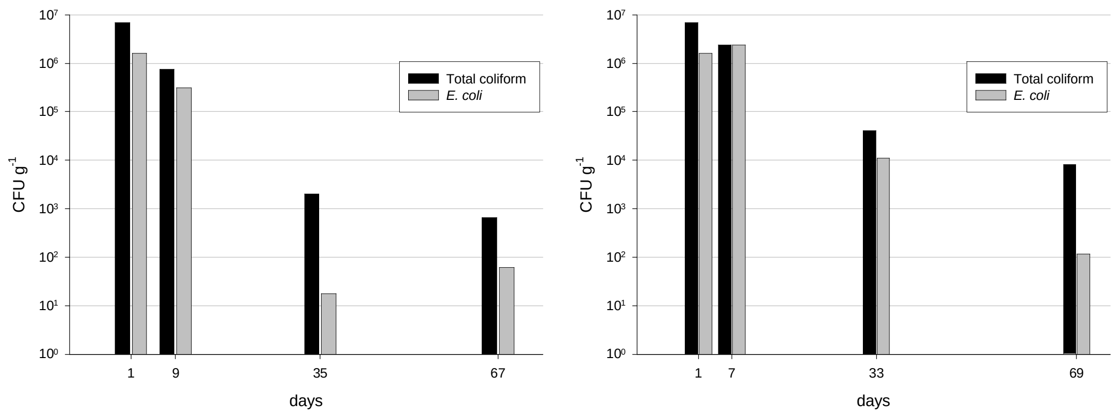

Figure 6-6: Concentration of total coliforms and *E. coli* in pig manure during composting: mean of three replicates of T4 (left) and T5 (right)

Starting with the same amounts of ~10^6 CFU g^{-1} reduction rates of 3-4 $\log_{10}$ units was reached. Similarly to the results of T1 and T2, the reduction in the treatment without straw was better.

In other experiments in Vietnam with pig manure-straw, temperatures of 50°C for three weeks lead to a similar reduction of *E. coli, Salmonella* and worm eggs (Phan *et al.* 2003).

In the conducted experiments no correlation between temperature and die-off could be found although Haug (1980) postulates the inactivation of microbes as a function of temperature and length of exposure.

Vietnamese health authorities are regarding forty-five days in a sealed vault (of a double vault latrine) as adequate for the complete destruction of all bacteria and intestinal parasites in human excrements. Eighty-five percent of intestinal worm eggs, one of the most persistently viable forms of human pathogens, were found to be destroyed after a two months composting period in this system (Jenkins 2005).

In studies on composting toilets in Germany, usual storage time was found to be two years and more for more than half (56%) of the toilets, and 1 to 1½ years for 28% of the toilets (Fittschen 1999; Naudascher 2001).

US-EPA demands 55°C for 3 consecutive days at in-vessel and aerated static pile composting. For turned windrows 15 days with > 55°C are necessary (Turner 2002). Survival times of *E. coli* and *Salmonella* sp. in cow manure and cow slurry ranged – depending on temperature - from 6 days to 5 weeks in studies in the US (Turner 2002).

However, not only temperature but also other parameters, e.g. ammonia may be of influence (Nordin & Vinnerås 2007).

Haug (1980) proposed the following standards to be achievable with good management practices:

Total coliform:	median < 10 MPN g^{-1} solids not more than 20% exceeding 1000 MPN g^{-1}, no sample exceeding 10^4 MPN g^{-1}
Salmonella sp.:	median < 1 MPN g^{-1} solids not more than 10% exceeding 10 MPN g^{-1}, no sample >100 MPN g^{-1}
Ascaris ova:	no viable *Ascaris* ova detected, representative for all parasites, e.g. helminth ova or protozoan cysts

Recommendations have been given by different authors (Blumenthal *et al.* 2000a; Blumenthal *et al.* 2000b; Ongley 1996).
The WHO has been working on health based targets for reuse of wastewater in agriculture for their update of guidelines (WHO 1989, 2006a).

6.3.2.3 Heavy Metals

Heavy metals may cause toxic effects on plants, if present in high concentrations in the soil or nutrient solution. As they can not be degraded biochemically they are enriched during composting - due to decreasing total mass.

Biogas sludge composts (T1, T3) showed similar heavy metal concentrations. These were lower compared to the pig manure (T5): around two times for Cd, Cr, Cu, three times for Pb and four times for Ni. Only Zn concentration of pig manure (T5) was lower than the one in biogas composts (Table 6-15).

To compare with, Table 6-15 also lists the upper limits of the US Biosolids and the German Biowaste Ordinance (BioAbfV) as well as the range of ordinances in European countries. The German legislation only allows the application of compost-substrates if all heavy metals are below the given threshold, and if the soil concentration at the application site is below a certain level, too (Bundesminsterium für Umwelt Naturschutz und Reaktorsicherheit 1998) to exclude health risks for humans and animals. The regulation (BBodSchV) differentiates between soil texture, i.e. clay which can absorb more ions may receive a higher amount of the problematic element or substance.

Table 6-15: Heavy metal concentration in composts of biogas sludge (T1), biogas sludge-straw (T3), pig manure–straw (T5) compared to legal regulations of different countries (Brinton 2000), values exceeding the limits are underlined

		T1 $mg\ kg^{-1}$	T3 $mg\ kg^{-1}$	T5 $mg\ kg^{-1}$	German BioAbfV $mg\ kg^{-1}$	EU range $mg\ kg^{-1}$	USA biosolids $mg\ kg^{-1}$	German BBSchV $Kg\ ha^{-1}\ a^{-1}$
Cd	Cadmium	0.13	0.17	0.31	1.5	0.7 - 10	39	0.15
Cr	Chromium	9.4	10.3	19.9	100	70 - 200	1,200	2
Cu	Copper	<u>215</u>	<u>207</u>	<u>433</u>	<u>100</u>	70 - 600	1,500	12
Ni	Nickel	4.3	6.0	21.9	50	20 - 200	420	3
Pb	Lead	5.5	6.9	16.5	150	70 - 1,000	300	15
Zn	Zinc	<u>792</u>	<u>660</u>	<u>472</u>	<u>400</u>	210 - 4,000	2,800	30

The table shows that two elements, copper and zinc exceed the tolerable limits in Germany and in most other European countries, but not the ones in the USA (Brinton 2000).

High amounts of Cu and Zn are regularly found in slurry, composts and biogas sludge in Europe, too (Kühnen 2004; Kühnen & Goldbach 2004; Zethner *et al.* 2007). The main sources for the high concentrations were identified:

- Fodder (concentrated feed) with high concentrations of Cu and Zn, is often fed to pigs, especially to piglets as it is reputed to increase their resistance against diarrhoea and enhance performance (Zethner et al. 2007). The physiological demand of pigs is 3 – 10 mg Cu kg^{-1} and 50 - 100 mg Zn kg^{-1} fodder per kg live weight. (Landwirtschaft 1999; Zethner *et al.* 2007). In Europe, concentrations in fodder are now limited to 25 mg Cu kg^{-1} (170 mg Cu kg^{-1} for piglet feed) and 150 mg Zn kg^{-1} (EG 1334/2003).
- Metal parts in stables that may release heavy metals, e.g. when pigs are licking at these.
- Cu may also originate from pesticides and ash of wood fire, but this is usually of less importance.

These results on heavy metals in pig manure and pig manure compost in the Mekong Delta require source-tracking first. Then necessary means can be suggested and put forward.

A regular analysis of the produced compost and of each field receiving the compost - as required in the European legislation - seems not feasible for Vietnam, at least not for now. It seems more practicable instead, to minimise the input, e.g. if concentrated feed is identified as the source, limit the additives, and control production and selling. Additionally, information for the farmers has to be provided, e.g. by the Vietnamese Agricultural Extension Service.

6.3.2.4 Maturity and Plant Acceptance

The respiration rate analysed within four days (AT_4) is a German standard method to determine the maturity of composts. It reflects the microbial activity. Table 6-16 shows the results of the AT_4-test of three selected composts. According to the total oxygen consumption measured in the AT_4-test, all composts had reached a mature state after two months treatment.

Table 6-16: Results of AT_4 test and "Rottegrad" of composted substrates

	Treatment	AT_4 active [mg O_2 g^{-1} DM]	AT_4 passive [mg O_2 g^{-1} DM]	"Rottegrad"
T1	Biogas sludge	11	7	V
T3	Biogas sludge with rice straw	19	22	V
T5	Pig excrements with rice straw	27	44	V

Self-heating capacity (Rottegrad-index used in Germany) is another means to determine the maturity of the compost. Depending on the heat development in the laboratory experiment, the compost is classified[6]. According to these results, all composts had reached "Rottegrad V", ready for use.
These results are reasonable as composting time is usually expected from one month (for intense composting) up to six months for passive treatment at lower temperatures as shown in Table 6-4 (Bidlingmaier 2000; Rechenburg 2005).

In the cress test, the tested compost demonstrates if it is suitable for or an impediment to plant growth. The composts gained from biogas sludge showed better results compared to the pig manure composts in germination and root length/growth (Table 6-17).
Compost of the active treatment T1, biogas sludge showed best results in the cress test, the compost of the corresponding passive treatment was less successful but still good.
Compost of T3, biogas sludge with rice straw obtained good results, too. Here the passive treatment was better than the active. According to the cress test, the biogas sludge composts proved to be suitable for plants.
Only few seed germinated on the composts of T5, the pig manure-straw compost. This might be due to plant inhibiting substances, e.g. heavy metals or antibiotics given to the pigs which had been denied by the farmer. The more reasonable reason may be that the substrate still needed more time for development - T5 had the highest O_2-consumption among the tested composts, although all were classified "Rottegrad V".

Table 6-17: Results of different composts in the cress test

		Germination[7]		Root length[8]	
		active	passive	active	passive
T1	Biogas sludge without amendment	1	3	1	2
T3	Biogas sludge with rice straw	2	1	1	1
T5	Pig excrements with rice straw	5	5	5	5

[6] "Rottegrad" according to temperature rise in the laboratory experiment:
< 10°C Category V: very stable, well-aged compost
10-20°C Category IV: moderately stable, curing compost
20-30°C Category III: material still decomposing, active compost
30-40°C Category II: immature, young very active compost
> 40°C Category I: fresh, raw compost
[7] Germination according to a scale from 1 to 5: 1 is best, 5 is lowest response
[8] Root length according to a scale from 1 to 5: 1 is best, 5 is lowest response

6.3.2.5 Field Application

In field experiments different composts (pig manure-straw, biogas sludge compost, vermicompost) and sludges (biogas sludge, fishpond bottom sludge) were compared on an alluvial and on an acid sulphate soil (Becker *et al.* 2005; Chiem *et al.* 2005).

The substrates were similar to the one in this experiment, composts of biogas sludge after 2 and 6 months and pig manure-rice straw compost. Several other substrates were applied, each with 3 t ha^{-1} dry matter. Due to use of equal dry mass, different nutrient amounts were applied (Table 6-18).

Table 6-18: Nutrients applied with relevant substrate (Becker et al. 2005)

Aerobic substrates	Nitrogen kg ha^{-1}	Phosphor kg ha^{-1}	Potassium kg ha^{-1}
Biogas sludge compost (2 months)	60.0	8.1	8.4
Biogas sludge compost (6 months)	55.5	18.0	17.1
Pig manure–rice straw compost	78.0	49.5	59.4
Pig manure vermicompost	66.0	41.7	27.6
Goat manure vermicompost	88.5	9.0	41.1
Rice straw compost	82.5	4.5	17.1
Mushroom compost	73.5	6.6	43.5
Anaerobic substrates			
Biogas sludge (from plastic tank, < 4 months)	76.5	24.0	19.5
Biogas sludge (from cement tank, > 12 months)	72.0	20.4	8.4
Fishpond residue (3 – 5 cm)	48.0	12.0	54.3

The dry matter yields of mung bean *(Vignia radiata)* fertilised with different composts were measured separately for grain and stover (leaves and stokes) and compared to each other (Figure 6-7).

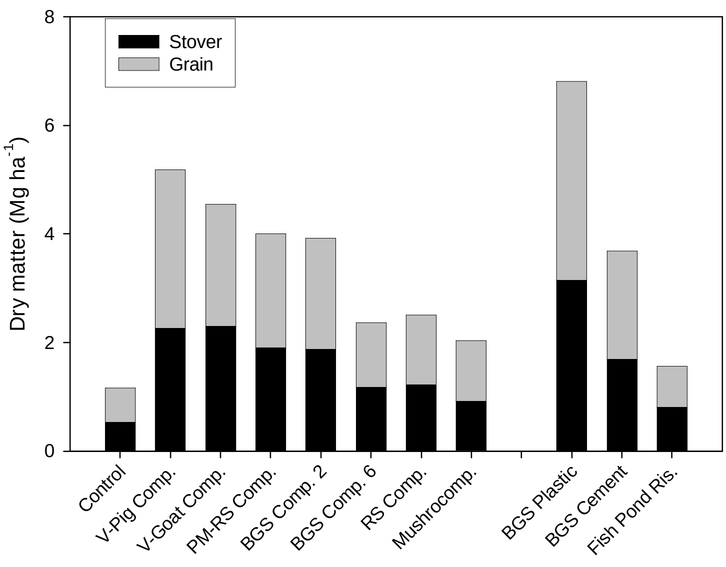

Figure 6-7: Mung bean response -grain yield and total dry matter accumulation- on an alluvial soil in An Binh after application of 3 t ha^{-1} substrate (Becker et al. 2005); V=vermicompost, PM=pig manure, BGS=biogas sludge, RS= rice straw

Yields of mung bean *(Vignia radiata)* were 4 t ha^{-1} for the pig manure-rice straw compost (PM-RS Comp) and the biogas sludge after 2 months of composting (BGS comp 2) compared to 1 t ha^{-1} for the control without fertiliser (Figure 6-7).
Biogas sludge compost after 6-months (BSS comp 6) got lower yields but was still better than the control.

Further results on use of organic substrates from waste materials have been conducted and are described elsewhere (Guong et al. 2009; Ni et al. 2009; Shibistova et al. 2009).
A number of other studies on recycling substrates in the Mekong Delta can be found (Mikled et al. 2002; Watanabe 2003).

6.4. Conclusions and Outlook

6.4.1 General

Within two months, mature compost was produced out of biogas sludge at ambient temperatures. The final material had improved in handling and transportability: no smell, reduced water content, earth like structure. Hygienic quality was better, too.
In general, composting is a suitable post-treatment for biogas sludge. Composting can also be used for direct treatment of manure. Detailed conclusions and recommendations are given in the following, split into process and application aspects.

6.4.2 Process and Experimental Set-up

Composting reduced the pathogen concentration. Management has to focus on temperature conservation. By keeping high temperatures in all parts of the pile, degradation and pathogen die-off will increase.
Instead of labour-intensive active management - which did not improve the process in this study - insulation is a simple means to reduce heat losses and achieve high temperatures. Otherwise longer composting times are recommended to ensure sufficient pathogen die-off although this will increase the nitrogen losses.
Inoculation with a suitable micro-flora is recommended by several authors to initiate the process. Especially biogas sludge with an anaerobic flora may otherwise need more time to establish a "composting flora" (Anh 2002). Therefore an addition of 20% mature compost is recommended (Anh 2002; Bidlingmaier 2000). Other sources deny the enhancing effect of inoculation (Jenkins 2005).

Regarding the co-substrate, biogas sludge with rice straw performed better than sludge without straw. This is most likely due to the fact that rice straw brought in structure and improved aeration. Furthermore the C/N ratio was adjusted.
The tested mixing ratio of 20:1 (sludge to straw) worked well. According to literature, further improvement could be expected at ratios of 10:1 or even lower due to better C/N ratios.
Besides, use of rice straw for composting proofed not only favourable for the process itself. It also reduces the amount of straw that is usually burnt (thereby causing air pollution and bad smell).

Besides rice straw, other organic waste could be composted too. Among others, sugar cane bagasse or market wastes are available in large amounts.
Bagasse is a residual product of sugar cane processing, containing high amounts of valuable phosphorus (which is introduced within the process). As it contains a considerable amount of lignin it can not be degraded in an anaerobic process. Compared to straw manure, the structure of bagasse and market waste is a lot softer. Thus, their contribution to structure improvement would be lower. Nevertheless, both are interesting materials for composting.

If the slurry is composted with other materials, the dry waste materials around the farm and homestead can be recycled, too. Due to the added material, an increased amount of compost will be available in the farm.

This experiment was carried out open-air during the "sunny season". When carrying out composting during other times of the year, heavy rain and higher air moisture have to be considered. For controlled conditions, a roof is recommended.

6.4.3 Application and implementation in the Mekong Delta

In the Mekong Delta, waste and waste water are discharged to the surface water to a large extent. Thereby nutrients and organic matter are dissipated and pathogens are spread in the environment. Waste treatment in general and composting in particular are not popular in the Mekong Delta.

However, reuse of organic waste would reduce the water contamination and by composting even excrements can be transformed to hygienically acceptable products for reuse in agriculture. The conducted experiment demonstrates that this kind of treatment is technically feasible without big investment of cost or labour.

At the moment the following recommendations for using biogas sludge and pig manure composts are given:

- Use for fruit trees: this kind of application is highly recommended, preferably for material from non-controlled processes as low hygienic quality is required. Especially in old orchards with low carbon and dense structure compost application is expected to improve the conditions (Shibistova *et al.* 2009). Orchards account for ~20% of the agricultural area (see chapter 1).
- Use for upland crops: the use for upland crops can be enhanced at places where an increase of organic matter is favourable (e.g. old, degraded soils). Application may be difficult due to transportation problems and the resulting work load. The application could be enhanced (so far upland crops are a minor area only 5% of the regarded area).
- Use of compost as a potting material is suitable.

The use of compost in paddy rice fields which account for around 75% of the area is not recommended due to additional carbon input and resulting CH_4 emissions.

For actual implementation, farmer's acceptance is the key factor. In this regard, the transformation to a pleasant and hygienic safe substrate seems to be important for the acceptance as handing without nuisance was regarded essential by farmers in the Mekong Delta (Wieneke 2005).
As production and application of compost is more labour intensive than use of mineral fertiliser (e.g. low nutrient content, transportation), farmers have to be positive about additional effects, like supplement of additional micro nutrients, structural improvement etc. (Reloe 1993).
Therefore more information, e.g. on composting techniques, effects on water and soil, risks etc. has to be supplied. Furthermore demonstrations of successful application are required in the villages. This is quite important in the Mekong Delta as the main path for information access for farmers (~80%) was found to be neighbours, friends and relatives (Wieneke 2005).

7. Vermicomposting

"It may be doubted whether there are any other animals which have played so important a part in the history of the world, as have these lowly creatures."

Charles Darwin

7.1. Introduction

The transformation of organic material by the activity of earthworms (Oligochaeta) is called vermicomposting (from Latin "vermis"=worm). Essentially, earthworms function as natural bioreactors by taking up and digesting organic waste to a peat like substrate. Microorganisms are involved in this process too but in contrast to normal composting, no increase in temperature takes place ("cold rotting").

The produced vermicompost can be used as a soil amendment: besides nutrients, vermicompost contains organic matter which is acknowledged for improving various soil conditions, e.g. structure, water holding capacity, ion exchange capacity (Edwards & Arancon 2004a; Scheffer 2002). Microorganic communities may be enhanced (Arancon *et al.* 2003a). Its effect on crop yields has been demonstrated (Atiyeh *et al.* 2001) and it may suppress plant pests and soil born diseases (Edwards & Arancon 2004b; Edwards *et al.* 2007).

"Vermicomposting" is focusing on compost production, whereas the similar process, "vermiculture" is aiming on worm growth. Worms are not only known for cultivating soils, their biomass also presents a protein rich fodder for fish, hens, pigs and others (Harwood & Sabine 1978; Sabine 1978).
"Vermiconversion" (Aalok *et al.* 2008; Gajalakshmi *et al.* 2001), or "vermifiltration" (Sinha *et al.* 2008) are further, more rarely used terms related to the biological treatment by worms.

7.1.1 Process

The process of vermicomposting is based on the activity of worms. The organic waste is ingested and broken up by worms. Worms can take up more than half of their own biomass per day, consume the digestible portion and excrete about 15% of their uptake as castings (Buch 1986; NIIR 2004).

The better the food supply the more castings or vermicompost is produced (Edwards & Bohlen 1996). The process depends mainly on the quality and quantity of supplied organic matter (Edwards 1988). For natural ecosystems up to 100-200 t ha^{-1} a^{-1} of casts are produced (Gisi 1997; Scheffer 2002), in culture even higher production rates are possible. In general, worms prefer animal waste with a lot of nitrogen but they live on all kind of organic waste, if possible mixed and pre-composted (Gunadi *et al.* 2002). Plant material is preferably soft and water rich. Microorganisms, especially protoza and fungi constitute an important nutritional component of the earthworm diet (Arancon *et al.* 2003a; Edwards & Bohlen 1996; Tomlin 2004). The best C/N ratio for earthworm feed is 16 to 18:1 (Buch 1986) but earthworms are able to live in material with higher C/N ratios (Edwards & Bohlen 1996).

During digestion in the worm, carbonate is added (by a gland in the gastroesophageal vestibule) and clay-humic complexes are formed when the organic material is merged with small mineral particles in the gizzard, the "muscle stomach" (Edwards & Bohlen 1996). These clay-humic-complexes are well-known for their ability to increase ion exchange capacity or

water holding capacity in soils (Scheffer 2002). Within this digestion process, nutrients are concentrated, too. For P, a 400-fold concentration in the worm and a 6-fold in the casting is mentioned (Gisi 1997).
Furthermore, biochemical components within the slimy lubricating mucus in the castings, seem to have an anti-pesticidal against plant diseases (Buch 1986), effects of phytohormonal activity by Cytokinins, Auxins, Gibberelins (Fleddermann 1990), or humic acids (Arancon *et al.* 2003b) were published.

To establish a successful vermicomposting process, appropriate worms have to be selected. Besides their tolerance and adaptation capacity to organic wastes, worms for vermicomposting should be able to convert a lot of material in a short time, have a high reproduction and a short development period from cocoon to adult.

Principally, out of three morpho-ecological groupings - epigeic, endogeic and anecics worms (Bouche 1977) - the epigeic species are suitable for composting as their natural habitat are organic horizons. They also tend to be small with rapid generation times.
The other grouping, endogeic species forage below the surface, in organic rich soil, and build mostly horizontal burrows. Anecics species penetrate the soil deeply by vertical burrows, e.g. *Lubricus terrestris* L. (dew worm) (Gisi 1997).

Of a worldwide total of almost 4,000 described megadrile[9] species, detailed ecological studies have been made on fewer than 20 of these (Reynolds 1998). Only a few species have been implemented in vermiculture so far - a lot more worms should be adaptable to this kind of environment (Edwards & Bohlen 1996; Ziegelasch 1984).

Four common earthworms and their requirements are listed in Table 7-1 (Edwards 1988; Edwards & Bohlen 1996; Tomlin 2004).

In general, earthworms are very sensitive to touch, light, and dryness (Edwards & Bohlen 1996; Sinha *et al.* 2008). Oxygen supply is necessary although earthworms can survive in relatively low oxygen and high carbon dioxide environments. They even stay alive when submerged in water if it contains dissolved oxygen (Edwards & Bohlen 1996). Nevertheless, usually the material should feel crumbly-moist, not soggy-wet although tolerable moisture depends on the material.
The tolerable pH range for the selected species is about pH 5 to 9; there are others being more acid tolerant. For commercial production, earthworm beds should be kept at a pH range of 6.8 to 7.2.

Table 7-1: Required optimum and tolerable environmental conditions for raising composting worms (Rivero-Hernandez 1991)

Parameter	*Eisenia fetida* Red wiggler, Tiger worm	*Dendrobaena veneta*	*Eudrilus eugeniae* African night crawler	*Perionyx excavatus* "Trun Que"
Temperature	15 – 20°C (4-30°C)	15 - 25°C	9 - 30°C	9 - 30°C
Moisture content	80 - 90 (60 - 90)%	65 - 85%	70 - 90%	70 - 90%
pH	7 - 8 (5 - 9)*	5 – 9	5 - 9	5 – 9
NH_3	< 0.5 mg g^{-1}	< 0.5 mg g^{-1}	< 0.5 mg g^{-1}	< 0.5 mg g^{-1}
Salinity	< 0.5%	< 0.5%	< 0.5%	< 0.5%

[9] megadrile = terrestrial group of earthworms; forms - together with microdrile species - the subclass of Oligochaeta (of the phylum Annelida)

Depending on the kind and size of earthworms their area demand will be different. In consequence, a vermiculture with small worms will have a higher worm density than the one with big worms (Edwards & Arancon 2004a; Tomlin 2004).

Earthworms are sensitive to NH_3, NH_4^+ (e.g. Ammonium-sulphate), to salts and heavy metals (e.g. Cu-sulphates). Besides the single factors, their combination is important, too (Nam *et al.* 2008).
For pesticides, some are known to have a toxic effect (e.g. Lindan, Eldrin, Endrin), others e.g. fast-degraded herbicides may not influence earthworms (Buch 1986).
Earthworms are also at risk by predators like birds, lizards, ants, toads, or snakes.

On the other hand, earthworms may transmit animal parasites either as an intermediate host or merely as reservoir host. Earthworms are essential intermediate hosts to a number of tapeworms (cestoda) and gape-worms (nematodes). *Lumbricus* sp., *Allolobophora* sp., *Eisenia* sp. are known to be alternate host for pig-lungworms (*Metastrongylus* sp.). They may also be a vector of viruses or protozoa (Edwards & Bohlen 1996).

The status of a vermiculture can be evaluated by the morphological appearance of the worms and by the number of adult, young worms and cocoons. In good living conditions where reproduction is regular all three groups are present. When conditions get worse, proliferation increases, and the number of cocoons is raised. Earthworms, like all Oligochaete are bisexual but need a partner for reproduction (biparentally).

7.1.2 Common Techniques and History

Earthworms are regarded as one of the most important organisms for fertile soils in sustainable (agro) ecosystems worldwide. This understanding had been subject to controversy for a long time in history.

Early, Aristoteles had called earthworms the “intestine of earth”, and Cleopatra apparently prohibited exporting worms from the country.
Later, in the 18th and 19th century Hönert, Giebel and others were afraid earthworms would damage plants whereas White (1720-1793), Darwin (1881, reprint 1983) or Steiner (1924) recognised the importance of worms for the soil (Buch 1986).

Earthworms were collected and used for different reasons, e.g. as fodder, bait, but also as a remedy for different diseases (Graff 1983). At various places worms have been consumed by humans, a aspect that can be seen at the ending name “edulis”, “esculentus” (Meinhardt 1986).

“Professional” cultivation of worms, vermiculture, started the in the US last century in the 30´s when an agricultural engineer wanted to use the potential of worms in his orchard (Scheffield Oliver 1937, in Buch 1986).
Nowadays the earthworms gained interest due to their capacity to degrade waste, and regain fertiliser from bio-wastes and agricultural by-products. However, worm farms are not only based on waste substrates only, but may also use commercial feed.

Vermicomposting is carried out in small scale, home composting and on technical, “industrial” scale. Small scale vermicomposting is usually done in boxes or baskets:

- non-continuously in an undivided container (“batch-flow”)
- continuously in a series of trays stacked vertically or in a series of trays lined horizontally (using the fact that worms after finishing on tray migrate to the next one).

Industrial scale plants may operate by the following techniques:

- beds or windrows
- barrel or drum composting
- automated continuous flow vermicomposting systems, mainly as indoor technique (Arancon *et al.* 2007).

There are large plants especially in the US or Japan, treating waste sludge, e.g. pulp (Aalok *et al.* 2008), animal waste, sewage sludge and others (Edwards & Neuhauser 1988). Special devices, like mechanical separators for worms and biomass were developed (Price & Phillips 1990).

Recent research has covered wastewater treatment, too (Bajsa *et al.* 2003; Garg *et al.* 2006; Heck *et al.* 2008; Hughes *et al.* 2007; Sinha *et al.* 2008).

Some research has been done on source separated brown water (Birkel *et al.* 2009; Shalabi 2006; Simons 2008).

Earthworms have also been investigated for remediation of contaminated soils (Ceccanti *et al.* 2006; Safwat & Weaver 2002).

Vermicomposting is often carried out to degrade organic waste. Frequently it is accomplished to gain vermicompost for later use as a soil amendment. Besides, production of worms is intended which might be used as fodder or bait. They also received interest in pharmaceutics, e.g. dried earthworm powder is used as an active ingredient (US-Patent-5128148 1992).

7.1.3 Vermicomposting in Vietnam

Vermicomposting was not found to be a common practice in Vietnam (Anh 2002; Hellebaut 2001). However, a number of vermicomposting farms could be identified in several provinces of the Mekong Delta.
In the peri-urban area of Can Tho, 3 % of the interviewed farming households (7 out of 218) practiced vermicomposting. Nearly 60 % of the households had heard about the technique and 17 % were interested to start in the future (Wieneke 2005).

Their major motives to start with were profitability, better handling and hygienic improvement of their starting material, which was usually pig manure. They all assumed that the organic fertiliser will be of higher quality.
The main purpose of vermiculture was mostly business by selling worms and/or fertiliser substrate. Sometimes further products were created, e.g. one farm was growing worms as fodder for turtles raised for restaurants. There, cow manure was used as the initial substrate.

Composting worms were *Eisenia fetida* or *Perionyx excavatus* (Vietnamese name: "Trun Que") which is growing popularly in Asian countries, such as Philippines, India, or Australia for biowaste control (Xuyen *et al.* 2008).

7.1.4 Summary and Research Questions

Vermicomposting transforms organic waste to a peat like substrate and allows the growth and proliferation of worms. The technique has been used in the Mekong Delta already by a few operators to grow worm biomass and more farmers showed interest to adopt the system.

According to various authors a lot of different organic substrates may be used for vermicomposting (Edwards & Arancon 2004a; Edwards & Neuhauser 1988; Frederickson & Knight 1988).

In contrast to composting the material does not heat up during the process. "Cold-rotting" therefore affects nitrogen changes and pathogens differently, although the main characteristics of the final material are similar to compost (Table 6-5).

There are still research questions regarding the efficiency of the process in the tropical climate (especially nutrient changes, hygienisation) and the plant response to these substrates. Vermicomposting experiments were set up to answer the following questions:

- Is vermicomposting a suitable post-treatment for biogas sludge?
- Can nitrogen losses be avoided by vermicomposting?
- Is pathogen reduction sufficient for reuse in agriculture?

Processing of anaerobically digested animal waste was compared to direct treatment of pig and cattle manure. Main interest was put on process parameters, nutrient transfer, pathogen reduction, and plant acceptance.

7.2. Material and Methods

7.2.1 Set-up

Besides biogas sludge (BS), two further substrates were investigated: pig manure (PM) and cattle manure (CM). Cattle manure was selected as it is a common substrate for vermicomposting in the region. Pig manure is far the most frequent substrate available; biogas sludge also derived from digested pig waste.

Four replications of each substrate (BS, PM, CM) were set up in bamboo baskets (Figure 7-1). The baskets were chosen to allow aeration. Saw dust was put underneath in order to keep an adequate moisture level. After adding the worms (grown at a worm farm in An Giang Province) the baskets were covered by a net to keep out predators like lizards or rats. They were stored on metal sideboards in a locked shelter at Can Tho University (Figure 7-1).

Figure 7-1: Baskets stored in a shelter at Can Tho University and shelter from outside

The mean daily temperatures in the shelter were 28-32°C, the humidity was around 85%.

The worm species *Eisenia fetida* Savigny 1826 (Oligochaeta, Lumbricidae) was chosen because of its special adaptation to waste and its tolerance to environmental conditions. *E. fetida*, known as manure worm, red wiggler, or tiger worm (in German: "Mist-, Kompost-wurm"), belongs to the family of Lumbricidae, within the class of Ogligochaeta (Rota 2008). With a diameter of around 2 mm and a length of 3 to 14 cm, it weighs 50 up to 350 mg. For one live cycle from egg to reproductive worm it needs 10-20 weeks or 52 days at laboratory conditions (Tomlin 2004).

Substrate was kept at moisture of 70-80%. Moisture was checked daily with the fist test on-site, after having done "calibration" measurements in the lab. Water was sprayed in a way that no excess water left the experimental baskets. Details on supplied masses are given in Table 7-2.

Table 7-2: Supplied substrates for vermicomposting

Initial Substrate		Storage before application	kg $_{wet\ mass}$	kg $_{dry\ mass}$	Worms (g)
Biogas sludge (BS)	Fresh + composted biogas sludge 1:1	1 day	4	1.7	200
Pig manure (PM)	Pig manure, mix from two farms	3 weeks	3.5	1.5	200
Cattle manure (CM)	Manure from animal husbandry at CTU	3 days	4 ($3.5_{t=0}$ + $0.5_{t=44}$)*	0.9	200

*3.5 kg were supplied initially, 0.5 kg manure were added when necessary on day 44

7.2.2 Measurements

Chemical and microbial analysis was carried out according to the methods given in Table 7-3. Most parameters were measured separately for each replicate. Only microbiological analysis was carried out for one mixed sample of all four replications.

Table 7-3: Analysis methods

Parameter	Method	Method Principle	Instrument	Laboratory
Shape and colour	Organoleptic		Eye, nose, ear	On-site
Temperature	Measurement		Thermometer	On-site
PH	VDLUFA A 5.1.1	Electrode	WTW, MultiLine 340i	On-site
Electric conductivity	VDLUFA A 10.1.1	Electrode	WTW, MultiLine 340i	On-site
Dry matter	VDLUFA A 2.1.1		Memmert, Oven	CoT, Can Tho
Volatile Solids (Glühverlust)	VDLUFA A 15.2		Lenton, EF 11/88	CoT, Can Tho
AT_4	LAGA 1995		WTW, Oxitop OC 110	CoT, Can Tho
N_{tot}	VDLUFA A 2.2.1	Kjeldahl	Gerhard, Kjeldahltherm KB20 & Vapodest 20	CoT, Can Tho
P	*	Photometric	Eppendorf Digital Photometer 6114	IPE, Bonn
K	*	AAS	Eppendorf, Elex 6361	IPE, Bonn
Zn, Cr, Cu, Ni, Pb	VDLUFA A 2.4.3.1		Perkin Elmer, AAS 1100 B	IPE, Bonn
Cd	VDLUFA A 2.4.3.1		Perkin Elmer, AAS Zeemann 3030	IPE, Bonn
Protein	Weeder food analysis			IPE, Bonn
E. coli	ISO 9308-2	MPN	Incubator	CoT, Can Tho
Total coliforms		MPN	Incubator	CoT, Can Tho
Salmonella	ISO 6579	Growth on agar plate	Incubator	CoT, Can Tho
Helminth eggs	WHO guideline for analyses wastewater	Microscopy after centrifugation		CoT, Can Tho

The maturity of vermicompost was determined by measuring microbiological respiration activity in the AT_4–test (LAGA 1995). Additionally, plant acceptance was tested by germination and growth of garden cress (*Lepidium sativum*).

The experiment was carried out mainly by Julia Fuchs who conducted her diploma thesis in Can Tho in 2004. Laboratory analysis was supported by Tran Khuu Tien, Nguyen Van.

7.3. Results and Discussion

7.3.1 Management and Process Parameters

Management

The first challenge was to start the vermicomposting process by getting one batch of worms into different new environments.
In pre-tests it was found out that transportation of worms is best when kept in their previous media, and then let them change slowly to the new one.
Cattle manure was accepted best by *E. fetida*. Pig excrements were accepted after a storage period of 2-3 weeks. As fresh (1 day old) biogas sludge had been rejected in the pretest, a mixture with more solid, composted biogas sludge (1:1) was provided.

Two days after the experiment had started, the first worms left the biogas sludge for the saw dust in the bottom layer. Migration started in the other substrates - PM and CM - after one week, too.
One reason may be that nutrient relocation from substrate into saw dust had occurred (see chemical analysis next chapter). These migration activities mixed the initially separated layers.

CM needed less watering compared to the other two substrates, indicating a better water holding capacity. Watering had been necessary quite frequently, usually daily.
To facilitate this, plastic boxes may be used instead of bamboo baskets. This has been proofed successfully in Germany with faecal matter (Simons 2008; Simons *et al.* 2005) and in Vietnam with biogas sludge (Xuyen *et al.* 2008). Then saw dust can be saved and mixing is avoided.

An other observation was that the worms´ colour in the biogas vermicompost (BV) was paler compared to the ones from pig and cattle manure vermicompost (PV, CV).

Process Parameters

Temperature was close to ambient temperature between 25-30°C. The material did not show any self heating.
The pH of the three substrates differed (Figure 7-2): highest pH was found in cattle manure (pH>8), lower pH in pig manure (pH 7-8), and lowest pH in biogas sludge (pH~6). All substrate pH were in the suitable range for the growth of *E. fetida* although biogas sludge tended to the lower end of the optimum range, as (Rivero-Hernandez 1991) pointed out that the worms prefer soils with a pH between 7.0 and 8.0.

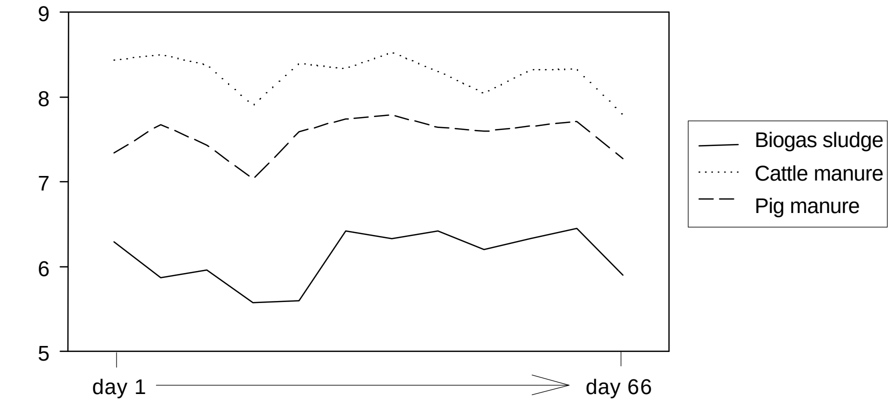

Figure 7-2: Development of pH in different substrates during 66 days of vermicomposting

In all substrates, the pH level decreased after some days of incubation, and increased again to about then initial level. Differences were less than one pH-unit per variant. The reason for these pH differences may be due to changes in microbiologic activity (leading to release of acids) caused by the initial worm introduction. Analysis on microbiological activity should give further information on this topic.

7.3.2 Mass Development

Dry matter was reduced significantly in all substrate variants (Figure 7-3). Within 66 days dry matter of initial substrates was reduced by 34% (biogas sludge), 46% (cattle manure) and 49% (pig manure).

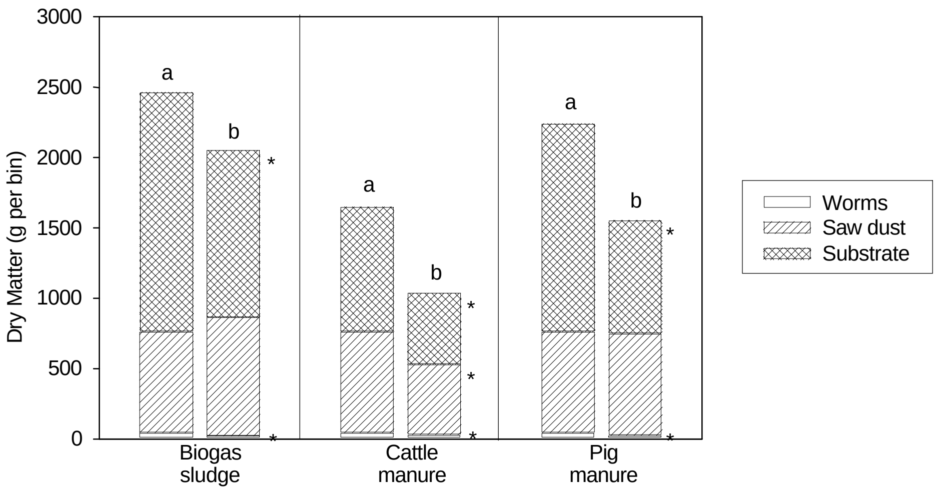

Figure 7-3: Dry mass at the beginning and the end of vermicomposting, per bin ("a, b" significant differences in total mass, "*"significant difference of the component)

Saw dust was reduced by 30% in cattle manure; its mass stayed the same in pig manure and increased in biogas sludge. Degradation of lignin rich saw dust with a C/N ratio of 500:1 will be marginal, so these changes are more likely due to mixing effects. Supposing no degradation of saw dust, net reduction of biogas sludge was 27%. In contrary, cattle manure degradation would be higher. Then net degradation of cattle manure added up to 71%.

Volume reduction was similar to the reduction within the composting process (chapter 6). Compared to other authors - reporting reductions to 15% (Buch 1986; NIIR 2004), or 25% (Mathur *et al.* 2006) of the initial weight - the achieved one was lower.

Organic dry matter (Figure 7-4) developed similar to total mass: highest degradation was found in cattle manure (63%), less in pig manure (39%) and lowest in biogas sludge (36%).

The total worm weight (dry matter) was lower at the end than at the beginning: the worm mass decreased from 42 g to 29.0 g in cattle vermicompost, to 23.7 g in pig vermicompost, and to 21.4 g in biogas sludge vermicompost (31%, 43% and 49%, respectively).

Reasons for this decline may be unhealthy living conditions, or unpleasant or insufficient food supply. However, the process parameters, temperature, moisture, pH were in a range well suitable for *E. fetida.* Regarding the food supply, there was still a small amount of untreated substrate left in the buckets at the end of the experiment, indicating that there had been enough potential food available. Salt content and NH_4^+ are discussed later.

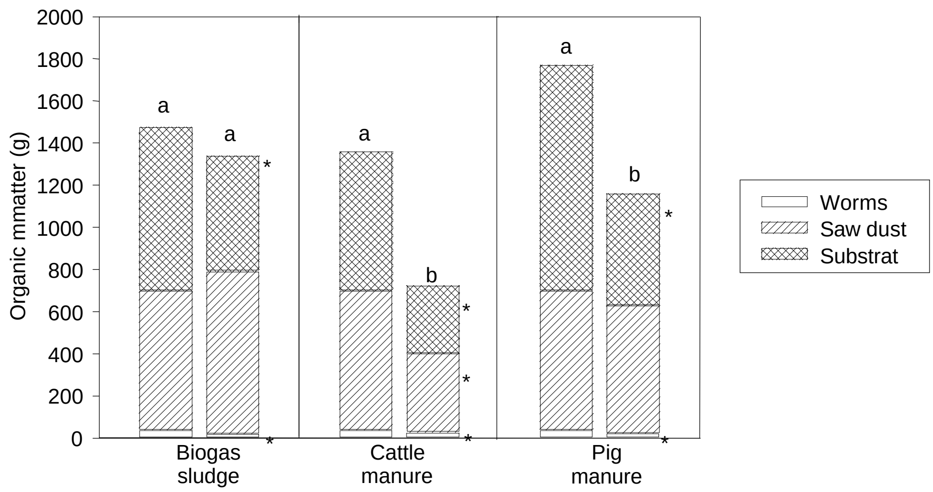

Figure 7-4: Organic dry matter at the beginning and the end of vermicomposting, mass per pile ("a, b" significant differences in total, "*"significant difference of the component)

The reluctance to fresh biogas sludge may have been due to low structure, still present anaerobic conditions and/or ammonia content. Addition of composted substrate permitted worm inoculation; nevertheless mass degradation and worm development were poor. Instead of compost, leaf litter and/or soil to biogas slurry is suggested as an amendment (NIIR 2004).

A study examining the mortality of *E. fetida* in different substrates (Gunadi & Edwards 2003) concluded that *E. fetida* cannot survive in fresh substrates of young pig, cattle, fruit and vegetable wastes. The death of worms in cattle manure is explained by the turn to anaerobic conditions after about two weeks, this leading to production of lethal substances, e.g. alcohol, ammonia (Frederickson & Knight 1988; Gunadi *et al.* 2002). Several authors demand storage times prior to worm inoculation, especially for pig solids as they tend to contain some NH_3 and develop high temperatures by self-heating.

Authors consent that in general cow manure is accepted most easily as fodder by worms (Edwards & Bohlen 1996) even though no specific reason is given. A result that was found in this experiment too: the fastest colonisation took place in cow manure.

The finding that pig manure is the most productive (Edwards & Bohlen 1996; Gunadi *et al.* 2002) could not be confirmed in this experiment.

7.3.3 Nutrients

Nutrient content and concentration of the substrates are important for application. Ideally no mass losses occur during treatment and the applicable substrate contains high concentrations finally.

The concentrations of the main nutrients (N, P, K) and the C/N ratio in the different untreated and treated substrates is listed in Table 7-4 and explained in the following chapters.

Table 7-4: Nutrient concentrations in the initial substrates and in vermicomposts (in % dry matter)

%	N	P	K	C/N
Biogas Sludge	0.79	1.58	0.17	36:1
Biogas Vermicompost	0.71	1.41	0.19	39:1
Pig Manure	1.23	1.80	0.73	36:1
Pig Vermicompost	1.11	2.06	0.68	35:1
Cattle Manure	1.54	0.39	1.35	30:1
Cattle Vermicompost	1.92	0.63	1.52	19:1

Nitrogen

Nitrogen concentrations in the initial substrates varied from 0.79% in biogas sludge to 1.54% in cattle manure (Table 7-4).
After 66 days, nitrogen concentration had reduced by 10% in biogas and pig vermicompost. By contrast, cattle vermicompost, ended up with a 25% higher N concentration. This concentration increase resulted from the mass loss of cattle manure (49%).

Compared to other organic wastes in the area, the intial nitrogen values were comparably low, especially the values in the examined biogas sludge.
N-concentrations of 2.6% and 2.4% in different fresh biogas sludges and 2.0% and 1.9% after 2 and 6 months of composting were found (Becker *et al.* 2005). Examined pig and goat vermicompost contained 2.2% and 3.0%, respectively.

The N mass in the cattle vermicompost, however, was reduced by 33% (Figure 7-5). This mass had been transferred into worms (19%) and into sawdust (10%). The remaining 4% of the balance were possibly emitted to the air. In absolute figures, the loss equals 40 g N per kilogram N, or 0.6 g per kg substrate, respectively. As this loss is very low, the total N balance does not show a significant difference between initial and final N content.

Biogas vermicompost contained 40% less N than the biogas sludge. 9% had shifted to worms and 24% to sawdust, 7% could not be recovered. Despite these alterations, the total amount of N did not change significantly from beginning to end.

In the pig manure systems, nitrogen was transferred from the manure into worm biomass and sawdust, too. Of 54% N less in the substrate, 13% were taken up by worms, 25% were found in saw dust. The remaining, 16% of the substrate-N or 14% of the total N could not be recovered, most probably due to gaseous losses. The mass loss of 160 g N per kilogram N or 2 (1.97) g N per kg substrate respectively was a significant mass loss for the system.

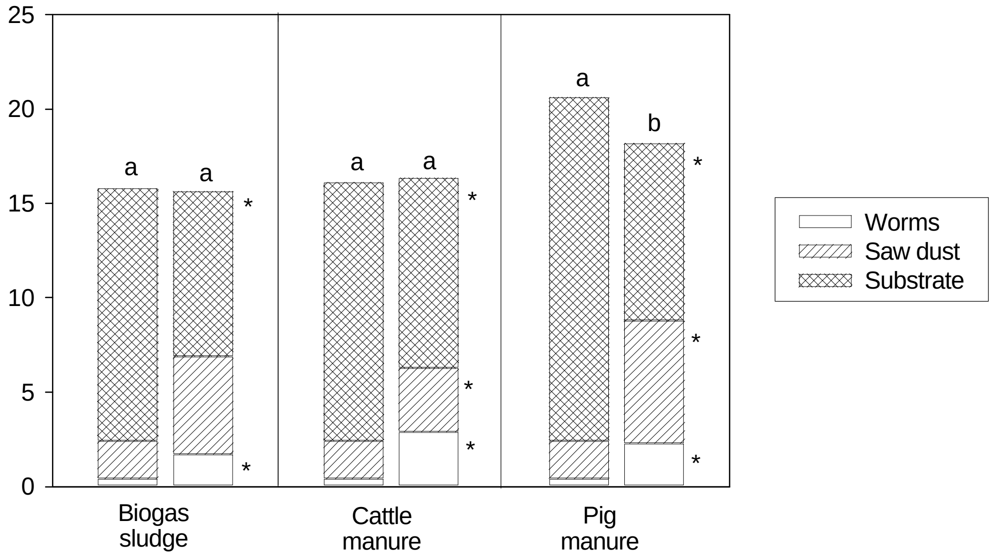

Figure 7-5: N mass in the beginning and end of vermicomposting (letters a, b indicate significant difference in total N-mass; * significant difference of the component at 5%)

Compared to hot-rotting (see chapter 6), nitrogen losses during vermicomposting were smaller.

Interestingly, the N-content in worms increased from 1% to 8 - 10%. Worms usually consist of 60 - 70% of protein (Buch 1986). For *E. fetida*, a percentage of 62 - 64% crude protein is given (Chan & Griffiths 1988). This amount is equivalent to about 10% N (taking the conversion: Raw protein = Kjeldahl N x 6,25). Nevertheless, there seems to be some variance, e.g. Simons (2005) found 2.2% N for *E. fetida* and 1.8% for *Dendrobena veneta*.

Carbon-Nitrogen Ratio

The C/N ratio in cattle manure was 30:1, in pig manure and biogas sludge it was 36:1 (Table 7-4). Compared to other studies, these values are quite high. Becker *et al.* (2005) found values from 12:1 up to 23:1 in ten different sludges and composts in the area. The comparably high values in the pig derived substrates (BS, PM) may be due to non digested rice husks (hulls) which were not degraded within the 66 days of vermicomposting either. The high silica and lignin content in rice husk prevented fast degradation (Ndazi *et al.* 2005).

Surprisingly, the examined biogas sludge did not have a lower C/N ratio than pig manure although biogas production leads to a loss of carbon. Therefore one would expect a lower C/N ratio in the digested material compared to manure. But possibly soluble nitrogen (NH_4^+) had been released, as with the effluent before, thereby increasing the C/N ratio. Additionally, fast degradable carbon structures (sugars, starch) are digested and the proportion of less degradable structures (cellulose, lignin) increases. Thus, a faster and higher organic matter reduction would be expected in pig manure compared to the biogas sludge, leading to a lower C/N ratio in the end. This effect could not be seen: the pig vermicompost´s C/N ratio remained at 35:1 and in biogas sludge C/N ratio even increased slightly. The transfer of nitrogen into saw dust and worms and the input of saw dust into the compost seem to have prevented a decrease of the C/N ratio (part "nitrogen" in chapter 7.3.3, Figure 7-5 and Figure 7-3). The analysis of the saw dust at the beginning and the end of composting - C/N ratio 500:1 and 200:1 – clearly demonstrates the input of nitrogen. C/N ratio decreased only in one substrate, the cattle manure, to a final value of 19:1.

Phosphorus

Initial phosphorus concentration of cattle manure (Table 7-4) was about one quarter of the concentration in the pig derived substrates (PM, BS). Cattle manure regularly contains less P than pig manure, although differences tend to be smaller (KTBL 2005). However, as contents depend on the feeding customs, a wide variation is possible.

Compared to other local studies, P values are in the same range. Becker et al (2005) found P concentrations mainly below 1%, except for a pig vermicompost and a pig manure-straw-compost with P contents of 1.39 and 1.65.

Mass balances showed that P was conserved in all systems during vermicomposting; differences between initial and final P-content were not significant (Figure 6-7).

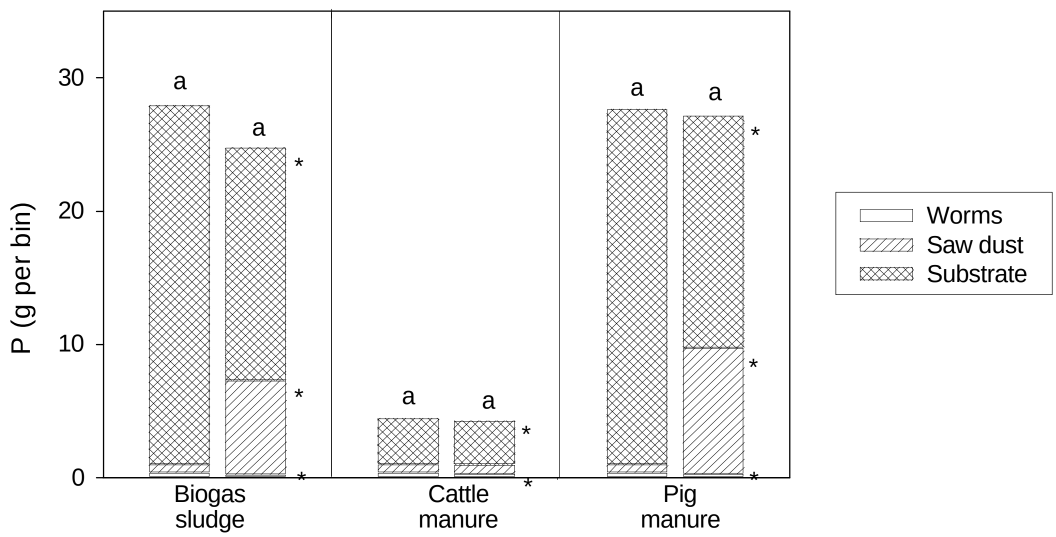

Figure 7-6: P mass in the beginning and end of vermicomposting (letters a, b indicate significant difference in total N-mass; * significant difference of the component at 5%)

Nevertheless, changes within the different compartments of the system occurred during the vermicomposting process: losses were significant for all substrates. Similar to nitrogen, P transfer rate was higher for biogas sludge and pig manure compared to cattle manure.

As only total phosphorus was analysed, no details on plant availability can be given in this study. However, a shift to more soluble P fractions during vermicomposting has been reported (Edwards & Arancon 2004b; Gisi 1997).

Potassium

The initial potassium content of biogas sludge was 0.17%, thus much smaller than the content of pig with 0.7% and cattle manure with 1.3% (Table 7-4).

The biogas sludge system conserved its initial potassium content during vermicomposting whereas the cattle and pig manure systems lost K significantly (Figure 7-7).

Contrary to the N and P balances where cattle manure showed lowest losses, K mass losses were highest in the cattle manure system (although the concentration increased.)
As potassium is not volatile, gaseous losses can not take place. An increase in saw dust or worms could not be noticed. Excluding an analytical mistake or loss by percolation, adsorption is left as an explanation. The fact that the pH of the two substrates showing K losses (CM, PM) was higher than the one without loss (BS) might support that explanation, supposing that the bamboo basket had a negative surface load. However, only further analysis can proof this "adsorption theory".

At outdoor composting sites high K losses are occurring regularly when the well soluble element is washed out with precipitating water is not gained and recycled (Gottschall *et al.* 1995).

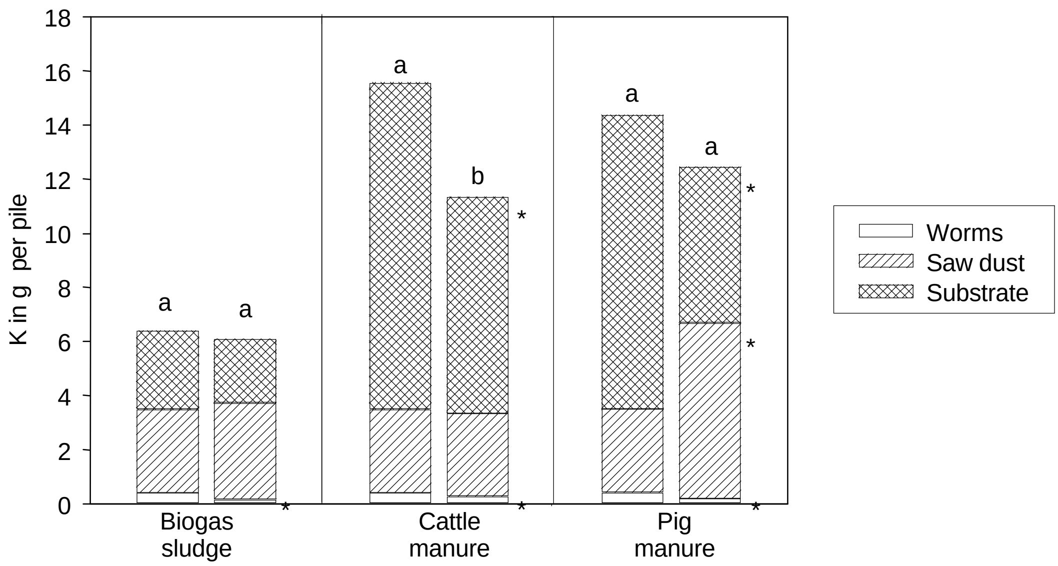

Figure 7-7: K mass in the beginning and end of vermicomposting (letters a, b indicate significant difference in total N-mass; * significant difference of the component at 5%)

7.3.4 Hazardous Substances

7.3.4.1 Hygienic Parameters

All hygienic parameters measured (*E. coli*, coliforms, helminth eggs) were improved considerably during the two months vermicomposting process.

The number of *E. coli* and total coliforms dropped notably in all three tested substrates, in most samples ~99% (2 $\log_{10}$ units). Detailed results are listed in Table 7-5.

The highest concentration of *E. coli* remained in the pig manure vermicompost, containing 8.3×10^2 MPN g^{-1}. Total coliforms were highest in the biogas sludge vermicompost (1.6×10^3 MPN g^{-1}).

The number of helminth eggs was reduced significantly in the PV. Helminth eggs were not present in the other two vermicomposts although these had been detected in all initial samples.

Salmonella sp. which had been found in the biogas sludge at the beginning could not be detected in any compost sample at the end.

Table 7-5: Hygienic parameters in manures and vermicomposts

	E. coli MPN g^{-1}	Total coliforms MPN g^{-1}	Helminth eggs # g^{-1}	*Salmonella* sp.
Biogas Sludge	5×10^3	5×10^3	300	+
Biogas Vermicompost	1.5×10^1	1.6×10^3	0	0
Pig Manure	4×10^4	3.1×10^5	500	0
Pig Vermicompost	8.3×10^2	8.4×10^2	40	0
Cattle Manure	3.5×10^4	3.5×10^4	200	0
Cattle Vermicompost	1.1×10^2	7×10^2	0	0

The achieved reduction was about 2 log units (99%) in average, from ~10^4 to ~10^2 MPN g^{-1} for both groups of bacteria. Compared to other studies, this result is reasonable but at the lower end of the scale:

- In a study in India, faecal coliform bacteria dropped from 3.9×10^4 MPN g^{-1} to 0 MPN g^{-1} after 60 days of vermicomposting bio-solids from a hospital. In the same period, *Salmonella* sp. dropped from <3 MPN g^{-1} to <1 MPN g^{-1}. Furthermore, human pathogens did not survive vermicomposting and microorganisms like *Staphylococcus aureus, Proteus vulgaris, Pseudomonas pyocyaneae* and *E. coli* were not isolated any more after twelve weeks of vermicomposting (Mathur et al. 2006).
- In experiments with faecal matter, reduction rates of 99.9% after two months and 99.99% after 6 months were found for *E. coli* (Rechenburg 2005) despite lower temperatures in Germany.
- In a study in Florida, a faster reduction of indicator organisms,faecal coliform, *Salmonella* sp., enteric virus, helminth ova in biosolids (= waste water treatment plant sludges) composted with and without *E. fetida* was shown (Eastman 1999). Within 6 days of vermicomposting high reduction rates were achieved, e.g. a 6.4 log reduction of faecal coliforms. US standard (EPA biosolids Class A) was reached.

Without heat development and thus no thermal inactivation of microorganisms during vermicomposting, there must be other reasons for the die-off. The most probable ones are storage time of the substrate and inactivation by worms´ digestion.

Regarding earthworm´s digestion, an interesting result has been gained when studying vermicomposting of faecal matter (Rechenburg 2005): *E. coli* were detected in low concentrations (average 25 CFU g^{-1} *E. fetida). Salmonella* sp. were determined in the worm species *E. fetida* after 3 months of composting but were not found in the other worm species tested.

Due to the low concentrations, the actual risk is estimated rather low. Nevertheless this finding – earthworms as carrier of pathogens - has to be kept in mind especially when worms are used as fodder or moved to new substrates. Even though a number of results on earthworms as carrier of pathogens is available (Edwards & Bohlen 1996) this topic needs to be evaluated further.

7.3.4.2 Anorganic Salts and Heavy Metals

Salt content is an important parameter in vermicomposting as high salt concentrations may inhibit worm and plant growth. The salt concentration of the vermicomposts is listed in Table 7-6.

Table 7-6: Salt concentrations in vermicomposts

	EC	Salt content mg g^{-1}	Salt content %
BV	6.1	3.7	0.4
PV	7.5	4.3	0.4
CV	7.7	4.4	0.4

In general, concentrations up to 0.5% seem to be tolerable for most compost worm species (Edwards 1998), although depending on the author, the worm species and the kind of salt, thresholds differ.
Kerr & Stewart (2006) describe higher salt tolerance for *E. fetida* than for other worm species: in a study of Sinha *et al.* (2008) reproduction was still 80 - 100% at 1.5% salt content. Higher earthworm mortality was found for NaCl (from 0.5%) compared to other salts, e.g. KCl (1%) (Srinivasan 2007). New results indicate that rather high salt concentrations can be tolerated by *E. fetida* if NH_3 concentration is low (Nam *et al.* 2008).

Salt content is important factor when applying a substrate in agriculture. Low to moderate salt concentrations are usually best; an increase is rarely desired in agricultural soils. The limits depend on different parameters e.g. soil type, climate. A content > 200 mg salt 100 g^{-1} soil - in carbonate free soils - may harm plants (LUFA Methodenbuch I 1991). With increasing carbonate content, twice the amount is no problem for plants (Hasenbäumer 1931, in LUFA 1991).

Besides salts, heavy metals may harm worms and plants if the concentration is too high.

Cu and Zn were significantly higher in the two pig derived substrates compared to the cattle substrates (Table 7-7). High copper contents in vermicomposts from pig substrates had been found by other authors e.g. Edwards & Bohlen (1996). High copper values are frequently found in pig manure as a result of feed additives. This is discussed at length in the part on composting (chapter 6.3.2.3).
The Cu and Zn concentration in the pig-vermicomposts reached a level that is regarded as critical for application in Germany according to the BioAbfV (Bundesminsterium für Umwelt Naturschutz und Reaktorsicherheit 1998). Effects on cocoon production of worms were found at soil contents of 560 mg Zn kg^{-1} and higher (Van Gestel *et al.* 1993).

Different to Cu and Zn, the Cd concentrations were higher in the cattle vermicompost than in the pig derived vermicomposts. The higher Cd difference is possibly due to farm conditions.

Cr was higher in the vermicompost of biogas sludge compared to the ones of manure. Possibly, some metal parts of the biogas plant may have caused this result.

However, the concentrations are lower than the threshold for reduced cocoon production which was given for Cd with 10 mg kg^{-1}, for Cr with 100 mg kg^{-1} (Van Gestel *et al.* 1993). Cd, Cr, Ni, Pb were clearly below other common thresholds as well (Table 7-7).

Table 7-7: Heavy metal concentrations in manures and vermicomposts compared to international thresholds

	Cu mg kg^{-1}	Zn mg kg^{-1}	Cd mg kg^{-1}	Pb mg kg^{-1}	Cr mg kg^{-1}	Ni mg kg^{-1}
BS	131	336	0.23	17.4	22.1	15.4
BV	110	301	0.23	16	21	14
PM	91	328	0.17	7.0	5.3	6.4
PV	106	396	0.22	12	12	7.8
CM	17	87	0.20	9.0	6.5	6.1
CV	32	143	0.38	13	9	10
German-BioAbfV	100	400	1.5	150	100	50
EU range	70 - 600	210 - 4000	0.7 - 10	70 - 1000	70 - 200	20 - 200
US biosolids ord.	1500	2800	39	300	1200	420

As worms may take up heavy metals with the food, the initial concentration of worm was measured and is listed in Table 7-8.

Table 7-8: Initial heavy metal concentrations in *Eisenia fetida* worms in µg kg^{-1}

	Cu µg kg^{-1}	Zn µg kg^{-1}	Cd µg kg^{-1}	Pb µg kg^{-1}	Cr µg kg^{-1}	Ni µg kg^{-1}
E. fetida	26.7	107	0.47	1.94	6.8	2.6

The heavy metal concentration in the worms was lower finally for all elements except for Cd. Final Cd values were 1.36 µg kg^{-1} (BV), 1.22 µg kg^{-1} (PV) and 0.47 µg kg^{-1} (CV).

The transfer coefficient of Cd is reported with 11-22 (Ireland) in Edwards & Bohlen (1996), depending on the worm species. It seems to get lower the higher the concentration in the substrate, but it is generally higher than for other metals, e.g. Pb, Cu and Zn.

Bio-concentration and transfer factors from fodder to worm depend on the element, its concentration in the feed, and on the species. Reviews on accumulation of heavy metals in earthworms are given by different authors (Edwards & Bohlen 1996; Nahmani *et al.* 2007).

7.3.5 Maturity and Plant Acceptance

The maturity of vermicompost was determined by measuring microbiological respiration activity in the AT_4 – test (LAGA 1995) and the plant acceptance by the cress germination test.

Highest respiration was found in the cattle vermicompost with 11.4 mg O_2 g^{-1} VS, closely followed by BV. Activity in PV was considerably lower (Table 7-9).
Still, all three composts ranged within 6.1-16 mg O_2 g^{-1} VS, thus belong to category IV, labelled as finished compost with good decomposition[10].

[10] Scale ranges from I (raw compost) to V (finished compost with finished decomposition)

Table 7-9: Compost maturity according to AT_4 and cress test (LAGA 1995)

	AT_4 mg O_2 g^{-1} VS	Degree of de-composition	Germination in %	Growth	Root length	Classification of Plant acceptance
BV	10.7	IV	70	Like standard soil	Like standard soil	2
PV	6.5	IV	60	Reduced	Reduced	3
CV	11.4	IV	100	Like standard soil	Like standard soil	1

Vermicomposting farms in the area operate in continuous feed systems for 6 months and thus the results are difficult to compare with the batch experiment described here.

The results of the cress test showed best response for the vermicompost of cattle, good results for biogas sludge, and lower response for pig-vermicompost. On a scale from 1-5 with 1 (=100% germination, well grown) and 5 (=20% germination, no growth) the substrates were classed as category 1, 2, and 3, respectively.

Vermicompost produced in a subsequent study out of biogas sludge combined with water hyacinth, weed or kitchen waste showed good plant acceptance too. The cress test showed good results for all substrates, e.g. germination was 100% or 75% (Xuyen *et al.* 2008).

In field experiments, two vermicomposts from pig and goat manure were applied to upland crops (tubers, legumes, vegetables) on an alluvial Tropaquept and an acid Sulphaquent in the Mekong Delta (Chiem *et al.* 2005). Compared to eight other substrates (e.g. pig manure, biogas sludges) vermicomposts from pig and goat manure showed one of the best plant responses. In the trial with mung bean, only one type of biogas sludge had a higher yield.

Figure 6-7 shows the details on dry matter for stover and grain of mung bean supplied by different composts).

Other authors reported the positive effect of vermicompost on growth and productivity of marigold (Atiyeh et al. 2001), on tomatoes (Atiyeh et al. 1999) or on peppers (Arancon et al. 2005). Growth and yields of Capsicum annuum were significantly higher when vermicompost of cow manure and paper waste had been applied (at 10 t ha^{-1} and 20 t ha^{-1}) compared to mineral fertiliser (at the same nutrient rate of NK 95 kg ha^{-1}). Application of vermicompost had resulted in 14-16 t ha^{-1}, without fertiliser 12 t ha^{-1} were acheived. Besides, microbial biomass and dehydrogenase activity were found to be significantly higher with vermicompost (Arancon et al. 2005).

7.4. Conclusions and Outlook

7.4.1 Treatment Process and Experimental Set-up

The experiment showed that biogas sludge as well as animal manure can be treated by vermicomposting. Within the 66 days treatment periode, *Eisenia fetida* had transformed the initial waste substrates to a more peat like material. However, improvements are necessary, especially to increase worm proliferation.

Compared to the experiment on hot rotting (chapter 6), nitrogen losses were smaller, i.e. more nitrogen could be recycled instead of being emitted into the air.

Within the two months a significant pathogen reduction was reached. However, the values are still above the recommendations of WHO guidelines.

The following findings were gained as information for future set-ups:

- Worms get accustomed to a feeding material. In case of changes, it is necessary to allow some time for adaptation.
- Saw dust was not helpful as a water barrier but increased the C/N ratio of the substrates which had been quite wide already in the applied substrates. In other cases, however, saw dust may be a useful additive to biosolids (Herlihy 2002).
- The advantage of bamboo baskets (better aeration) did not win over plastic boxes which is an inert material, conserving moisture and nutrients in handling and balancing (Simons *et al.* 2005). For experiments with small volumes, plastic boxes seem to be the practical, feasible solution. For large amounts special devices may be more favourable (Heck *et al.* 2008).
- For evaluation, adults, young worms and additional counting of cocoons should be counted separately to provide additional data on proliferation and thus further information on effects of different treatments.
- Biogas sludge contains anaerobic microflora when collected and thus aerobic flora needs some time to settle in. In succeeding experiments where biogas sludge had been stored and incubated with *Trichoderma* sp. before application to the worms, worm growth and proliferation was higher (Xuyen *et al.* 2008).
  This incubation has an additional effect: *Trichoderma* sp. which is used for plant disease control (Chaoui et al. 2002; Minh et al. 2001; Sharon et al. 2001) is multiplied and may be introduced to the place of interest by worms or vermicomposting.

Regarding the living conditions of worms in organic substrates, more information is necessary, especially on limiting parameters and combined effects of two or more different parameters (Nam *et al.* 2008).

Further substrates and combinations should be checked for additional recycling potentials in the Mekong Delta. Quite obvious appears the use of market waste produced in large amounts every day. This is far more advantageous than the actual disposal on a landfill (with methane production, subsidence of ground).

Progress is expected by combining different organic wastes to improve C/N ratio and pH and provide a proper structure (Xuyen *et al.* 2008). The formulas can be adapted to increase worm production or to produce special plant substrates.

Demonstration of successful vermicomposting experiments – leading to a transformation of unpleasant waste to earth like substrate - may also increase acceptance to recycling of a usually refused filthy material: source separated human faeces has been treated successfully in Vietnam and in Germany (Birkel *et al.* 2009; Nam *et al.* 2008; Simons 2008; Simons *et al.* 2005). Goat manure has been vermicomposted continuously for several years at Hoa An research station of Can Tho University (Ni 2005). Vermicompost production in the Mekong Delta seems to be especially interesting for small to medium scale farms as small amounts of waste may be added more or less continuously. It can be started with small initial investment as a low-tech system and provide an extra income.

7.4.2 Application of Vermicompost in Agriculture

The study confirmed that vermicompost is plant acceptable and is a valuable organic fertiliser. However, the amount of material needed for agriculture is high (estimation: 10 – 20 t ha^{-1}) and its production is time consuming. Consequently, vermicompost is a product of high value. Compared to compost, higher prices were requested. In the Mekong Delta, vermicompost was sold at 5,000 - 10,000 VND kg^{-1}.

Due to these facts, the application might be preferably pot cultivation and horticulture, especially in peri-urban areas. Vietnamese favour the use of vermicompost for e.g. in tree nurseries, bonsai cultivation or for flower production.

7.4.3 Value of Produced Worms

Besides the vermicompost, worms or worm biomass can be used. Living worms can be a supplement to ameliorate compacted soils which are frequently found in the Mekong Delta. They may also serve as carrier of antagonistic microorganisms (Minh *et al.* 2005; Minh *et al.* 2001) although this topic needs more research.
In other countries, e.g. Canada, there is a big market for worms due to fishermen using esp. dew worms as bait (Tomlin 2004).

Worm biomass is a valuable fodder as it consists to 60-70% of protein. Its composition and the (essential) amino acid spectrum is comparable to standard feed and suitable for animal nutrition (Edwards & Niederer 1988). It provides long-chain fatty acids that cannot be synthesised by monogastric animals.

Compared to other protein sources e.g. soja bean, worm protein is more expensive, thus can usually not compete in price (Tomlin 2004). Due to his calculations, the difference is 10 $ vs. 2.5 $. In the Mekong Delta worms were sold at 150,000 to 200,000 VND kg^{-1} during the last years.

7.4.4 Evaluation and Comparision to Composting

Similar to composting, different organic materials can be treated by worms. Composting time is comparable, and the resulting product, the compost may be used for the same purposes.

Different to composting, nitrogen losses were smaller. The hygienisation within the same amount of time was smaller, too. However, this is still subject of controversy (Dominguez *et al.* 1997; Rechenburg 2005). A comparison between composting and vermicomposting is given in chapter 9 (Table 9-7).

8. Source Separation and Urine Storage

8.1. Introduction

Human excrements contain a lot of nutrients that could potentially substitute a larger part of mineral fertilisers for food production instead of being wasted and polluting surface and groundwater.

The amount and consistency of excreta depends on the nutrition and will therefore differ between countries, rural and urban areas and even between social groups (Feachem *et al.* 1983; Jönnson & Vinneras 2003; Pieper 1987).
For urine, volumes of around 1 – 1.5 l $pers^{-1}d^{-1}$ sum up to around 500 l $pers^{-1}$ a^{-1}.
For faeces, compiled data of more than 20 surveys revealed that Europeans and North Americans usually produce 100 - 200 g faeces per day compared to 130-520 g of persons in developing countries (Feachem *et al.* 1983). On average, around 50 kg $pers^{-1}$ a^{-1} of faeces can be expected, although a range of amounts is given by different authors (Table 8-1). Water contents range from 75-90% between the different groups. Dry masses of faeces will usually be higher where more vegetables are eaten, e.g. in developing countries.

Table 8-1: Amount of urine and faeces per person and day in different countries

Urine l d^{-1}	Dry matter g l^{-1}	Faeces g d^{-1}	Dry matter	Country	Source
1.25	40	--	--	Germany	(Roche 1984)
1.25	-	123	12-25%	not defined	(Lange & Otterpohl 1997)
1.5	58	140	30 g	Sweden	(Vinneras 2002)
1.6		315		China	(Gao 2002 in Jönnson et al. 2004)
--	--	489	--	China	(Feachem *et al.* 1983)
--	--	520	--	Kenia	(Feachem *et al.* 1983; Pieper 1987)
1.0–1.3	50-70	100 - 400	30-60 g	summarised	(Feachem *et al.* 1983; Pieper 1987)
1.2		93		Germany	(Dunbar 1912, reprint 1998)

The major part of the most important nutrients (N, P, K) is excreted in urine whereas the major part of pathogens is emitted in the faeces together with organic matter (Otterpohl 2006; Roche 1984; Vinneras 2002). The main characteristics of urine and faeces are put aside in Table 8-2.

Table 8-2: Main characteristics of urine and faeces

	Urine	Faeces
Organic matter	No organic matter	Rich in organic matter
Nutrients	Plant available nutrients	Nutrients (N, P) partly plant available
N	~70 - 90% of excreted N (N-concentration 3 - 7 g N l^{-1})	~10 - 30% of excreted N
P	~25 - 60% of excreted P	Contains ~40 - 50% of excreted P
	P/N ratio smaller than in most mineral fertilisers	
K	~80% of excreted K	
Pathogens	Only few diseases cause excretion of pathogens in urine	Contains the main part of pathogens

Depending on the nutrition and physiological status, around 2.5 - 5 kg of nitrogen and 0.4 - 1 kg of phosphorus are excreted per person and year (Jönsson & Vinneras 2003; Jönsson & Vinneras 2004; Otterpohl 2001; Winker *et al.* 2009). However, reliable data on nutrient amounts in developing counties are scarce, and need to be estimated (Jönsson & Vinneras 2003).

8.1.1 Principle, History and Common Techniques

From the "invention" of fertilisation until the industrial revolution, animal and human excrements have been the main nutrient source in many countries (Barnard 2007; Finck 1991a). Human excrement has commonly been used as night soil, often untreated, to grow food (Höglund 2001; Jenkins 2005).

In China there is a long tradition of using urine and faeces. According to literature sources (Jenkins 2005; King 1911, reprint 1984; Shiming 2002) it started around 3000 years ago and has been diminished only during the last decades due to increasing availability of mineral fertiliser.

Nowadays, there are several different systems and different standards of toilets available (Jenkins 2005). Beside the traditional dry composting toilets (Figure 8-1) there are more high-tech solutions such as waterless urinals, low-flush separation toilets (Figure 8-2) or vacuum toilets with their respective pipe systems.

Figure 8-1: Simple composting toilets with separate collection of urine and faces

Figure 8-2: Low flush separation toilets, flushing for yellow water only (left), flushing for yellow and brown water (right picture)

The principle of a separation system is quite simple - different systems have been used to treat the separated fractions (Figure 8-3).

Nevertheless, even for the low-tech solutions experience and know-how is necessary to run it successfully. This knowledge - as well as the number of separation systems worldwide - is still limited, although an increasing number of projects worldwide have been started during the last years.

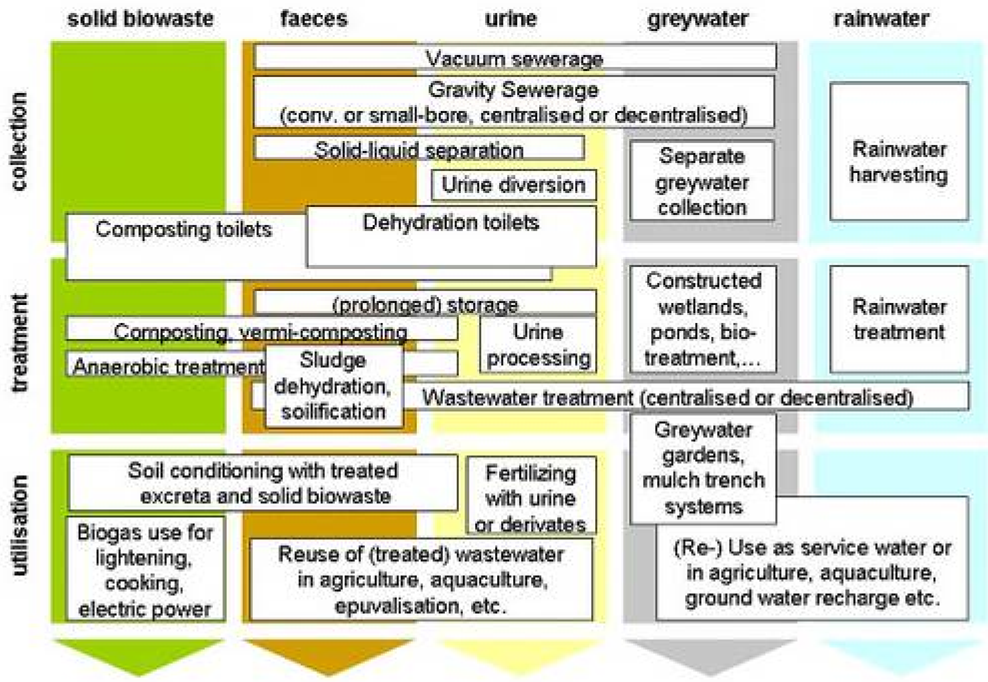

Figure 8-3: Technologies based on the principle of source separation (ecological sanitation, Gtz 2009)

Various projects in developing countries, especially the installation of composting toilets, were funded by Gtz ecosan (Werner 2003; Werner *et al.* 2007). Together with UNESCO and WHO around 180 projects in more than 40 countries are carried out worldwide (UNESCO 2008).

In Europe several pilot projects have been established in Sweden (Vinneras 2002), Switzerland (Udert *et al.* 2006), Austria (Linz AG 2008), Netherlands (Betuw *et al.* 2007), Denmark (Tarnow *et al.* 2003), and Germany (Bischof & Paris 2007; Oldenburg *et al.* 2003; Peter-Fröhlich *et al.* 2007; Werner *et al.* 2008). In Sweden an increasing number of private households are installing separation toilets (Jönsson 2008, personal communication).

8.1.2 Source Separation in Vietnam and in the Mekong Delta

In Vietnam the demand for improved sanitation is high: despite government actions only 44 % of the population had sufficient sanitation, i.e. any kind of latrine according to data of the World Bank and Vietnamese surveys (MONRE *et al.* 2003; Soussan *et al.* 2005). The double vault separation toilet had been the key of a Rural Sanitation Programme for disease prevention and increase of food production that began 1956 in North Vietnam (Austin *et al.* 2005).

The gtz sector program "Ecosan" (from "Ecological sanitation") also started activities with the implementation of separation toilets at several places, e.g. Nha Trang (Nghien & Calvert 2000), Nam Dinh (Klingel 2001). Different types of double vault composting toilets have also been tested in Khan Hoa Province (Chien *et al.* 2002). There, wood ash was recommended for smell prevention and for fertiliser improvement. Solar heating proved to be more complicated but more efficient in killing pathogens.

The results of the government initiative some years ago showed a north-south difference (Ni 2005, personal communication). Separation of urine and faeces is practiced on farms in Northern Vietnam, probably introduced/infiltrated by the Chinese emperors (Carlander & Westrell 1999; Humphries *et al.* 1997).

In the Mekong Delta separation systems are uncommon and the most popular system are still pond toilets used by two thirds of the population (Soussan *et al.* 2005). As the government wants to abolish that practice of pond toilets the number of systems is decreasing slowly (Wohlsager *et al.* 2008). However, no change to dry sanitation and composting toilets can be seen in the Mekong Delta. If people can afford changing, they install modern flush toilets (Wieneke 2005).

8.1.3 Research Questions

As excreta contain valuable nutrients, these might be reused to contribute to environmental and personal benefit.

To find out about the nutrient potential of human excrements in the Mekong Delta and about the chances and constraints of a modern source separation system in this area, the follwing studies were carried out to regared the possibilities of source separation for the Mekong Delta more closely:

1. Setting-up a source separation system:
   What are the main parameters of the set-up (size, construction) and what are the constraints in the Mekong Delta?
2. Measurement of nutrient fluxes in the system:
   How much nutrients can be gained per person in a separation system?
3. Treatment of yellow water by open and closed storage:
   Does storage when carried out to reduce potential pathogens in urine lead to nutrient losses?

8.1.4 Collaboration data

The source separation system was developed together with Uwe Klaus, who prepeared the detail planning.

Data on nutrient fluxes in the sanitary system and on yellow water treatment were collected by Susanne Wohlsager who conducted her diploma thesis in Can Tho in 2007-2008 , supported by Ms. Nguyet and Mr. Phong from Can Tho University (Wohlsager *et al.* 2009). Sociological data were evaluated by Christopher Scharfe within his diploma thesis at Bonn University (Scharfe 2009; Wieneke 2005; Wohlsager *et al.* 2008).

8.2. Setting-up a Source Separation System

8.2.1 Planning and Construction of Drain and Treatment

The new source separation system was planned to allow reuse of nutrients that are discharged and wasted elsewhere.

To facilitate extraction of nutrients from waste water and later application in the field, a high nutrient concentration in waste water was intended. High nutrient concentation is reached by separating the nutrient-rich yellow water fraction and by minimising the input of water.

For a successful collection of waste water it was furthermore necessary to find a system that is accepted and well-liked by users.
In a survey carried out in Can Tho, houshold owners had stated that flush systems are their preferred kind of sanitary system (Wieneke 2005). This preference was confirmed by the affiliated Vietnamese scientists: dry composting toilets suffer of low acceptance in a "modern" Mekong Delta. Thus, regardless the advantages of dry sanitation, flushing toilets and waterless urinals were selected (toilets produced by Swedish Gustafsberg Company, see Figure 8-2).

As study site, a students´dormitory was chosen. The "B23" of Can Tho University is a one storey building where 100 male students were living and sharing 10 bathrooms.

The system consists of four different fractions of waste water:

1. Yellow water from waterless urinals and separation toilets with flushing system
2. Brown water from separation toilets as well
3. Grey water from taps and showers in the bathroom
4. Rain water from the dormitory roof

Rainwater pipes had been installed at the dormitory already. Rainwater collection is quite common in rural households in the Delta (in 57% of the households during the rainy season (Wieneke 2005)). This fraction – due to its low nutrient content - was not regarded further in this study.

For fractions 1-3, a new pipe system was installed to take each fraction separately from the bathroom to the collection tanks.

The collection tanks were construced below ground level to allow a free flow of waste water. The top of the tanks was also higher than the regular flooding level. and the tanks were equipped with close-fitting lids.

The size of the collection tanks was determined by the amount of flushing water expected:

- Urine is discharged undiluted from the urinals, just accompanied by cleaning water.
- Separation toilets will allow a flush of around 300 ml through the small front hole each time when flushing (Figure 8-2).
- Flushing water will mainly run through the big outflow in the toilet rear to the brown water pipe. This flushing volume can be adjusted at the toilet tank from 3 to 6 l.

It was estimated that a person will use the toilet four times a day. However, in case of absence, the students will not use their dormitory toilet every time. The expected amount of wastewater is listed in Table 8-3.

Table 8-3: Expected amount of wastewater from the dormitory

Type of wastewater	Expected Amount 100 pers d^{-1}
Yellow water from urinals	100 l
Yellow water from toilet	100 l
Brown water	1200 l
Grey water	11 m^3

To prevent gaseous emissions that are causing nutrient losses and smell, urinals were equipped with smell-stops. These are plastic lips just opening when a water flush is passing (by Keramag company).
As the brown water tank has to be equipped with an exhaust pipe to avoid explosive gas production in the tank, volatilisation can not be avoided completely there.

Figure 8-4 shows the pipe system and the collection of the separarated streams which was installed at the students´dormitory B23.

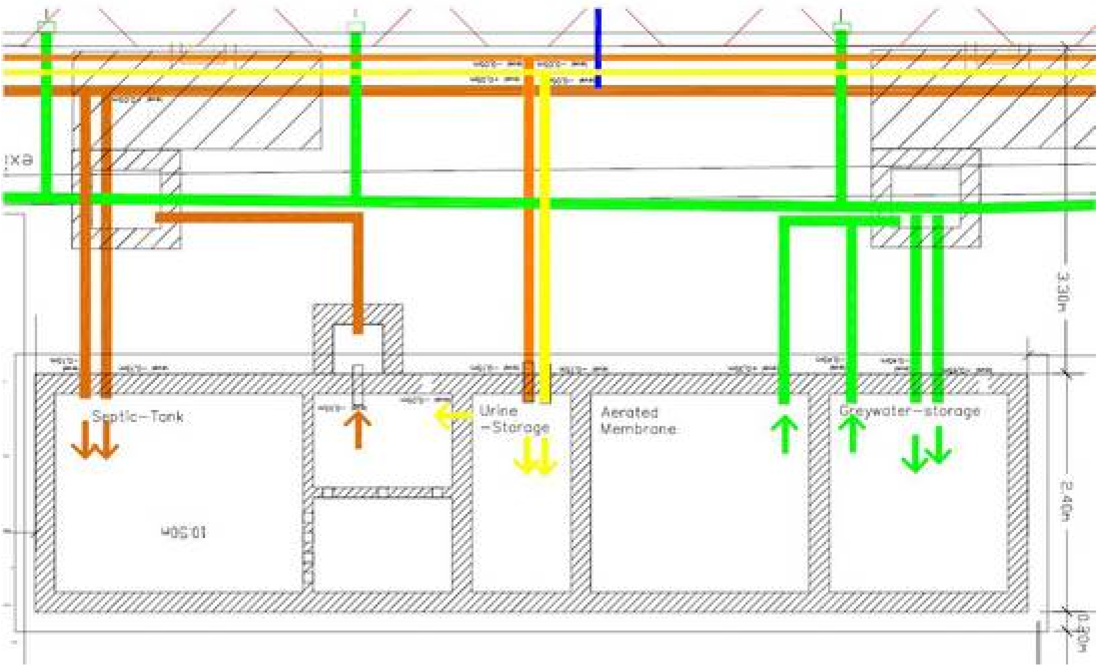

Figure 8-4: Piping system and collection tanks at the dormitory B23 of Can Tho University

From the tanks, material was taken out for analysis and for further experiments, e.g. for the surveys of chapter 8.3 and 8.4.

Further treatment comprised one unit for yellow water (precipitation and stripping), an anaerobic two-step mesophilic reactor for brown water and a membrane treatment for grey water. Details of this DeSaR® system designed by Hans Huber AG are described elsewhere (Antonini *et al.* 2008; Bischof & Paris 2007; Clemens *et al.* 2008; Paris & Netter 2009).

Planning and installation of the new sanitation system had been time consuming (Table 8-4).

Table 8-4: Time schedule of implementing the source separation system

Task	Year
Project proposal after research on demand and status-quo in the MD	2004
Acceptance and confirmation of funding	2005
Detail planning, translation to Vietnamese, adaptation to VN standards	2005
Tendering	2006
Main construction work	2006
First measurements	2007
First results	2007

8.2.2 Results Regarding Construction and Installation

During construction of the separation system several mistakes occurred and had to be corrected:

- Instead of three pipelines as indicated in the plans, the construction company installed just the usual single pipe.
- Plastic pipes were not connected tightly at several urinals. When bigger amounts of liquid were discharged, leakage occurred.
- Smell stops – although indicated on the plans - had been forgotten initially.

During the first periode after installation the following problems had to be solved (Table 8-5).

Table 8-5: Problems and solutions when operating the source separation system

Problem	Solution
Blockage occurred in several yellow water pipes at the ledges and bulges, especially when the toilets or urinals were not used regularly.	Regular cleaning with (acetic) acid and the use of a sewage auger reduced the problem.
Smell lead to reduced use of the urinals.	Improvement was reached by increasing the number of regular checks of the installations (tightness/blockage of pipes, aging of smell stops) and repeated cleaning instructions.
Small screens in the dormitory sink were removed repeatedly by the students to allow free outflow.	Explanation of the system and instructions for the users improved the situation.
Blockage of grey water pipes occurred due to rubbish, e.g. plastic bags, food residues.	Additional openings in the pipe lines of grey and yellow water (T-connection at the turns) proofed to be very effective to remove blockages.
	Installation of a larger sieve at the inflow of the tank made sure that waste was kept out of the storage tank and the treatment system.
A swimming layer formed on top of the stored brown water, despite daily pumping was not enough.	The layer has to be destroyed manually every few months until other possibilities, like a stirrer can be realised.
Metal lids used as the tank cover suffered of corrosion in the tropical wet climate.	Replacement after ~2 years; superior materials would be steel or plastic.

8.2.3 Conclusions on Construction and Implementation

During construction and the starting phase careful supervision on a regular and frequent basis is necessary. New systems need to be explained and need attention as problems may deter (or put off) users. Cooperation of the users, regular cleaning as well as tight collaboration between technicians and users is crucial.

For a successful implementation of a new system, users have to be informed well. This will also increase their acceptance of the new system (Scharfe et al. 2008).

The smell stops from Keramag Company proved efficient to avoid smell. Although they had to be replaced from time to time, this simple system was preferable to other systems e.g. a flap or a separation liquid in the u-bend. If necessary, a flap might be installed additionally at the inflow of the yellow water tank.

Diameters of urine pipes should be big and inclination should be steep enough to avoid blockages. Jönnsson & Vinneras (2007) suggest bigger than ∅ 75 mm and at least 1% inclination. Furthermore a possibility for opening the pipes should be included.

8.3. Nutrient Fluxes in the Source Separation System

As only few data on separation systems outside of Europe are available, mass flows as well as nutrient flows were determined in the newly installed separation system at the students´ dormitory. Furthermore heavy metal and hygienic parameters were determined. For better understanding a sociological survey was conducted.

8.3.1 Material and Methods

The inflowing volume was determined by the increase of the water level which was then multiplied by the ground area of the tank. Data on water consumption were measured by a fixed water meter and recorded by the dormitory manager.

The following analyses were carried out:

- Field parameters were determined directly by electrodes (WTW).
- Main nutrients were determined at CTU laboratory: N (Kjeldahl-N), NH_4 (Kjeldahl and Quantofix method), NO_3, P, K (Hach Lange cuvette test, Photometer).
- Microbiological parameters, *E. coli,* coliforms, coliphages, *Salmonella* sp. and worm eggs were determined in Can Tho (see chapter 4).
- Heavy metals were analysed at Bonn University (AAS).

Urine samples were taken from plastic tanks colleceted from the dormitory for the storage experiment (details see chapter 8.4).
Fresh yellow water was taken at the end of the pipe, at the inflow to the tank by collecting the incoming water for half a day.
Furhtermore yellow water was taken from the storage tank 1 day after emptying the tank, as well as 1 month and 4 months after emptying. Within this period, fresh urine added to the tank continously.

The nutrient amount measured was compared to values based on calculated data on the food supply. The following formulas were used (Jönnson & Vinneras 2003):

N = 0.13 x total food protein,

P = 0.011 x [total food protein + vegetal food protein].

Taking the available nutrition data from the FAO (2007, data for 2002-2004), for Vietnam 66 g $person^{-1}day^{-1}$ grand total protein and 48,8 g $person^{-1}$ day^{-1} vegetal protein have to be assumed.

Using a questionnaire with 44 questions, 50 students in the B23 and the same number of students in the neighbouring dormitory B22 (control group) were interviewed face-to-face by Vietnamese interviewers in November 2007 by Scharfe (2008). The interviews were conducted within a short period of time and in a discreet manner, avoiding other students being able to listen to the interviewers' responses. For this sociological study Icek Ajzen's 'Theory of Planned Behavior' (Ajzen 1991) was taken as a theoretical basis for the empirical survey.

8.3.2 Results and Discussion

8.3.2.1 Discharged Volumes

On average, per day 1 m^3 brown water, 50 l yellow water and 5.5 m^3 grey water were discharged into the tanks (Table 8-6). The highest inflow rates were measured between 6 and 8 a.m. for all fractions. Grey water inflow was high between 12 - 2 p.m. and 6 - 8 p.m., too.

The total amounts were lower than expected for all fractions. Assuming that one student urinates about 3 times per day in the dormitory - leading to approximately 70 l of urine, diluted with about 300 ml water per flush - the measured 50 l inflow is very low.

Table 8-6: Measured amounts of separated wastewater fractions at the dormitory B23

Fraction	Measured Inflow [m^3 per 100 $pers^{-1}$ d^{-1}]
Yellow water	0.05
Brown water	1.0
Grey water	5.5

Compared to the water consumption of 11.5 m^3 (average of three months given by the dormitory manager) the measured total water volumes were small, too.
Supposed reasons for the low inflow were frequent use of other toilets outside the dormitory and water losses in the pipe system. A succeeding inspection could indeed detect leakages at several pipe connections.

8.3.2.2 Nutrient Concentrations

Nutrient concentrations were determined in different types of yellow water.
In fresh urine 6.68 g l^{-1} total nitrogen were determined. At the tank inflow 2.35 g l^{-1} were measured. After storage N_{tot} was around 15% less after one month and 30% less in samples of 4 months storage time (Table 8-7). The lower nitrogen contents in the stored samples were most possibly due to ammonia losses that occur at the prevailing high pH of 8.5 - 9.1 (details see chapter 8.4.2.2).

The main nitrogen fraction was ammonium contributing ~95% to N_{tot}. Nitrate is found only in traces (~0.1% of N_{tot}). The remaining N is urea from fresh urine that has not been transformed to NH_4 yet.

Table 8-7: Nitrogen concentrations in fresh urine and in the collection tank of dormitory B23 after different storage periods

	Unit	Fresh urine	Tank inflow	$Tank_{4\ weeks}$	$Tank_{5\ weeks}$	$Tank_{4\ months}$
N_{total}	mg l^{-1}	6680	2350	2030	1600	1620
NH_4-N (Quantofix)	mg l^{-1}	n.d.	2230	1950	1590	1800
NH_4-N (Kjeldahl)	mg l^{-1}	n.d.	1990	1880	1580	-
NO_3-N	mg l^{-1}	n.d.	2.9	2.5	2.0	1.5

n.d.= not determined

Due to the flush water dilution factor in separation toilets, loads in the yellow water were expected to lower than in urine.

The dilution ratio of fresh urine compared to the diluted urine in the inflow tank was 3:1 for nitrogen in the beginning (day 1). The ratio increased to 4:1 after some time of storage (Table 8-8), most possibly due to gaseous nitrogen losses.
The ratio was lower for phosphorus: 4:1 and 6:1, respectively. As the dilution in the toilet bowl was evidently the same, it is likely that phosphorus losses were occurring in the pipe system.

Table 8-8: Ratios of nutrients in fresh urine compared to separated yellow water in the tank

	N_{tot}	P
Fresh urine to yellow water at inflow	3:1	4:1
Fresh urine to stored yellow water	4:1	6:1

However, concentrations of soluble ions such as potassium (K^+), sulphate (SO_4^{2-}), sodium (Na^+) and chloride (Cl^-) were about 6 times lower in the yellow water (Table 8-9).

Phosphorus concentrations ranged from 135 to 86 mg l^{-1} ortho-phosphate. The lower concentrations in the stored urine might be due to precipitation of phosphates, which formed a layer at the bottom of the tank at B23. This phenomena (spontaneous precipitation) was observed at other places as well, e.g. in the separation system at Bonn University and by other authors (Gethke *et al.* 2007; Udert *et al.* 2003b). It was regarded at more detail in the storage experiment (see chapter 8.4).
Phosphorus losses also took place in the pipe system which corresponds to findings of other authors (Udert *et al.* 2003b).

Table 8-9: Nutrient concentrations in fresh urine and yellow water (s standard deviation +/-)

		Urine	s	Yellow water	s
N_{total}	mg l^{-1}	6890	7.8	2350	351
P_{total}	mg l^{-1}	816	80	210	80
K	mg l^{-1}	1670	189	605	28
Mg	mg l^{-1}	50	7.5	2.6	0.4
Ca	mg l^{-1}	100	18	20	0.8
SO_4	mg l^{-1}	1430	52	580	40
Na	mg l^{-1}	3510	295	1220	73
Cl	mg l^{-1}	5670	441	2120	87

8.3.2.3 Heavy Metal Concentrations

Heavy metals may be harmful to plants and soil microorganisms when applied in high concentrations. The concentrations measured in urine and yellow water, however, were quite low (Table 8-10). Compared to guidelines of the WHO, all elements were below the given thresholds for drinking water (WHO 2004).

Table 8-10: Heavy metal concentrations in fresh urine and yellow water (s = standard deviation +/-)

		Urine	s	Yellow water	s	Drinking water guideline (WHO 2004)
Zn	µg l^{-1}	340	21	61	8.8	3000
Cu	µg l^{-1}	20	3.0	40	5.6	2000
Cr	µg l^{-1}	4.4	1.3	0.3	0.6	50
Ni	µg l^{-1}	< 5.0		10	1.0	20
Pb	µg l^{-1}	< 2.0		3.0	1.2	10
Cd	µg l^{-1}	< 0.5		< 0.5		3

Interestingly, concentration in yellow water was higher than in urine despite the dilution in the system. Additional input of Cu, Ni, and Pb seems to have happened, due to incoming tap water for flushing or within the pipe system itself. As the material of the wastewater pipes is plastic, it seems more probable that these elements have already been in the tap water before, released from the water supply or distribution system.

Copper is used to make pipes, valves and fittings and is present in alloys and coatings. Copper (Cu-sulphate-pentahydrate) is sometimes added to surface water for the control of algae. Levels in running or fully flushed water tend to be low, whereas those in standing or partially flushed water samples are more variable and can be substantially higher (frequently > 1 mg l^{-1}). Its level often increases during distribution, especially in systems with an acid pH or high-carbonate waters with an alkaline pH. However, even where copper tubing is used as a plumbing material, concentrations will usually be below the guideline value (WHO 2004).

Nickel is used mainly in the production of stainless steel and nickel alloys. Water is generally a minor contributor to the total daily oral intake but due to heavy pollution or use of certain types of kettles, of nonresistant material in wells or of contact with nickel- or chromium-plated taps, nickel concentrations may be high (WHO 2004).

Lead is used principally in the production of lead-acid batteries, solder and alloys. Its presence in tap water is primarily from household plumbing systems containing lead in pipes, solder, fittings or the service connections to homes. The amount of lead dissolved from the plumbing system depends on several factors, including pH, temperature, water hardness and standing time of the water, with soft, acidic water being the most plumbosolvent. Concentrations are generally below 5 mg l^{-1}, although much higher concentrations (above 100 mg l^{-1}) have been measured where lead fittings are present (WHO 2004).

The results on heavy metals don't indicate restrictions for futher use of urine and yellow water as a fertiliser in agriculture.

8.3.2.4 Fluxes of Nutrients and Heavy Metals

The fluxes in the sanitary system were obtained by the concentrations of yellow and brown water and the daily wastewater volumes.

The measured nitrogen and phosphorus fluxes were compared to calculated values received by a simple formula (Jönsson & Vinneras 2003) using food nutrition data of Vietnam (FAO 2005). The calculated nutrient amounts were 3.13 kg N and 0.45 kg P $pers^{-1}\ a^{-1}$, of which 2.75 kg N and 0.30 kg P should be found in urine (Table 8-1).

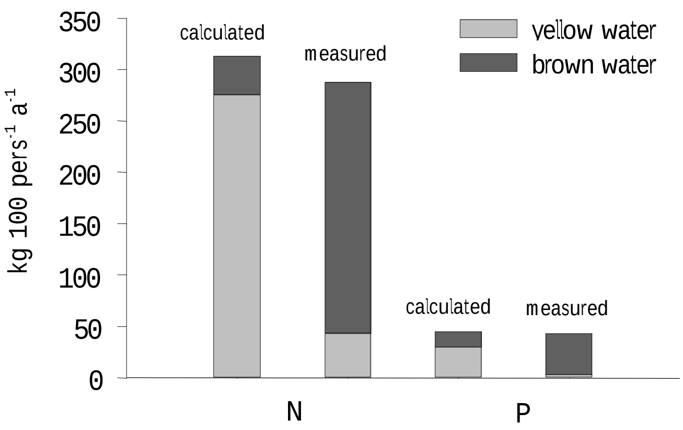

Figure 8-5: N and P fluxes and distribution between yellow and brown water at a 100 students' dormitory ("measured" extrapolated own data, "calculated" according Jönsson & Vinneras 2003)

The total nutrient load of N and P corresponded quite well to the calculation (Figure 8-5). Contrary to the expectations the nutrients were found in brown water. In yellow water the nutrient load was much lower than expected: only 43 kg N per year and 100 persons were collected. This amount equals the excretion of ~15 persons.

The hypotheses that the unusual distribution had been the result of misused toilets was confirmed by the interviews of the students: male students had been usually standing when urinating (see chapter 8.3.2.6). This kind of misuse had been observed in Sweden, too (Vinneras & Jönsson 2002).

Further calculations on nutrient fluxes are based on the results of fresh urine.

The persons in the study excreted an average of 2.5 kg nitrogen, 0.3 kg phosphorus, 0.6 kg potassium, 0.02 kg magnesium, 0.04 kg calcium, 0.5 kg sulphate per year via urine, but also high salt concentrations, 1.3 kg sodium and 2.1 kg chloride.

Heavy metals were excreted in substantially smaller amounts: 124 mg zinc, around 7.2 mg copper and 1.6 mg chromium per person and year. Nickel, lead and cadmium were below the detection limit.
The fluxes measured in Vietnam were compared to results from Thailand and Europe (Table 8-11) to find out about differences between the contries and improve the database for calulating the fluxes word wide.

Table 8-11: Nutrient and heavy metal load excreted per person and year (mean) in Vietnamese urine in comparison to Thai and Swedish reference data

	Urine			Yellow water	
	Vietnam	Thailand[a]	Europe[b]	Vietnam	Sweden
g $pers^{-1}$ a^{-1}					
N_{total}	2520	2360-2740	2500-4000	380	330-480
P_{total}	300	320-430	250-365	38	37-38
K	610	390-770	820-1190	110	160-210
Mg	18	18-38	35-55	0.5	0.3
Ca	37	30-140	59-90	3.6	2.5
SO_4^{2-}	520	320-400	340-680	110	32-41
Na	1280	n.a.	1190-1700	220	177
Cl^-	2070	n.a.	1730-2390	390	1020-1140
mg $pers^{-1}$ a^{-1}					
Zn	124	33-130	16-150	11	13-20
Cu	7.2	0-6	17-1490	7.2	28
Cr	1.6	n.a.	0.2-10	0.1	0.5
Ni	< 1.8	20-42	2.6-30	1.8	2.7-41
Pb	< 0.7	2.6-5	0.7-15	0.5	0.4
Cd	< 0.2	0.2-1	0.08-0.5	< 0.1	0.03

a (Schouw *et al.* 2002): loads in excreta multiplied with denoted percentage that originated from urine

b (Ciba-Geigy 1977; Udert *et al.* 2003a; Vinneras 2002): Loads calculated with 1.25 litres of urine per person

c (Kirchmann & Pettersson 1995)

n.a. not available

8.3.2.5 Hygienic Parameters

The main indicator organisms and selected pathogens were determined in this study. Indicator organisms *Escheria coli* and total coliforms indicate that other – pathogenic - organism may be present as well (Feachem *et al.* 1983).

All analysed micoorganisms had the lowest concentration in the samples from urinal, and higher concentrations in the samples from separation toilets and storage tank (Table 8-12).

Compared to the storage tank, bacterial concentrations in fresh urine were similar (*E. coli*, coliform bacteria, *Salmonella* sp.) or higher (faecal streptococci). *Salmonella* sp. wass found in fresh urine and in the storage tank, but not in yellow water from the separation toilet or the urinals.

Faecal streptococci were comparably high to other faecal indicators (e.g. *E. coli)*. Neither coliphages nor helminth eggs were detected.

Table 8-12: Concentration of *E. coli*, total coliform, faecal streptococci, *Salmonella* sp. in fresh urine, urine from storage tank, separation toilet and urinal

	Unit	Fresh urine	Tank	Separation toilet	Urinal
E. coli	cfu 100 ml^{-1}	< 100,000	21,000	9,300	200
Coliform bacteria	cfu 100 ml^{-1}	230,000	240,000	93,000	200
Faecal streptococci	cfu 100 ml^{-1}	2.1×10^9	300,000	270,000	3,000
Salmonella sp.		positive	positive	negative	negative

In other studies faecal streptococci were also determined in high concentrations in yellow water collection tanks. Höglund (2001) found 16 % of the samples being > 100,000 cfu ml^{-1}, 76 % of the samples > 1000 cfu ml^{-1}. Generally, the concentrations in samples taken from the bottom layer of the tanks were higher than the ones from the supernatant. According to Höglund (2001) re-growth of faecal streptococci in the pipe system can occur.
She suggests determining faecal contamination by analysing coprostanol (5β-cholestan-3β-ol) which is the principal faecal sterol produced by the hydrogenation of cholesterol (cholest-5-en-3β-ol) in the digestive tract.

In another study where four urine tanks had been studied for 6 months, *Salmonella* or *Campylobacter* could not be detected. Enterococci were less than 100 cfu 100 ml^{-1} in total viable counts after 2-4 month of storage (Tarnow *et al.* 2003).

These results – indicator bacteria and pathogens detected in urine and in yellow water – imply cautious use. Spreading of pathogens has to be prevented, and it has to be made sure that neither surface water nor food crops are harmed when applying urine or yellow water. One possible treatment option was examined (see chapter 8.4) and results of hygienic parameters are discussed detailed in part 8.4.2.4.

8.3.2.6 Sociological Findings

The sociological survey, conducted in 2007, revealed several interesting key facts concerning the social background of the students: two third of the 100 interviewed students were of rural descent and 73 % had used predominantly traditional fishpond squat latrines during childhood. Only 8 % grew up with a modern water closet. However, an upward tendency was discovered since the rate of students still using fishpond latrines at home has decreased to 56 % while 21 % have installed modern toilets now. Thus, conventional sitting toilets with automatic flush are becoming more prevalent, a trend matching the chosen solution of a source separating flush system as well as Vietnam's National Rural Clean Water Supply and Sanitation Strategy up to 2020 issued by the government (MARD & MOC 2000).

The unusual distribution of nutrients (high concentration in brown water, low in yellow water) was explained by the survey: a total of 90 % of the students in B23 were standing when using the separation toilet for urination. Accordingly, the urine with its high nutrient concentrations is placed into the rear brown water drain. When using a conventional one-pipe toilet, even 100 % of the students questioned stated to be standing when urinating. The surmise, students would use nature for urinating, and therefore less nutrients were collected in the yellow water tank, was not affirmed. 98 % of the students from B23 use the sanitation facilities in the dormitory when they are present (Wohlsager *et al.* 2008).

However, several reasons were given by the students for not using the urinals in the bathrooms of the dormitory (Figure 8-6).

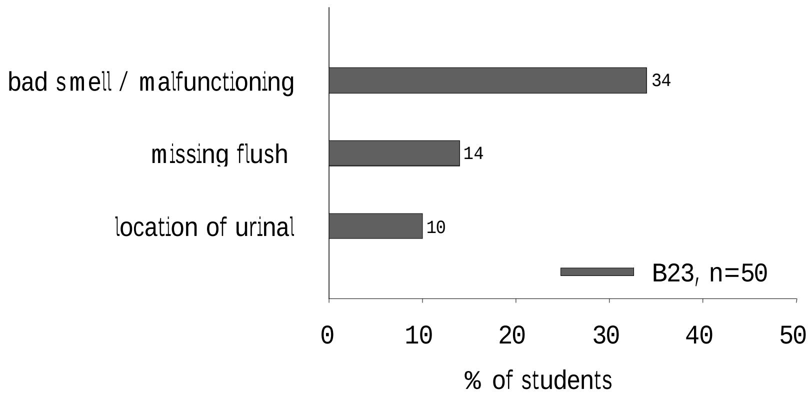

Figure 8-6: Main reasons given by students for not using waterless urinals in the dormitory

After an informative meeting with the students (repeated cleaning instructions, appeal to use the urinals), the urinal use could be enhanced. 46% of the students replied to use the urinal when urinating. Still another 54% continued to use the separation toilets for urinating.

Despite that, overall acceptance of the new systems installed at the dormitory B23 was high. When asked to compare the separation toilets in their room with conventional sitting water closets, 70% of the respondents in B23 assessed the separating toilets as positive, while 26% uttered a neutral and only 4% a negative attitude. Being asked about their preference, if they had the free choice between any toilet modell for their dorm, an impressive 84% of all 50 respondents in B23 would choose a separation toilet as the one installed.

By combining the analytical results with the information provided by the sociological survey, findings could be explained more clearly and suitable improvements could be made. Due to this, the use of urinals could be increased - providing higher nutrient loads in the yellow water fraction in the future.

8.3.3 Summary and Conclusions

The nutrient fluxes measured in the sanitation system were close to the expected amounts but distribution was unexpected due to irregular toilet use.

The main findings at the implemented separation system at the student dormitory were:

- The amount of yellow water collected was below expectations due to the users' habits to avoid urinals and stand when using separation toilets.
- This habit also led to a lower nutrient amount in yellow water than expected.
- The nutrient amount in the brown water fraction was much higher than expected, though total amount of nutrients met the expectations.

Our findings met the conclusion of a study conducted in Sweden (Johansson et al. 2000): "There is a direct correlation between residents´ motivation and the amount of urine collected".

The results on heavy metals don't indicate any restrictions for futher use of urine and yellow water as a fertiliser in agriculture. However, hygienic concerns need to be regarded when using it as a fertiliser.

8.4. Treatment of the Yellow Water Fraction by Storage

Any reuse of secondary raw material needs to be safe, i.e. transmission of pathogens and transfer of undesired substances has to be avoided.

In sanitary waste, brown water is the most critical fraction regarding pathogens. Most bacteria, virus, protozoa, worm eggs, are excreted this way (Table 8-13).

The yellow water fraction is less problematic: in a healthy individual, urine is sterile after kidney filtration. By passing the urinary tract, it will pick up dermal bacteria which are usually harmless (Höglund 2001; Schönning & Stenström 2004). However, infections and even more "cross contamination" with faecal matter may lead to pathogens in urine (Höglund 2001).

Table 8-13: Potential pathogens in human urine and faeces (Höglund 2001)

	Faeces	Urine
Bacteria	*Campylobacter*+, *Salmonella*+, *Yersinia, Shigella, Entamoeba, E. coli (EHEC O157*+*)*	*Salmonella thypi**, *S. parathyphi**, *Leptospira interrogans**, *Mycobacterium tuberculosis**, *M. bovis**
Virus	*Hepatitis, Rota, enteric Adeno*	*Cytomegalo (CMV)**
Worm eggs	*Shistosoma, Helmiths (Ascaris) Hookworms (Ancylostomiasis) Whipworms: Trichuris trichiura*	*Shistosoma haematobium*
Protozoa	*Giardia* sp., *Cryptosporidium parvum, Entamoeba histolytica*	*Microsporidia**

* infection route via urine is not considered to be of relevance

+ infection in human and animals

If the reuse of yellow water is promoted in the Mekong Delta, this puts up the question for the necessary treatment. Depending on the pollutants, the intended reuse and the capacities (e.g. finances, space, knowledge) a suitable treatment has to be chosen.

The main factors influencing the survival times of microorganisms in the environment are temperature, pH, ammonia, solar radiation, moisture, presence of other microorganisms, ionic strength (or salinity), oxygen, nutrients (Feachem et al. 1983; Höglund 2001).
In general, the surival time of different microorganism is longest for helminth eggs and protozoa. Decreasing survivial time is expected for virus and bacteriophages, and even lower for bacteria (usually gram-positive > gram-negative).

Survival times in faeces, compost and other media have been studied (Feachem et al. 1983; Haug 1980), but there are few results available on urine (Höglund 2001; Niederste-Hollenberg 2003).

According to these results, the persistence of microorganisms in urine is influenced mainly by the pH value, concentration of ammonia, temperature and dilution (Schönning & Stenström 2004). Storage of urine could be carried out successfully under European conditions (Höglund 2001; Schönning & Stenström 2004).

Storage is a simple treatment which makes use of the fact that pathogens die-off with time outside their natural habitat. However, gaseous losses during storage may diminish the nitrogen content and precipitation reduces the phosphorus concentration. Little has been studied about the nutrient changes during storage (Udert *et al.* 2003b), and data for tropical countries were not found at all.

The following experiment was conducted to find out about the changes of nutrient composition during open and closed storage.

8.4.1 Material and Methods

The experimental set-up of the batch experiment comprised four treatments with four replications each:

1. Open storage of fresh undiluted urine (OS)
2. Closed storage of fresh undiluted urine (CS)
3. Undisturbed closed storage of fresh undiluted urine (uCS)
4. Closed storage of fresh diluted urine (CS_d)

Urine of male students was collected during 1 day in plastic canisters (30 l) serving as urinals and collection tanks at the same time. The content of the canisters was distributed to the experimental 5 l plastic bottles of treatment 1 to 3.
For treatment 4, yellow water was taken at the inflow of the storage tank containing effluent of the separating toilets and urinals at the dormitory.

The bottles were stored in a sheltered place next to the laboratory at ambient temperatures. Samples of treatment 1, 2, 4 were taken at regular intervals during 8 weeks after shaking. Bottles of treatment 3 were opened the first time after 8 weeks. Further samples of all treatments were taken after 3, 6 and 9 months.

Field parameters, NH_4, N_{tot}, NO_3, P and microbiological parameters were determined at CTU laboratory. K, P, heavy metals were measured at IPE Bonn (see part 8.3.1). Mineral analysis was done by x-ray diffractometry at the Mineralogical Institute of Bonn University.

8.4.2 Results and Discussion

8.4.2.1 Observations and Field Parameters

During the first day of storage white sediment precipitated in undiluted urine, in all open and closed stored bottles. In diluted urine no sediment was formed. The sediment did not change further during the experimental time. In the open stored bottles, an orange-red skin formed on the surface.

The first pH analysis resulted in pH 8.2 (Figure 8-7). For closed storage, the pH stayed at pH 8.9 - 9 from day 2 on. Open storage showed significantly lower values, around 8.9 until week 3, then dropped slightly to 8.8 (week 7) and to 8.6 (week 8).

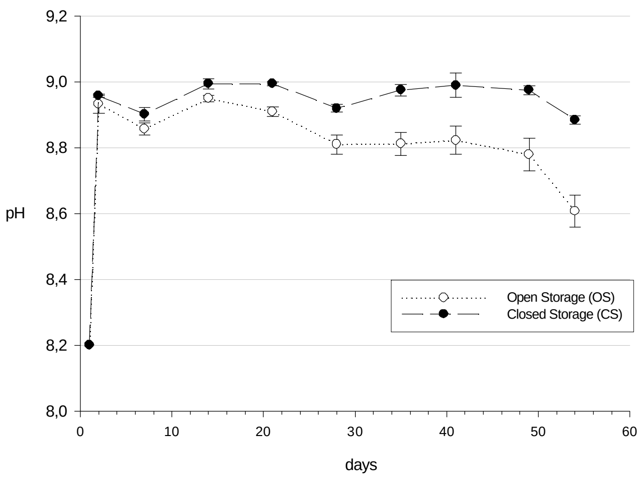

Figure 8-7: pH values during open and closed storage

Diluted urine showed similar pH values from 8.6 to 9 (data not shown).

Oxygen concentration dropped within one day from 0.67 mg l^{-1} (day 1) to 0.05 mg l^{-1} (day 2) in open and closed bottles. It stayed at that low level during the whole experiment.

Electric conductivity (EC) increased during the first three weeks of storage from 23.5 µS cm^{-1} up to 35.6 µS cm^{-1} and 38.9 µS cm^{-1} in OS and CS, respectively. From week 4 on, values were reduced in open storage (OS) to 32 mS cm^{-1} finally. Closed storage in contrary, led to increased values of (41 mS cm^{-1}). Undisturbed closed storage also increased conductivity to 40 mS cm^{-2} in the end.
The EC development is closely related to the N-changes (transformation from urea to ammonium, emission of ammonia) and is explained in the next paragraph more detailed.

8.4.2.2 Nutrients

Nitrogen

Initial nitrogen concentration of undiluted urine was 6.89 g N l^{-1} (Figure 8-8).
In closed storage, N_{tot} remained close to this level within the whole experimental period. After 8 weeks 96% and after 9 months 93% of N_{tot} could still be recovered. Undisturbed bottles even contained 97.3%.
By contrast, open storage led to a loss of 53% within the first 8 weeks. After 9 months only 10% of N was left in the OS-samples.

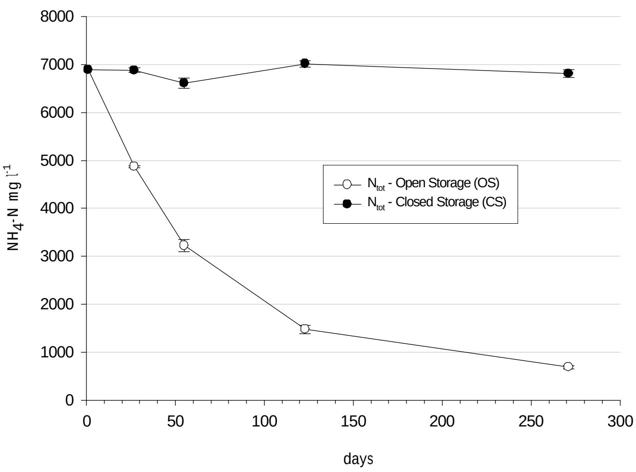

Figure 8-8: N_{tot} concentrations in urine in open and closed storage during 9 months

Within short time most of the nitrogen was transformed to ammonium: in all bottles, NH_4^+ increased during the first three weeks (Figure 8-9).

After three weeks, ammonium values dropped in open stored samples from 4.4 g N l^{-1} to 3.2 g N l^{-1} (week 8) and 0.5 g N l^{-1} finally (month 9).
In closed storage, NH_4^+ concentrations increased further until week 6 to 6.7 g N l^{-1}. At that time, all nitrogen was present as ammonium. During the next weeks, NH_4^+ decreased slightly to 6.3 g N l^{-1} (month 9), with NH_4^+ concentrations being slightly below N_{tot} concentration.

The initially present urea is transformed by urease to ammonium and carbon dioxide:

$$CO(NH_2)_2 + 3\ H_2O \rightarrow 2\ NH_4^+ + OH^- + HCO_3^-$$

The reaction is enhanced by increasing temperature, up to the optimum at 50°C. At higher temperatures, the enzymes will denaturise.

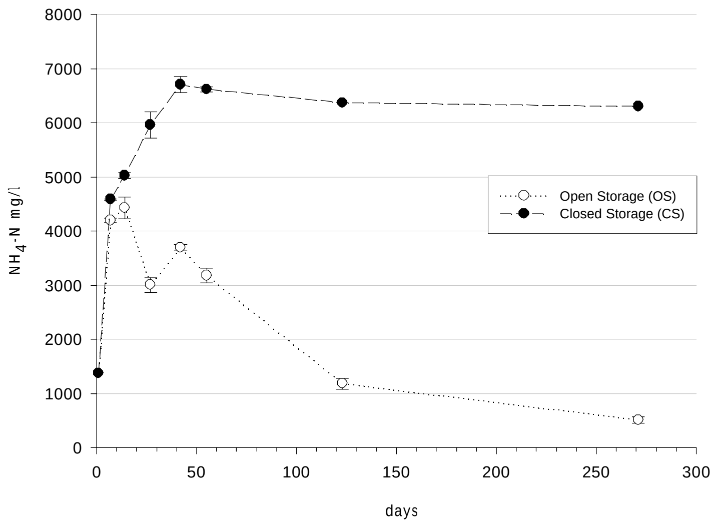

Figure 8-9: NH_4^+ concentrations in urine in open and closed storage during 9 months

The nitrogen losses are mainly due to emission of volatile NH_3. The pK_a, the pH where the ammonia and ammonium concentration are equal depends on the temperature: the higher the temperature, the lower the pK_a and therefore more NH_3 is present. At the prevailing temperature of 30°C the pK_a is at pH 9.031. When volatile NH_3 can expand, it is lost to the air and further NH_4^+ is formed to maintain the equilibrium. Therefore open bottles continuously loose nitrogen.

Besides gaseous losses, part of the nitrogen was precipitated to the sediment. The nitrogen concentration was higher in the precipitate compared to the the supernatant (10.3 g N l^{-1} vs. 6.6 g N l^{-1}).

This reaction can be observed by the increase of electric conductivity which is proportional to the concentration of the formed ions. The increase and decrease of NH_4^+ is also followed by pH changes (Figure 8-10), an effect that has been described by other authors , too (Udert *et al.* 2003b).

The experiment showed clearly that open storage was lead to high nitrogen losses. These losses can be prevented - nearly entirely - simply by closed storage.

N concentration in urine and yellow water is similar to the concentration of cattle slurry but lower than in mineral fertilisers. The mainly used commercial urea contains 46% N, the common DAP 16% N. However, N concentrations of urine may be increased by further treatment (Antonini *et al.* 2008; Herbst *et al.* 2006; Maurer *et al.* 2003).

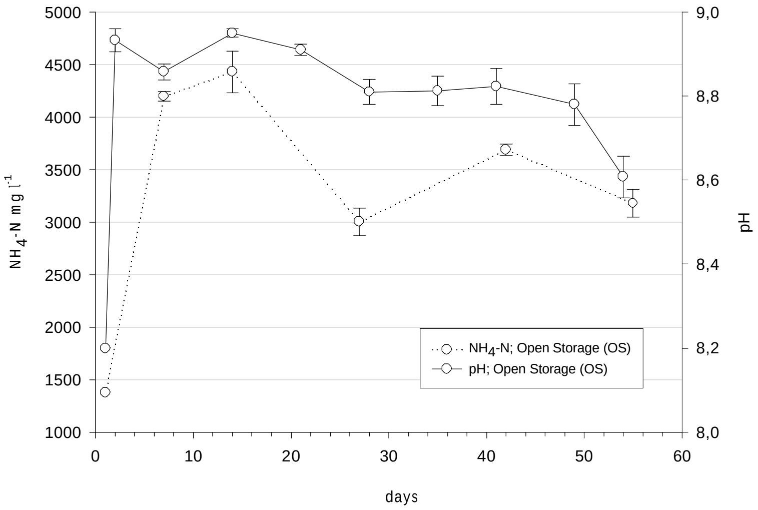

Figure 8-10: NH_4^+ and pH in urine in open storage during 2 months

Phosphorus

The initial concentration measured in students´ urine was 0.54 g P l^{-1} ortho-phosphate. In week 8, the P-concentration in the sediment was 26 times higher than in the liquid supernatant; concentrations in the separately measured fractions were 10.8 g P l^{-1} and 0.42 g P l^{-1}, respectively.

As the volume of the precipitate was small - dry matter content of undiluted urine was ~ 5% - the liquid contained 1.85 g and the precipitate 0.59 g P. However, one quarter, 24% of the phosphorus was recovered in only 1.2% of the volume (Figure 8-11).

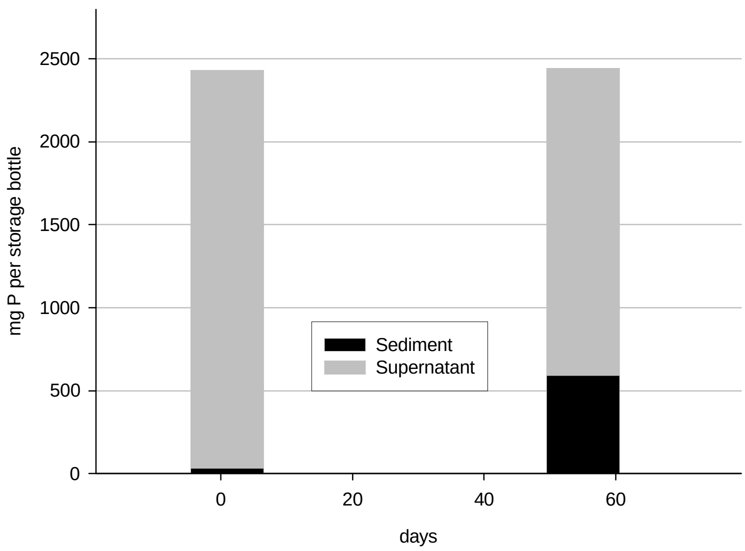

Figure 8-11: Distribution of P between sediment and supernatant in 5 liter storage bottles at day 1 and 55 (mean of four replicates)

Higher P concentration in the bottom part of the tank has been reported before (Johansson *et al.* 2000). This finding is important when the material is intended for use as a fertiliser: as the small amount of bottom sludge contains a considerable amount of total phosphorus, this fraction has to be considered when emptying the tank, and recollected e.g. by stirring. On the other hand, separate use of supernatant or precipitate may be wished depending on the situation.

Compared to other fertilisers, P concentration in urine was about two times higher than in liquid pig manure which is usually around 170 mg P l^{-1} (KTBL 2005). In the precipitate phosphorus concentration reached 1% P which is close to the P concentration of cattle slurry but still quite low compared to mineral fertiliser consisting of 18% P (Triple-phosphate) or even 23% P (Ammon-phosphate).

Higher P concentrations may be reached by further treatment: precipitation can be enhanced by addition of salts, e.g. MgO. Another possibility to increase P concentration is the evaporation of water (Antonini *et al.* 2008; Herbst *et al.* 2006; Maurer *et al.* 2003).

The minerals in the precipitate were determined as calcium phosphate (74%) and sodium chloride (26%) in the x-ray diffractometry.

8.4.2.3 Anorganic Hazardous Substances

Heavy metal concentrations in wastewater and especially in sewage sludge may be too high to allow further safe reuse. Controversial discussions on tolerable limits have been carried on by scientists and ministries.

However, in human urine heavy metal concentrations are usually very low (Jönsson & Vinneras 2004; Roche 1984).
In animal urine or animal slurry concentrations strongly depend on the living conditions, e.g. kind of fodder, kind of metal used in the stable. For example copper may reach quite remarkable amounts in swine manure (Kühnen 2004).

8.4.2.4 Hygienic Parameters

Indicator organims, *E. coli*, total coliforms as well as *Salmonella* sp. and fecal streptococci were measured in the diffent variants: open storage, closed storage (shaken at sampling dates and undisturbed) and diluted yellow water.

The highest *E. coli* concentration in fresh urine samples was 10^4 cfu ml^{-1}. After four weeks of storage at 30°C no more *E. coli* could be detected.

The initial concentration of total coliforms in the urine was higher than for *E. coli,* but after four weeks as well, no more coliforms could be detected. The diluted urine examined after five weeks did not contain coliforms, either.

In the undiluted and diluted urine *Salmonella* sp. were present. After a storage period of four (undiluted) and five weeks (diluted samples), respectively, *Salmonella* sp. were inactivated in all samples analysed. No influence by the type of storage (open, closed, undisturbed, diluted) was observed.

Table 8-14: Concentration of *E. coli*, total coliforms, and *Salmonella* sp. in diluted and undiluted urine over a storage period of eight weeks

Duration	OS – open storage	CS – closed	CS_u – undisturbed	CS_d – diluted
E. coli (cfu 100 ml^{-1})				
Fresh	<0,1x10^5	<0,1x10^5	<0,1x10^5	2,1x10^4
4(5)[1] weeks	0	0	n.d.	0
8 weeks	0	0	0	n.d.
Total coliforms (cfu 100 ml^{-1})				
Fresh	2,3x10^5	2,3x10^5	2,3x10^5	2,4x10^5
4 weeks	0	0	n.d.	0
8 weeks	0	0	0	n.d.
Salmonella sp. (cfu 100 ml^{-1})				
Fresh	positive	positive	positive	positive
4 weeks	negative	negative	n.d.	negative
8 weeks	negative	negative	negative	n.d.
Faecal streptococci (cfu 100 ml^{-1})				
Fresh	2,1x10^9	2,1x10^9	2,1x10^9	3,0x10^5
4 weeks	1,7x10^4	1,6x10^4	n.d.	0
8 weeks	3,8x10^3	1,7x10^3	4,0x10^3	n.d.

[1] undiluted OS and CS were measured after 4 weeks, diluted CS_d after 5 weeks
n.d. not determined

Similar results were reported by (Höglund 2001). She found a fast die-off of *E. coli, Salmonella typhimurium* and *S. senftenberg* in undiluted urine (T_{90} < 5 days). In diluted urine (1:9) *E. coli* had an around fivefold survival time than in undiluted urine, its T_{90} beeing < 20 days.
Rechenburg *et al.* (2008) reported about storage of sterile urine spiked with *E. coli* (10^5 cfu ml^{-1}) in comparison to untreated urine (>10^5 cfu ml^{-1}) at 22°C in Germany: *E. coli* showed a slight reduction in the untreated urine within six weeks. In the sterile urine, however, *E. coli* concentration decreased rapidly during the first two days, then a regrowth exceeding the original concentration was observed. After one week the final concentration reached 10^7cfu ml^{-1}, which slightly decreased to 10^5 cfu ml^{-1} within the next five weeks.

The reasons for the different results can not be stated clearly. Relevant parameters e.g. pH and NH^{4+} concentrations were quite similar in the experiments.
Different temperatures (30°C in Vietnam and 22°C in Germany) may have influenced the degradation and the regrowth, respectively. As UV light is leading to photochemical reactions, the place of storage (outside in the shade in Vietnam vs. inside the laboratory in Germany) may be of interest, too.
Another reason for regrowth in the lab experiment with spiked organisms may be that sterile media may allow longer survival times due to the lack of antagonists (Feachem *et al.* 1983).

However, among total coliforms there are species of non-faecal origin and especially in hot climates they can multiply (Feachem *et al.* 1983). Total coliforms usually consist of 90% *E. coli*, the remainder being species of *Citrobacter*, *Enterobacter*, and *Klebsiella*.

In comparison to the gram-negative bacteria discussed above, faecal streptococci were found in the examined urine in substantially higher concentrations. During the first four weeks of storage the concentrations decreased by 5 log_{10} units from 10^9 cfu 100 ml^{-1} to 1.7 x 10^4 cfu 100 ml^{-1}. After eight weeks, the concentrations had dropped for another log_{10} unit in all the different storage situations. In the diluted urine from the separation toilets containing 3.0 x 10^5 cfu 100 ml^{-1} in the beginning, faecal streptococci were completely undetectable after five weeks of storage. Höglund (2001) also reported higher persistence for gram-positive faecal streptococci compared to gram-negative bacteria. She determined T_{90} values with 30 days at 4°C and 5 days at 20°C, in diluted urine T_{90} was around three times higher.

Worm eggs belong to the persistent pathogens with longest survival times of several months Feachem *et al.* (1983). Worm eggs could not be detected in urine samples.

Survival times of microorganisms in different medias have been summarised by (Feachem *et al.* 1983). Whereas the differences between the media (sludge, fresh water, soil) were marginal for virus, protozoa and helminth eggs, survival of several bacteria was longer in faeces compared to sewage sludge and soil. On crops, survival times were found to be shorter than in other environments.
Longest survival time of several months was found for helminth eggs (Feachem *et al.* 1983). This time period also is expected for survival of cyst forming protozoa (*Giardia* sp. or *Cryptosporidum parvum)*: in a yellow water storage tank a low number of *Cryptospridium* oocysts – probably the most persistent protozoa in water – could be found even after 6 month of storage (Tarnow *et al.* 2003). Höglund (2001) found cysts of *Cryposporidium parvum* to be inactivated within 63 days below the detection limit, its T_{90} value was determined similar to *Clostridium perfringens* with 29 days at 4°C, and 5 days at 20°C.

Although the published studies vary in the determined survival times, the data indicate that sufficient storage time leads to death of excreted pathogens.

Nevertheless, the input of faeces – as the main contamination path - should be avoided as far as possible. Then a 6 months storage period as required e.g. in the Swedish regulation and suggested by different authors (Winker *et al.* 2009) is regarded as sufficient for the vast majority of situations. Urine from toilets that are used by sick persons bears a higher risk (hospital, epidemic area, etc.) and should therefore be handled differently. Depending on the situation, the recommendations would reach from longer storage time, other treatment, no reuse to food crops, up to no reuse at all.

8.4.2.5 Organic Pollutants

Organic pollutants are a big group of chemicals. There are substances used in human and veterinary medicine (e.g. antibiotics, pharmaceuticals) others used in agriculture or industry (e.g. pesticides). Due to anthropogenic discharge, these chemicals are occurring in the environment and may show negative effects when incorporated in high doses by humans, water organisms, or plants.
Within this study these chemicals could not be examined, so results of other studies are discussed shortly.

Antibiotics and pharmaceutical residues can be found in both, urine and faeces. Depending on the substance it may be mainly in one or in both fractions (Larsen & Lienert 2007). The authors found nearly two thirds of the examined "active ingredients" to be excreted in urine. Some categories of pharmaceuticals e.g. x-ray contrast agents are excreted nearly exclusively in urine (90-100%), as well as some cancer medicaments (98%). Other cancer medicaments however, only appeared in urine with 6% (Lienert *et al.* 2007). In general, lipophilic substances appear in faeces and hydrophilic in urine (Larsen & Lienert 2007). According to Winker (2009) the major degree of pharmaceuticals and their metabolites in wastewater originated from urine.

Some substances may be metabolised to a high degree, e.g. Sulfamethoxazol where 85% metabolites and only 15% of the initial substance is found in the excreta (Schneider 2005). Lienert *et al.* (2007) found out that about half of the "active ingredients" had been metabolised when found in urine whereas in faeces substances were mainly unchanged.

Degradation during storage of urine has rarely been investigated. In studies on slurry, degradation of pharmaceuticals during storage was found (Engels 2004; Tolls 2001). The results of several studies also showed that some components, e.g. Tetracylines, Sulfonamides may be rather stable (Schneider 2005).

Despite a number of studies, the environmental and eco-toxicological effects are mainly unknown (Pronk *et al.* 2007). In water and in soil degradation is caused by chemical, photochemical and - most importantly - by microbial reactions, mainly depending on the physico-chemical composition of the substance (Scheffer 2002). Depending on the type of chemical, this may take some time. When adsorbed to soil particles, plants can take up chemicals (Schneider 2005; Winker 2009).

So far, risks can not be completely excluded when using urine as a fertiliser (Schneider 2005; Winker *et al.* 2009). Nevertheless, compared to pharmaceutical concentrations in animal slurry, concentrations in human urine are usually lower (Hammer & Clemens 2007).

8.4.3 Summary and Conclusions Urine Treatment

The two main results obtained in this experiment were:

1. Nitrogen losses during storage of urine and yellow water can be minimised effectively just by closed tanks. The experiment demonstrated a great difference between open and closed storage with nitrogen losses of 90% and 7%, respectively, within in 9 months.
2. Storage can be recommended as a simple and cheap treatment to improve hygienic safety: Pathogens in urine and yellow water were reduced considerably after 4 weeks. Examined pathogens and indicator bacteria were below the recommended limits for reuse in agriculture and aquaculture proposed by FAO (Ongley 1996) and WHO (1989). The experiment indicates a faster degradation in tropical regions than under European conditions. However, until further proof a storage time of 6 months is recommended.

8.5. Conclusions and Outlook for Separation and Reuse

8.5.1 Source Separation Systems

In the Mekong Delta ~80% (Soussan *et al.* 2005), in Vietnam around 40% of the population is without proper sanitation (MONRE *et al.* 2003). Worldwide also around 40% of the population is living without toilets (Winblad & Simpson-Hébert 2004). Bad sanitation is one important factor causing diseases (Feachem *et al.* 1983).

Urine separation can contribute to solving the problem not only in Vietnam but also at other places where sanitation systems have to be introduced. Besides improving sanitation, nutrients and water can be recycled contributing to environmental protection.
Different solutions will have to be applied to homestays where people will handle their own excreta and urban situations where a communal or even a private service will take care (Winblad & Simpson-Hébert 2004). However, for successful implementation it is crucial to be accepted by the users (Johansson *et al.* 2000).
For those countries or situations where it is accepted, a dry sanitation system (e.g. composting toilets) has a lot of advantages. To introduce source separation systems to people feeling comfortable with wet systems, these have to be developed further.

Many authors also suggest urine separation as a technology for modern sanitation and environmental engineering in western situations (Larsen & Gujer 1997; Larsen *et al.* 2007; Otterpohl *et al.* 2003; Otterpohl *et al.* 1997; Udert *et al.* 2006; Wilderer & Paris 2001). Urine contributes less than 1% but is responsible for the extensive nutrient elimination steps at wastewater treatment sites. Without urine, treatment could be much smaller (Udert 2006). At the moment authors are elaborating architectural aspects (Kristinsson & Luising 2001; Schiere & Longstrup 2001), desings for new constructions (Wang & Bao 2007) or retrofitting of existing housing (Meinzinger *et al.* 2008). Furthermore, they are about to find out which solutions provide economical benefits at which situations (Oldenburg *et al.* 2007).

8.5.2 Treatment of Separated Urine

Besides storage that has been proofed successfully, other treatment alternatives have been tested in Can Tho. Depending on the situation and the intended purpose, different systems may be favourable.

Urine drying may be a favourable option for sites where urine can not be used close to the production site. This treatment will produce a fertiliser that can be transported easily and could be sold on the market. By drying the osmotic potential of microorganisms is decreased and pathogens will presumably die-off quite fast. Evaporation is energy consuming (Maurer *et al.* 2003) but by using solar energy this can be faced (Phong *et al.* 2008).

Precipitation may be another option for gaining a transportable fertiliser. Precipitation happens spontaneously in undiluted urine as described in chapter 8.4.2. It may be induced by addition of chemicals, e.g. CaO or others (Gethke *et al.* 2007; Udert *et al.* 2003b) and provide a P-rich powder that can be used and sold like a mineral fertiliser.
If precipitation is carried out, remaining N should be removed in a further treatment step, e.g. by stripping with a strong acid. H_2SO_4 has been applied already (Bischof & Paris 2007; Gethke *et al.* 2007). Other acids may be favourable for specific fertiliser formulas. Using H_3PO_4 would increase the phosphorus content which might be favourable at sites with low P supply like in the Mekong Delta.

Further treatment alternatives have been evaluated in Europe (Gethke *et al.* 2007; Larsen *et al.* 2007; Maurer *et al.* 2006), e.g. reverse osmosis 5-fold concentration (Mauer 2003), zeolithe loading (Simons 2008) and freezing processes (Herbst 2006). However, compared to simple storage, these options for yellow water treatment need more investment or energy input, or can not recycle sufficient nutrients.

8.5.3 Use of Urine and its Derviates in Agriculture

The potential of urine as a fertiliser has been shown already by some authors who applied urine and urine derivates successfully in pot and field experiments in Europe, Africa and Asia:

- Simons (2008) and Simons & Clemens (2003) compared the effects of urine, cattle slurry and urine/cattle slurry mix (NH_4^+-ratio 1:1) on grassland in Bonn, Germany. Yields did not show significant differences, although plant N uptake was higher when fertilised with urine than with slurry.
  Application of urine and a urine/slurry mix to winter barley (*Secare cerale*) resulted in higher yields than application of Calcium-ammonium-nitrate (Simons & Clemens 2003) possibly because of the additional supply of other nutrients with urine.

- Muskolus (2007) and Muskolus & Elmer (2007) tested the fertilising efficiency of urine compared to Calcium-ammonium-nitrate (CAN), in Brandenburg, Germany for two years, in 2004 and 2005. At the selected application rates of 50-100-150 kg N $ha^{-1}a^{-1}$, yields of rape (*Brassica napus*), winter rye (*Secale cereale*) and spring wheat (*Triticum aestivum*) did not differ significantly between the two fertilisers.
- Germer & Sauerborn (2005) studied urine fertiliser in an African tropical region (Ghana). Their results indicated a better efficiency of urine compared to other fertilisers which might have been due to the higher water content.
- Ni (2007) and Ni *et al.* (2009) tested successfully the application of urine at Hoa An in the Mekong Delta to tomato.

The results of the different authors indicate that urine can be applied very successfully for growing crops with effects comparable to mineral (Germer & Sauerborn 2005; Muskolus 2007) and organic fertilisers (Simons & Clemens 2003).

The nutrient concentration in urine is rather low, in diluted yellow water even lower. So transportation of untreated or stored urine will only be feasible for short distances. However, the use of yellow water would combine nutrient application with irrigation which is both necessary at many places in the Mekong Delta. Therefore it is necessary to look for suitable locally available application systems. Feasable European systems e.g. slurry tanks don't exist.

Higher concentrated urine-based fertilisers will be superior regarding transportation. The fertilising effect, however, depends on the chemical form of the derivate: Simons (2008) who compared several urine derivates and MAP from wastewater treatment plants found plant yields similar to mineral fertilisers for urine and Ca, Mg-phosphates. Fe- and Al-precipitates ranged close to the non-fertiliser variant (Simons & Clemens 2003).

Own recent pot experiments (Arnold *et al.* 2009) showed that MAP and natural precipitate of yellow water was superior to evaporated urine. If urine had been acidified before evaporation, yields and nutrient uptake were similar to MAP.

Depending on the local conditions (e.g. weather, soil), an advantage of urine or urine-based fertilisers may be expected.
At dry conditions, liquid urine will bring an additional "irrigation effect" and might therefore improve plant growth. At sites where salinity may arise, urine and even more evaporated urine should be handled with care.
Attention should also be put on the pH in low buffered systems where the pH effect of fertilisers is relevant. For NH_4-containing N-fertilisers (urea, ammonsulfate $(NH_4)_2SO_4$) - a decrease in soil pH is to be expected (Finck 1991b).

These effects of urine and urine-based fertilisers on soil and crops have to be studied in more detail.

Application Techniques and Risks

When applied to the field, a proper application technique has to be chosen to avoid gaseous losses. Similar to slurry application, especially NH_3 emissions may occur. Due to the high NH_4^+ concentrations and high pH, application techniques for minimising the volatilisation should be used (e.g. injection techniques) at suitable weather conditions.
These devices are not available in the Mekong Delta. There, pumping or application manually might be used. Drip irrigation would reduce the work load but might need solutions for avoiding formation of precipitates causing blockages.

Although the hygienic risk is estimated as generally low (Höglund 2001), urine should be handled with care (see 8.4.2.4). In case of known contamination with faces or excrements of sick persons, further precautions should be taken. Depending on the situation, longer storage time or no use for food crops might be advisable for safety reasons.

Root vegetables are more prone to contamination than others (Feachem *et al.* 1983). Vegetable eaten raw is very sensitive, too. So, the last application of urine should be sufficient time (e.g. 4 weeks) before harvesting.

Acceptance of Farmers and Consumers

Rather high acceptance of urine as a fertiliser was found in in the Mekong Delta (Scharfe 2008), confirming studies in Sweden (Schmidtbauer 1996), Switzerland (Lienert 2003), Germany (Muskolus 2008).

In the Mekong Delta farmers had no or very little restrictions against using urine as a fertiliser (Scharfe 2008). They would eat products where urine or urine products had been applied as fertiliser.

Brandenburg´s farmers were mainly concerned about legal liability, price and customer acceptance. The potential hazards resulting from micro pollutants or hygiene were ranked lower but still more important than logistic or ecological issues (Muskolus & Ellmer 2007). Only 18% would generally not apply urine on their fields; 25% would like to do so. A large percentage (53%) was not sure yet, most possibly because of lacking information (Muskolus & Ellmer 2007). Interviewed consumers in Berlin had little restrictions against vegetable grown with urine as a fertiliser – if safety was assured. 76% "would" and an additonal 16% "would maybe" accept food produced with urine as a fertiliser (Muskolus & Elmer 2007).

In Switzerland 57% of the farmers liked the idea of using urine fertiliser, 42% agreed to buy such a fertiliser if it is hazard free and not more expensive than mineral fertiliser (Lienert *et al.* 2003). Most interviewed citizens would accept urine-fertilised vegetable (72%). 80% even favour urine above artificial fertiliser (Pahl-Wostl *et al.* 2003).

9. Evaluation of Suitable Treatment Systems

9.1. Introduction

Various techniques are available to treat wastewater: simple and high-tech, small and large-scale, anaerobic and aerobic treatment etc. (Bank 1993; Gujer 2002).
Due to declining resources and the demand for sustainability, nutrient - especially phosphorus - recycling gets more important.

The development of "new sanitation systems" started several years ago (Johansson *et al.* 2000; Jönsson *et al.* 1997; Larsen & Gujer 1997; Lens *et al.* 2001; Otterpohl *et al.* 1997; Sasse 1998; Wilderer & Paris 2001) and is progressing (Berger & Lorenz-Ladener 2008; Londong 2006; Maurer *et al.* 2006; Oldenburg *et al.* 2007; Peter-Fröhlich *et al.* 2007; Winker *et al.* 2009). Improved systems do not only treat wastewater at the end of the pipe but aim to avoid, reduce and recycle water and its nutrient contents.

Although quite a number of different systems have been established worldwide, profound results are still limited. Especially for developing countries – where new systems are expected to be built - data on applicability and nutrient fluxes are scarce.

Especially data on plant availability are lacking although this is crucial for efficient reuse. Simons (2008) showed that there may be huge differences depending on the chemical form of the substrates, e.g. Al- and Fe- compared to Ca- and Mg-phosphates. Own experiments demonstated differences in nutrient uptake and biomass development of several urine based fertilisers (Arnold *et al.* 2009).

Thus, besides the classical wastewater parameters and the nutrient transfer rates, plant acceptance and site specific parameters have to be studied.

This is especially important and sensible in a region like the Mekong Delta where economic resources are small and soils and crops are suffering, e.g. from phosphorus deficiency or low carbon content.
In the Mekong Delta where no wastewater treatment is established yet, there is the chance to establish new systems that focus towards reuse instead of just removal, as practiced in many developed countries. For safety reasons, reduction of pathogens and toxic substances has to be monitored critically and enhanced where necessary.

In the previous chapters the performance and limitations of several suitable systems have been discussed for the Mekong Delta. In Table 9-1 these are listed, similar to the general reuse options of classical systems shown in chapter 3.

Table 9-1: Sustainable removal options of desired substances for wastewater from households and farms

Desired Substance	Sustainable removal options	Further use of recyclable substance	Limitations
Suspended solids, Organic material	Anaerobic transformation of organic matter into biogas	Production of a soil conditioner by composting or vermicomposting of sludge	Waste material, e.g. plastic needs to be handled differently Rarely restricted (e.g. heavy metal content)
Nitrogen	Separation of yellow water	Use as fertiliser after treatment, e.g. storage, stripping, concentration	Different toilet and piping necessary
	Reuse in a wastewater pond		Space for pond needed
Phosphorus	Separation of yellow water Precipitation with Ca, Mg	Use as fertiliser after treatment, e.g. precipitation, concentration	Cost for Ca, Mg, but elements useful plant nutrients

In the following, evaluation criteria and a selection tool are presented for wastewater treatment systems and produced fertilisers in the Mekong Delta.

9.2. Proceeding

To evaluate and select suitable systems for the Mekong Delta, the following steps were performed:

- Definition of system borders and criteria for evaluation (treatment and reuse)
- Selection of systems to be compared
- Evaluating the selected systems: experiments, expert interviews, literature
- Creating a database

9.2.1 System Borders and Criteria Definition

The Mekong Delta was defined as the assessment area. Focus was put on peri-urban and rural areas where most of the nutrients are produced by population and feedstock.

Suitable substrates for agriculture require the following components in general:

1. Organic matter for soil fertility
2. Alkalinity for buffering acidification
3. Hygienic and eco-toxic safety (no pathogens, toxic compounds)
4. Acceptance for handling, economically, sociologically

Different typical scenarios were checked as these demands vary depending on the agricultural production systems, the soil types and planted crops.

Soil types:
- Acid Sulphate Soils, severely and slightly acid
- Alluvial Soils, in good condition and degraded/compacted

Crop types:
- Trees
- Ornamental plants
- Maize, sugar, cereal
- Rice
- Vegetable

Treatments are to fulfil the following demands:

1. Technical characteristics:
   cleaning efficiency (reduction of turbidity, BOD, COD, P, N, pathogens)
   technical robustness (simplicity of construction and maintenance)
2. Sustainability, environmental issues:
   nutrient transfer to substrate and plant availability of nutrients
   emissions to the environment (climate gas, smell, leaching)
   demand of land, water, energy
3. Feasibility regarding infrastructure and society:
   availability of energy, technical know-how, spare parts
   acceptance (handling, odour)
4. Economic and financial issues:
   cost for construction and maintenance minus benefit

9.2.2 Compared Systems

The systems tested in this study were compared to each other and against a number of further solutions experienced in adjacent studies in the Delta:

- Brown and black water treatment: anaerobic treatment, composting, vermicomposting
- Different yellow water treatments: storage, evaporation, precipitation and stripping

Soil filter has not been evaluated although it might be a suitable option for improved pathogen removal.

9.2.3 System Evaluation and Database Creation

The conducted experiments were assessed (Antonini *et al.* 2008; Paris & Netter 2009; Scharfe 2009; Wieneke 2005; Wohlsager *et al.* 2008) and compared to literature data (Soussan *et al.* 2005; Sub-NIAPP 2002; Watanabe *et al.* 2002 and others).

Experts (producers and mangers of the systems) were interviewed by questionnaires. Their answers were assessed.

These informations were used to evaluate and fill in the criteria.

9.2.4 Creating Selection Tool

Criteria were classified and put together in a database. The questionnaire aimed on water treatment systems and on reuse of substrates of wastewater treatment (application) in agriculture. Both parts were separated in five blocks as pointed out in Table 9-2.

Table 9-2: Requested information on treatment systems and sites in the questionnaires

Treatment Systems	Sites/Substrates
1. Possible input material (type, capacity)	1. Existing sites (soils, cropping systems)
2. Cleaning efficiency and output material	2. Substrate demands (type and amounts)
3. Operation and maintenance	3. Feasibility
4. Costs	4. Benefits
5. Application/Implementation	5. Evaluation

9.3. Results and Discussion

9.3.1 Wastewater Types

The recycling potential of wastewater depends on the nutrient concentration. Different wastewater types such as yellow water, grey water, black water, brown water and animal slurry were examined. In Table 9-3, the different types of wastewater and their recycling potential are summarised.

In the examined sanitation system, about 80% of the total N and P were found in yellow and brown water (together: black water) which accounted for only 18% of the total wastewater volume (Wohlsager *et al.* 2008). By this, black water was five times more concentrated than mixed wastewater from a household.

Table 9-3: Wastewater types and potential benefits

Water type	Concentration of nutrients	Recycling of nutrients	Recycling of water	Biogas production
Rain water	*Low*	-	++	-
Fishpond (extensive)		0	++	-
Grey water	↓	0	++	0
Combined Sewer		0	+	0
Septic tank	↓	+	-	+
Black water, Animal Slurry		+	-	++
Yellow water	*High*	++	-	0

++ Very advantageous
\+ Advantageous
o Not recommendable
\- Not possible

9.3.2 Demands of Sites and Crops

The beneficial reuse of substrates depends on the needs of the agro-ecosystem. Table 9-4 illustrates the demands for the main soil types and typical cropping systems of the Mekong Delta.
The main soil types - acid sulphate soils and alluvial soils - have been characterised in chapter 2.4.2. Various studies have been carried out by members of Can Tho University and their partners, mainly on acid sulphate soils but also on alluvial soils to improve management and plant growth (Becker *et al.* 2005; Chiem *et al.* 2005; Guong & Revel 2002; Hanhart & Ni 1993; Hoa *et al.* 2004; Minh *et al.* 1997; Minh *et al.* 1998; Ni *et al.* 2002; Ni *et al.* 2009; Shibistova *et al.* 2009; Tri & Mensvoort 2004).

Favourable recommendations for Vietnam´s upland crops are given by Dinh (1989):

- Combination of organic and inorganic fertilisers
- Application of NH_4-SO_4 is better than urea
- Alkaline P-fertilisers (e.g. Mg-PO_4) is better than superphosphate

However, these soils are complex systems and there are still open questions to be answered. Nevertheless, main demands of crops grown on these soils are presented.

Table 9-4: Substrate requirements of site systems in the Mekong Delta

Demands for rural, peri-urban situations	N	P	C	Alcalinity	Hygiene
Acid sulphate soil	Alternating aerobic and anaerobic conditions (FeS_2 -> H_2SO_4), very low pH, availability of several nutrients limited (e.g. P,), Al-toxicity occurring				
Trees	+	+	+	+	0
Ornamental plants (flowers)	+	++	+	++	0
Maize, sugar cane	+	++	+	++	+
Rice	+	++	+	++	+
Vegetable	++	++	+	++	++
Alluvial soil (degraded)	Rich in nutrients, low organic matter, older soils tend to compaction				
Trees	+	+	++	+	0
Ornamental plants (flowers)	+	++	++	0	0
Maize, sugar cane	+	+	++	+	+
Rice	+	+	0	+	+
Vegetable	++	++	++	++	++

++ High demand for supply and availability increase

\+ Demand

0 Low demand

\- no demand

The nutrients N and P have to be supplied according to the expected removal by yield. In general, the demand of vegetable is higher compared to cereals or trees.
Carbon content on older alluvial soils is usually low and compaction can be addressed by organic matter supply. On acid sulphate soils it is the alkalinity that is demanded: by increasing pH and buffer capacity, fixed phosphorus can transform to a plant available form which in consequence reduces the necessity of direct application. For phosphorus application, usually a slow releasing, less soluble P fertiliser is recommended.

Hygienic parameters are an important issue and clearly depending on the type of crop: hygienic risk is highest when substrates are applied to vegetable eaten raw, lower for monocotyledons like maize or rice, and a lot less for crops which are not eaten at all (Blumenthal & Peasey 2002; Carr 2005; Ongley 1996; WHO 1989, 2006b).

9.3.3 Evaluation of Tested Treatment Technologies

Several technologies that are all producing reusable substrates have been tested in the Mekong Delta. The results have been compared to literature data, evaluating the technology itself as well as alternative systems. The main findings are summarised in Table 9-5. Further details are explained in the following tables.

Table 9-5: Tested treatment technologies, summarised advantages and disadvantages regarding wastewater treatment and reuse in the Mekong Delta

Technologies for rural, peri-urban situations	Wastewater treatment	Reuse in agriculture
Black / Brown Water		
Anaerobic treatment	+ COD/BOD reduction, additional energy gain, technology already established - investment cost, small pathogen reduction	+ nutrient rich irrigation water, plant available nutrients with low carbon load - danger of pathogen transmission
Composting	+ removal of pathogens, simple technique, no investment cost - (small) work load	+ "basal" fertiliser (Hsieh & Hsieh 1990), (usually) pathogen-free - reduced nitrogen content, nutrients slowly available
Vermicomposting	+ removal of pathogens, additional protein gain, rather simple technique, very small investment cost - work load	+ improved soil conditioner ("clay humic complexes" for structure improvement) - low amount of slowly available nutrients
Ponds	+ COD/BOD, pathogen reduction, simple treatment - space consuming	+ irrigation water - low nutrient load
Yellow water		
Direct use/after storage	+ removal of nutrients from main waste water stream - installation of urinal or diverting toilet, acceptance necessary	+ plant available nutrients rather high concentrated, low pathogen input - lower nutrient concentration than mineral fertiliser, endocrine disruptors

General obstacles appeared when operating more advanced or imported systems due to the existing infrastructure in the Mekong Delta:

- Materials (e.g. pumps, connecting valves, test-kits etc.) were not available locally, import was time-consuming and expensive.
- Electricity was not continuous and stable, not even for urban areas.
- Skilful operators with sufficient technical know-how and mechanical abilities were limited – restricting construction and operation of advanced technology.

Assuming ongoing fast development in Vietnam, these hindrances are likely to disappear in the future. However, for the near future robust technologies are recommended.

9.3.4 Black/Brown Water Treatment

In the Mekong Delta, brown and black water treatment was done in the two tested biogas systems, and septic tanks and (mainly uncontrolled) in anaerobic ponds.

The biogas treatments have been discussed in detail in chapter 4. Organic load can successfully be treated by biogas digesters. Even in low-tech biogas digesters without sophisticated management notable BOD-reduction was reached (see chapter 4). Further advantages are high nutrient transfer rates, small space requirements and additional biogas. As the achieved pathogen reduction of 90% is not sufficient for reuse in agriculture, subsequent treatment is necessary (Arnold & Clemens 2004; Arnold et al. 2006).

Septic tanks can be considered as a non mixed anaerobic digester with a low hydraulic retention time. Installation is mandatory for new houses in Vietnam. These concrete tanks are installed below the houses and are equipped with an overflow and an exhaust pipe.

Anaerobic ponds or anaerobic lagoons are the first step in a designed pond waste water treatment system (Mara 1997). They are reducing organic load by sedimentation and gas production. Usually the produced CH_4 and CO_2 are not collected.

The following table (Table 9-6) compares these treatments. The demands are grouped into technical issues, sustainability, feasibility and economics.

Reference base are the conducted experiments and questionnaires; the data are generalised and supplied by literature data where necessary.

Table 9-6: Comparison of two biogas systems, septic tank and anaerobic pond

	Fixed-dome reactor	Plastic-tube reactor	Septic tank (Sasse 1998)	Anaerobic pond (Mara 1996; Mara 1997; Sasse 1998)
Input	liquid to solid organic substrate			Liquid to sludgy organic substrate
Output	Effluent (daily), Sludge (once in years)		Effluent, Sludge (rarely)	Effluent, sludge difficult to gain
Technical issues	+	+	+	+
Treatment time	HRT: 5-12 days (better 20 days)	HRT: 10-20 days	HRT depending on size: 1-10 days	HRT ~1-2 days (~10-20 for all pond)
BOD/COD reduction	Up to 90%	Up to 70%		
Pathogen removal	90(-99)%	90(-99)%	Up to 90%?	~90% per cell
Technical robustness	Robust construction, maintenance feasible	Plastic sensible, maintenance feasible	Simple, solid system simple maintenance	Robust system simple maintenance
Sustainability	++	++	0	0
N transfer rate	More than 99%	More than 99%	Gaseous losses of N	Gaseous losses of N
Other elements (P, K, Ca, micronutrients)	Dissolved in effluent, partially in sludge	Dissolved in effluent, partially in sludge	Dissolved in effluent, partially in sludge	Dissolved in effluent, partially in sludge
Energy demand	Positive balance	Positive balance	No gain	No gain
Climate gas losses	CH_4 used as fuel	CH_4 used as fuel	Gas exposed to the air	Gas exposed to the air
Water	Quality improvement, nutrients gained	Quality improvement, nutrients gained	Quality improvement	Quality improvement
Feasibility	+	++	+	+
Space requirement for waste of 1 LU / PE	0.75 m^2 < 0.1 m^2/PE	1 m^2 < 0.1 m^2/PE	-- 0.5 m^2	-- 4 m^2
Technical know-how	Feasible	Feasible	Simple	Simple
Labour	Small	small	Small, sludge reuse difficult	Small, sludge reuse difficult
Acceptance (disgust)	high	high	high	Not known
Odour	Not expected	Not expected	Not expected	expected
Economics	+	++	0	+
Cost for construction	250 -300 €	25 – 50 €	Excavation, concrete (~50-100 €)	Excavation
Main economic risks	Several years for amortisation	Damage of plastic	No amortisation	No amortisation
Time for construction	Within weeks	Within days	Within days	Within days
Wastewater treatment for:	Farm, municipality, food processing place*	Small farm, rural community	Household	Low contaminated waste water, open area
Application				
of effluent	Pond, irrigation	Pond, irrigation	Soil filter	Pond
of sludge	Orchard, composting	Orchard, composting	Rarely gained	

* Other digester types will be superior for treatment of organic food waste if organic load is high

Material for reuse in agriculture or aquaculture has to fulfil a certain quality to be regarded as safe (Blumenthal *et al.* 2000a; Blumenthal & Peasey 2002; WHO 2006a, 2006b). To reach this quality, waste water or waste water residues may have to be processed further.

For reuse in agriculture furthermore mass reduction is wanted for facilitating substrate transport to the field.

Composting or vermicomposting improve hygienic quality and lead to water and mass reduction. These processes have been discussed in previous chapters.

The following table (Table 9-7) compares the two tested composting processes.

Table 9-7: Comparison of composting systems (post-treatment for biogas products)

	Composting ("Hot-rotting")	Vermicomposting
Input	Organic substrate, solid – sludgy	Organic substrate, solid – sludgy
Output	Fertiliser: earth, peat-like substrate (as soil conditioner)	Fertiliser: peat-like substrate (soil conditioner) & worms (for trading soils)
Technical issues	+	+
Treatment time	2-6 months	2-6 months
Mass loss	Here: 50-70%	Here: 34-57%
Pathogen removal (by indicator bacteria)	Very efficient, 4-5 log_{10} units in 2 months	Efficient, 2-3 log_{10} units within 2 months
Heavy metals	Increased concentration due to mass loss	Increased concentration due to mass loss
Sustainability	+	++
N transfer rate	~50%	~70%
Other elements (P, K, Ca, micronutrients)	100%, increased concentration due to mass loss	100%, into compost and worms
Energy demand	None – low	None - low
Climate gas losses	CO_2, NH_3	Some CO_2
Soil structural improvement	Long-term improvement of soil	Long-term effect by substrate (and worms)
Plant availability	Slow/continuous release of nutrients	Slow release of plant available nutrients
Feasibility	+	+
Space requirement for waste of 1	Small < 0.25 m^2 PE^{-1}	Small < 0.5 m^2 PE^{-1}
Technical know-how	Some experience	Some experience
Labour intensity	Low	Low - medium
Main Risks	Fly invasion, smell due to anaerobic conditions	Worm loss (predators, living conditions)
Acceptance (handling, disgust)	Rather low	Medium
Cost	0	+
Cost for construction	Marginal	Low for initial compost worms and shelter
Suitable waste(water) treatment for	Municipality, also for farms, backyard	Farm, specialised farm
Suitable application	Upland crops, potting	Potting media

Due to higher temperatures, nutrient losses but also pathogen die-off was higher in composting systems. A large part of the nutrients is transferred although losses of nitrogen can not be avoided completely. Space demand is higher for vermicomposting but in general not too high.

9.3.5 Yellow Water Treatment

Yellow water – containing the majority of nutrients excreted - is the most interesting fraction for reuse as fertiliser. To gain these nutrients separate collection of this fraction has to be reached.
Waterless urinals are applicable excellently if tightness and absence of smell are ensured. Separation toilets are functioning only when used appropriate, i.e. users need to sit on the toilet seat. If males are standing in front of the toilet when urinating – as done in the conducted research - separation toilets fail to separate yellow water from brown water. Furthermore, regular cleaning is necessary to avoid blockages in the yellow water pipes.

In the following, three different treatment options for yellow water are compared (Table 9-8). All have been operated in Can Tho within the SANSED project.

Besides treatment by storage (see chapter 8), urine was processed by solar radiation. The liquid was evaporated in a 2 m^2 sized concrete basin, covered by a inclined glass plate. The system produced a solid N-P-fertiliser and a liquid N-fertiliser (Phong *et al.* 2008).

A "high-tech" solution (DeSaR®) was tested to precipitate PO_4^{3-} by addition of MgO which is leading to MAP/struvite. Remaining nitrogen was stripped subsequently as NH_3 to produce $(NH_4)_2SO_4$ (Antonini *et al.* 2008).

Table 9-8: Comparison of three urine treatment systems

	Closed Storage	Solar evaporation	Precipitation/ stripping
Input		Urine, Yellow water	
Output	Liquid urine	solid N/P, liquid N fertiliser	Solid N/P,liquid N fertiliser
Technical issues	+	+	+
Treatment time	6 months	Depending on size, within days to weeks	Hours to days
Pathogen removal	Efficient with time	Efficient	Efficient
Technical robustness	Simple construction,	Robust construction,	Technical construction
Maintenance during project	Simple	Feasible	Difficult
Sustainability	++	++	+
N transfer rate	95%	In solid and liquid	Mainly in liquid
P transfer	100%	100%	100%, mainly in solid
K and other elements	100%	100%	In solid and liquid
Energy demand	None	Low	High
Climate gas losses	None	None	None
Water	Quality improvement, nutrients gained	Quality improvement, nutrients gained	Quality improvement nutrients gained
Feasibility	+	++	+
Space requirement	< 0.1 m^2 per person	> 0.5 m^2 per person	< 0.03 m^2 per person
Technical know-how	Feasible	Feasible	Simple
Labour	Small	Small	Small, sludge reuse difficult
Acceptance (estimated)	Medium - Low	High	High
Odour	Not expected	Small	Small
Economics	+	++	0
Cost for construction	Very low	Low	High
Main economic risks	Several years for amortisation	Damage of plastic	no amortisation
Time for construction	Within days	Within days	Within weeks
Waste water treatment for:	Household in rural, peri-urban area	Household or community in rural, peri-urban area	Hotel, industrial complex, ward or urban setting
Application	Aquaculture, home garden, adjacent fields	Aquaculture, field crops	Field crops

Storage can be done at household level, other treatment options will be more efficient when organised for a larger amount of material (e.g. more than ~100 inhabitants).

Separately collected urine can be applied as a fertiliser in agriculture or in aquaculture – conditions for reuse have been discussed before (Muskolus 2008; Schönning 2002; Schönning & Stenström 2004).

As nutrient transfer rates in the tropical climate of the Mekong Delta are higher in aquatic than in terrestrial ecosystems, fishponds seem to be a space-efficient way to recycle nutrients.

Treatment and recycling systems, especially for yellow water will save expenses for mineral fertiliser. But even more important, it allows reducing other treatment systems for the residual waste water.

9.3.6 Treatment and Recycling Systems

To illustrate appropriate combinations of treatment technologies and application possibilities to treatment and recycling systems, two examples are given (Table 9-9).

Table 9-9: Suitable combined systems

System	Water Type	Water / Substrate treatment	Application
Improved existing system	Black water	Digester & maturation pond Composting of solids	Fishpond
Separation System	1: Yellow water 2: Brown water	Urine Storage Vermicomposting of organics	Fishpond or garden Soil amendment for degraded alluvial soil

Considering space limitation, an efficient way to treat black or brown water is a comned system of a digester and a maturation pond followed by a fishpond. The digester reduces COD and produces biogas that can be used as regenerative energy. The maturation pond removes additionally pathogens. The effluent provides nutrients to the fishpond. Instead of a maturation pond, a soil filter could be used. If soil filters and/or ponds can be integrated in public green areas or in private garden design, the system can be realised even in peri-urban areas (Stokman 2008). The system should not serve less than 50 inhabitants, or receive combined waste water from a farm with several livestock units.

Urine derived fertilisers contain fast available nutrients these may be used in aquaculture to enhance primary production (Ut 2009). As these fertilisers do not contain organic carbon, they are very suitable for ponds where no carbon is desired.
They are useful to supply to systems where to maintain or increase the yield. However, the liquid $(NH_4)_2SO_4$ fertiliser is less useful on acid sulphate soils as the fertiliser has a rather low pH and does not adding alkalinity but adds more sulphate. They are not able to fulfil the needs of sites where organic matter is needed to improve soil fertility. As a consequence, solely urine fertiliser may not be sufficent in areas with compacted alluvial soils.
First experiments showed that the fertilising effect of evaporated urine was less compared to other urine based substrates, e.g. MAP or urine sediment (Arnold *et al.* 2009).

9.4. Conclusions and Outlook

The examined treatment technologies were able to improve waste water quality and recycle nutrients. Each of them has particular advantages related to the waste water sources or produced substrates. Systems have to be selected according to the demands which may vary according to the available space within the settlements or the natural and sociological conditions. Furthermore, demands on waste water quality will usually require more than one treatment step and be combined to a treatment system.

For appropriate application in agriculture and aquaculture, the produced substrates have to be selected according to the site specific conditions. The fertilising effect not only depends on the nutrient content but also on the consistency and chemical form of the substrate which strongly influences the plant availability and uptake. Further reseach is necessary to predict the effects of the new substate more clearly for different soils and crops.

An evaluation method has been developed to identify site specific recommendations for waste water treatment and agricultural application – based on the prerequisite to recycle nutrients. This may be applied in a computer programme or personally by well instructed local engineers.

This evaluation provides a basic database for selection and recommendations. Further data, on treatment efficency and on fertiliser effects on should be evaluated and added to improve the meaning.

Implementation of waste water treatment systems will become essential for the Mekong Delta especially when regarding the increasing population and intensification in agriculture and aquaculture. However, besides technical criteria, the acceptance of users and the political strength will be determining this development.

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