Study on Thermal Balance Model of Aqueous Biological Aerobic Treatment for Fertilizer Conversion
（高水分バイオマスの肥料変換における液状好気性生物処理の熱収支モデルに関する研究）

Peng Wang

2014

Study on Thermal Balance Model of Aqueous Biological Aerobic Treatment for Fertilizer Conversion

BY

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A Dissertation Presented to the United Graduate School of Agricultural Sciences,
Kagoshima University in Partial Fulfillment of the Requirements for
the Degree of Doctor of Philosophy

2014

ACCEPTANCE

This dissertation entitled "**Study on Thermal Balance Model of Aqueous Biological Aerobic Treatment for Fertilizer Conversion**" hereto attached, prepared and presented by **Peng Wang** in partial fulfillment of the requirements of Doctor of Philosophy is hereby accepted.

Prof. Dr. Masaru Mitsutomi
Adviser and Chairman
Guidance Committee

Summary

Recycling high-moisture waste biomass as a stable and hygienic fertilizer is very essential towards sustainable agriculture production and biomass resources circulation. Autothermal thermophilic aerobic treatment (ATAT) appears to be one of the most favourable technologies for stabilizing and sanitizing it into a biologically-safe, nutrient-rich and odour-free fertilizer product.

In this study, thermal balance analyses were investigated during aerobic treatment including activated-sludge process (ASP) and ATAT to elucidate the thermodynamic process with exothermic reaction and to explore the potential of autothermal thermophilic operation for efficient utilization of high-moisture waste biomass as a fertilizer for agriculture production. From these results, a thermal balance modelling method was developed to determine the specific biological heat yield in aerobic decomposition for quantifying the heating potential of organic substrate.

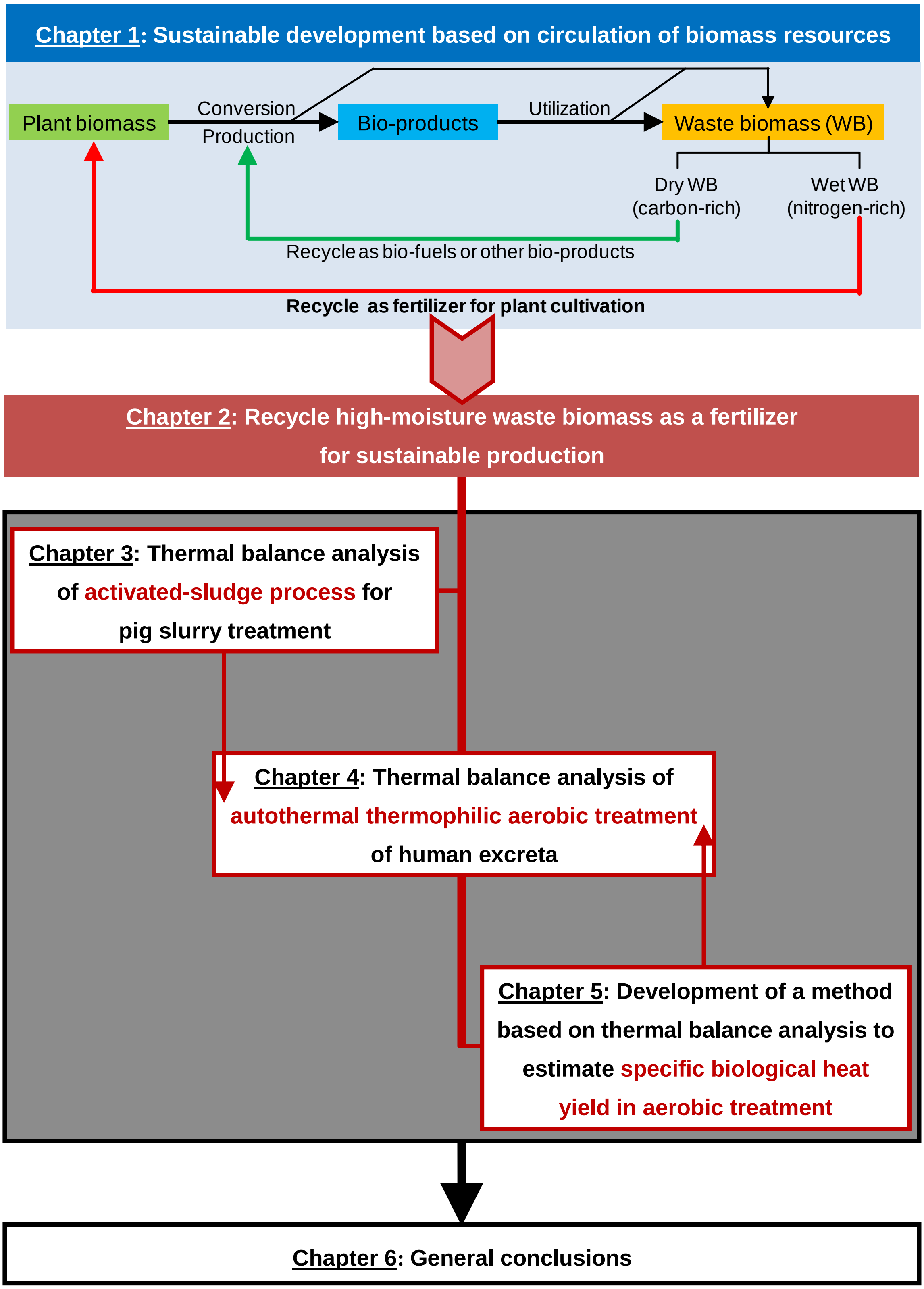
First step, to investigate the potential for heat generation by aerobic treatment, a thermal balance model for ASP was developed and validated with a full-scale ASP facility treating pig slurry. The ASP energy model could accurately simulate the heat budget of the reactor and its accuracy to estimate the reactor temperature was within 0.60 ^{o}C. The results showed that the heat was mostly gained from solar radiation and biological reaction, and mainly lost through slurry surface and slurry flow. Total heat gain had the potential to lead a reactor temperature

rise of 3.87 to 5.15 ℃/d. It implies the need for effective thermal insulation of the reactor. Next step, for quantitative interpretation of ATAT operation, a thermal balance model for ATAT was developed and validated by a full-scale ATAT reactor treating human excreta. The ATAT energy model simulated the thermal dynamic changes in the reactor and the results showed that the biological heat yield in aerobic biodegradation (Q_{bio}) was the principal heat source for self-heating. The specific biological heat yield (Y_{bh}) by degrading 1 g of chemical oxygen demand (COD) in human excreta was estimated at 12.1 kJ/g COD. The specific mean generation rate of Q_{bio} during self-heating phase was estimated at 1.190 MJ/h/t and 5.7 times higher than that after self-heating at 0.208 MJ/h/t. The heat loss from airflow, including sensible heat loss by airflow and latent heat loss by evaporation, was primarily responsible for the decrease in reactor temperature after self-heating.

To confirm the application potential of thermal balance modelling method, a lab-scale aerobic system was developed and this modelling method was examined to estimate Q_{bio} and Y_{bh} during aerobic treatment of artificial wastewater. Estimated Y_{bh} on volatile solids (VS) and COD removal basis were 16.98 ± 1.53 kJ/g VS and 16.40 ± 0.56 kJ/g COD, respectively.

These study results play an important role in understanding the process thermodynamics of aqueous biological aerobic treatment, which enable us to provide proper guidance for treatment system design and modification with the aim to efficient recycling the high-moisture waste biomass as a fertilizer.

Layout and Flow of Dissertation



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Acknowledgement

This dissertation would not have been possible without the guidance and the help of several individuals who in one way or another contributed their valuable assistance for this study. First and foremost, I am heartily thankful to my supervisor, Dr. Munehiro Tanaka (Saga University), for his long-term support, kind encouragement, valuable suggestions and constructive comments throughout the study, and I owe my deepest gratitude to Dr. Masaru Mitsutomi (Saga University), Dr. Shuji Fujita (Saga University), Dr. Iwasaki Koichi, (Kagashima University), Dr. Masami Ueno (University of the Ryukyus) and Dr. Susumu Arima (Saga University) for their valuable comments and kind advices to this study.

Then I would like to express my sincere gratitude to Dr. Yuichiro Wakiya and his colleagues in Saga Prefecture Livestock Experiment Station for their generous support and help to my experiment. And I also extend my heartfelt appreciation to Chikujo town office and the staff worked at the Chikujo Liquid Fertilizer Plant for their great help in the field work.

I am also very grateful to all the fellow students studied in the laboratory of agriculture production engineering, Saga University, especially to Dr. Warachit Phayom, Ms. Mika Ota, Ms. Chiharu Umezaki and Ms. Kikue Yokoo for their kind encouragement and continuous help in my study and life. I would also express my heartfelt thanks to Dr. Zaigui Li from China Agriculture University who introduced me to study in Japan and encouraged me a lot

throughout my study. And I am deeply indebted to China Scholarship Council for the long-term financial support for my study abroad in Japan.

Last but not least thanks, I will never forget my dear family, relatives and friends who always gave me the strength and the courage to overcome the obstacles in my study and life. Without their encouragement and support I cannot insist on the study until now. Finally I would like to offer my regards and blessings to all of those who supported me in any respect during my study and life.

Peng Wang

2014

Chapter 1

Sustainable development based on circulation of biomass resources

"Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs." This is the most widely recognised definition of sustainable development, which refers to a mode of human development in which resource use aims to meet human needs while ensuring the sustainability of natural systems and the environment, so that these needs can be met not only in the present, but also for generations to come (WCED, 1987).

However, the current economic growth is dependent upon high consumption of non-renewable resources, giving an enormous burden to the natural environment (Morioka et al., 2007). Moreover, as the world's population increases and economies become more industrialized, the existing conflict between economic growth and environmental conservation becomes more and more acute and evident. Now it is commonly aware that the conventional social system is unsustainable and environment destroying. There is an urgent need for transition to a sustainable social system (Hopwood et al., 2005).

1.1. Shift from conventional production to sustainable development

In the conventional production system, the materials and energy flow is linear (Fig. 1.1), and the energy used for production is heavily relying on non-renewable fossil fuels whose global reserves are rapidly decreasing (Klass, 1998). The large consumption of limited fossil fuels for sustaining production activities gives off greenhouse gases (GHGs) and air pollutants which are responsible for global warming and acid rain (Omer, 2008).

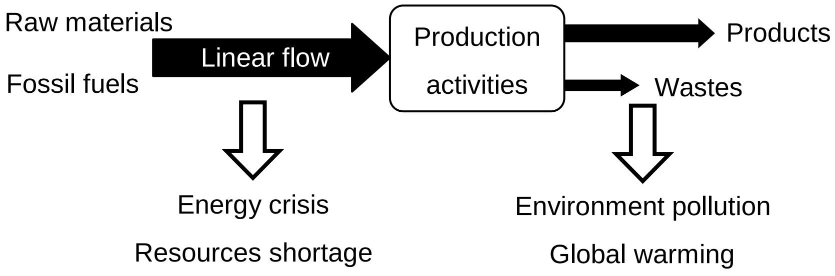


Fig. 1.1 Conventional linear flow production system relying on fossil fuels

To meet rapidly growing demands for food and other products, the linear flow production system only increases the input of materials and energy. Over the past decades, this one-way exploitation of natural resources has contributed to many problems like depletion of mineral resources, scarcity of fresh water, decline of soil productivity and loss of biodiversity, etc (Behrens et al., 2007). Furthermore, the materials and energy intensive production system often generates large quantities of waste that pose a threat to soil, water and atmosphere environment as well as human health (Omer, 2008). And current common waste disposal methods such as ocean dumping, sanitary landfill, open dumping or incineration do much to isolate them away from the public, but little to ensure resources recovery and recycle, putting an increasingly heavy load on the environment (Idris et al., 2004). The arising problems including energy crisis, resources shortage and environment destruction have necessitated the development of sustainable social system.

A circular flow production system based on utilization of renewable resources is expected to reduce the consumption of fossil fuels and its negative effects (Fig. 1.2). The renewable energy sources, such as biomass, wind, hydro, solar, geothermal and tidal, can be used sustainably and environmental-friendly (Midilli et al., 2006). One analysis provided by Lund (2007) showed that converting present energy systems into a 100% renewable energy system for sustainable development in Denmark is possible if further technological improvements on energy systems are achieved.

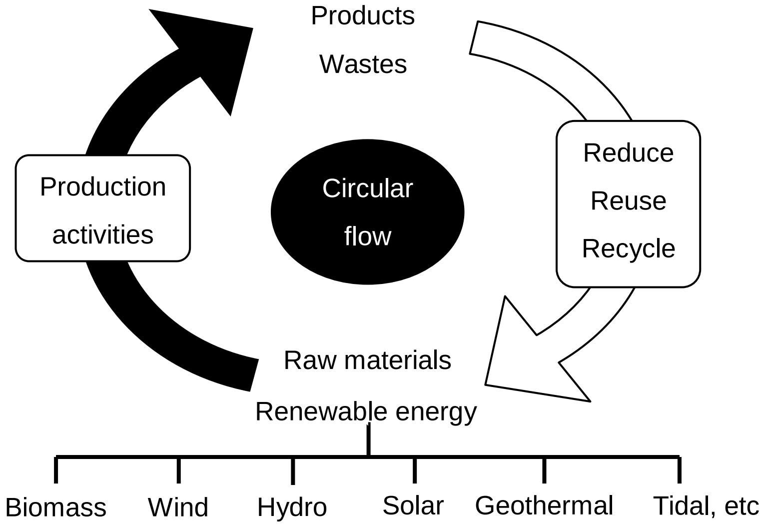


Fig. 1.2 Circular flow production system based on utilization of renewable resources

In sustainable societies, the materials and energy flow are basically circular, the resources value is maximized and the waste generation is minimized by implementation of "3R", i.e. reducing the amount of materials consumed in production and thereby the wastes generated, reusing the unwanted products with simple treatment, and recycling the wastes by using them as resources to produce same or modified products (Takiguchi and Takemoto, 2008). It is widely recognized that various types of wastes from agricultural (crop residues, livestock manure), forestry (wood residues), fishery (aquaculture waste), industrial (effluent from food processing, paper making or tanning industries) and municipal (kitchen garbage, paper waste, garden waste, human excreta, domestic wastewater and sewage sludge) sectors are the potential renewable energy or material sources to attain sustainability (Stehlík, 2009; Kothari et al., 2010).

1.2. Role of biomass in sustainable social system

1.2.1. Potential substitute for fossil fuels

Among all the renewable energy sources, only biomass is a carbonaceous material that stems from organisms, and can be used to make organic fuels or chemicals in place of fossil fuels. Also it is the only one which is stable, storable and transportable in the form of material utilized similarly as fossil fuels (JIE, 2008). In addition, biomass is abundant and available worldwide. The world production of biomass was estimated at 146 billion metric tons a year, mostly wild plant growth (Cuff and Young, 1980). JIE (2008) estimated that the global energy potential of waste biomass was approximately 128 exajoule a year. One analysis provided by the United Nations Conference on Environment and Development showed that biomass could potentially supply about 50% of the current world primary energy consumption by the year 2050 (Ramage and Scurlock, 1996). Biomass has been commonly perceived as one of the most promising substitutes for fossil fuels (Klass, 1998; JIE, 2008). Especially waste biomass conversion to bio-fuels has received a lot of attention (Psomopoulos et al., 2009; Stehlík, 2009; Kothari et al., 2010).

1.2.2. Multiple and cascade usages

Moreover, biomass differs from other renewable resources in that the sources are diversified, and it can be used stepwise like a cascade. Many species of plants are viewed as virgin biomass. They are cultivated, converted and utilized for various purposes, e.g. food, feed, forest products, fibre, industrial feedstock and fine chemicals (Klass, 1998; JIE, 2008). The organic materials discarded from production, conversion and utilization processes are generally regarded as waste biomass, which can be recycled as feedstock for producing feed, fertilizer or bio-fuel, etc (Klass, 1998; JIE, 2008). For instance, food waste can be processed into a good feed for livestock rearing, and the excreted livestock manure can be digested for conversion to biogas or fertilizer. Multiple and cascade utilization of biomass are highly effective, and recycle of waste biomass is very important in avoiding the conflict of bio-fuels production with food and feed supply (Haberl and Geissler, 2000).

1.2.3. Environmental compatibility

Biomass is originally produced via photosynthesis by green plants, and the use of biomass is the reversal of photosynthesis. Thus in principle, by utilizing biomass as energy source, zero net CO_2 emission can be achieved on the assumption that all the released CO_2 during the combustion of biomass will be removed from the atmosphere by carbon fixation via photosynthesis to grow new biomass (McKendry, 2002). In contrast, fossil fuels burning releases carbon which has been stored underground for millions of years, that makes a net contribution to the atmospheric GHGs (Demirbaş, 2001). Therefore, using sustainably produced biomass in place of fossil fuels will reduce the net flow of CO_2 in the atmosphere and assist in alleviating global warming (Gustavsson et al., 1995).

In addition, biomass has negligible sulphur content, and therefore makes little contribution to SO_2 emission that causes acid rain. Also burning biomass produces less ash and smoke than coal combustion that helps in improving air quality and public health (Demirbaş, 2001), and the ash is free of toxic metals and can be used as a soil additive (Haberl and Geissler, 2000).

Furthermore, waste biomass recycling can not only reduce waste disposal problems but also benefit agriculture production and natural systems. Biomass stems from plants, whose growth needs supply of nutrients (N, P, K, etc). In the conventional production system, the nutrients supply is heavily dependent upon petroleum-based chemically-synthesized fertilizers (Parr et al., 1986), and the nutrients flow is one-way, from fertilizer to food to waste (Winblad and Simpson-Hébert, 2004). The system has led to many problems including nutrient loss, water pollution and soil degradation (Esrey et al., 2001) which ultimately destroy the sustainability of biomass production. In recycling-based societies, the plant nutrients are recovered from waste biomass and returned to fields for agriculture production and food supply. The recycling practices benefit nutrients conservation and soil fertility; ensure sustainable biomass production and utilization.

1.3. Importance of waste biomass recycling

Biomass is quite various and there is no established way of categorizing biomass. According to the application of biomass in the conventional production systems, it can be classified into three general categories, namely virgin biomass, processed biomass and waste biomass, as shown in Table 1.1 (JIE, 2008).

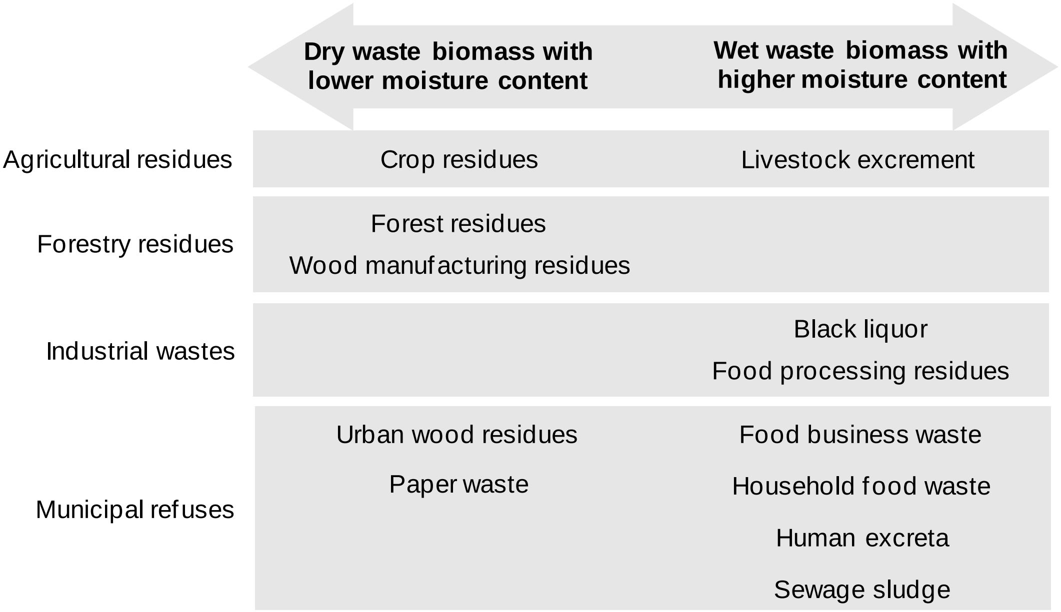
Virgin biomass is referred to a variety of terrestrial and aquatic plants, it is the necessary source of food supply to human and animals, and generally used as raw materials for making kinds of bio-products to meet human living needs. Processed biomass covers a wide range of bio-products derived from green plants, such as meat products, dairy products or wood products, etc. Waste biomass contains various types of discarded organic residues from the activities of human production and living. It plays a more important role as a potential renewable source for conversion to energy or chemicals, since virgin biomass and processed biomass should be preferentially used for providing food and other life necessities to people, and waste biomass recycling as fuels or chemicals would not affect land use and food supply, but would benefit resources utilization and environment protection (Odling-Smee, 2007).

Table 1.1 Classification of biomass according to its application

Classification	Application	Examples
Virgin biomass (Plant biomass)	Raw materials	a) Herbaceous plants: rice; corn; soybean b) Woody plants: poplar; willow c) Aquatic plants: water hyacinth; algae
Processed biomass	Bio-products	a) Agricultural products: food; feed b) Forestry products: timber; wood fuel c) Industrial products: fibre; fine chemicals
Waste biomass	Potential energy or material sources	a) Agricultural residues: straw; husk; manure b) Forestry residues: logging residues; bark c) Industrial wastes: effluent from food processing, fibre making or chemicals production industries d) Municipal refuses: waste wood, paper or food; human excreta; sewage sludge

1.4. Status-quo of waste-to-resources utilization

Currently, waste-to-resources utilization has gained increasing emphasis and encouragement all over the world. It is the inherent properties of the biomass, primarily the biomass moisture content, that determine both the choice of utilization route and any subsequent managing difficulties that may arise (McKendry, 2002). According to the moisture content, various types of waste biomass can be broadly classified as dry waste biomass with lower moisture content and wet waste biomass with higher moisture content, as shown in Fig. 1.3.

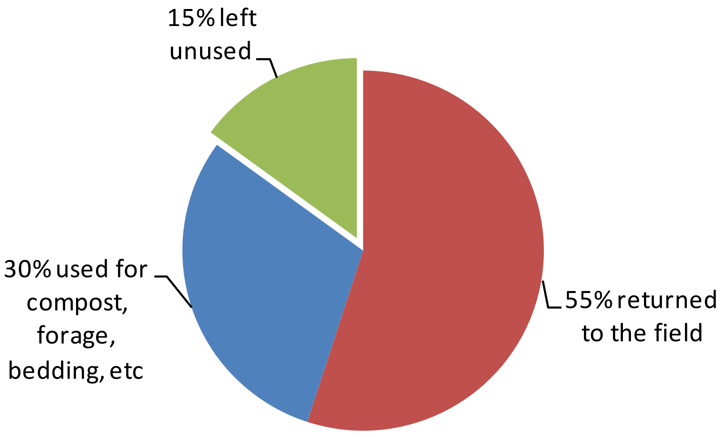


	Dry waste biomass with lower moisture content	Wet waste biomass with higher moisture content
Agricultural residues	Crop residues	Livestock excrement
Forestry residues	Forest residues Wood manufacturing residues	
Industrial wastes		Black liquor Food processing residues
Municipal refuses	Urban wood residues Paper waste	Food business waste Household food waste Human excreta Sewage sludge

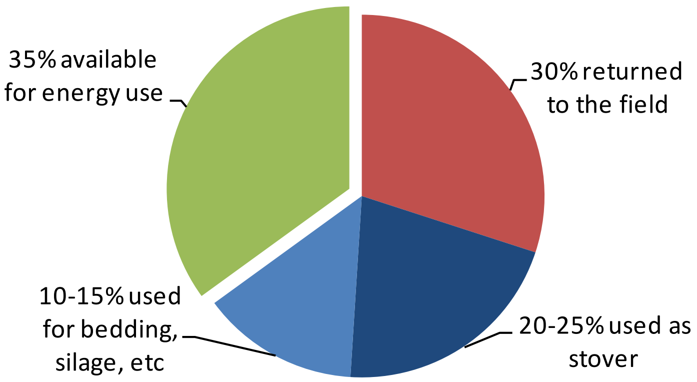
Fig. 1.3 Various waste biomass with different moisture contents

1.4.1. Crop residues

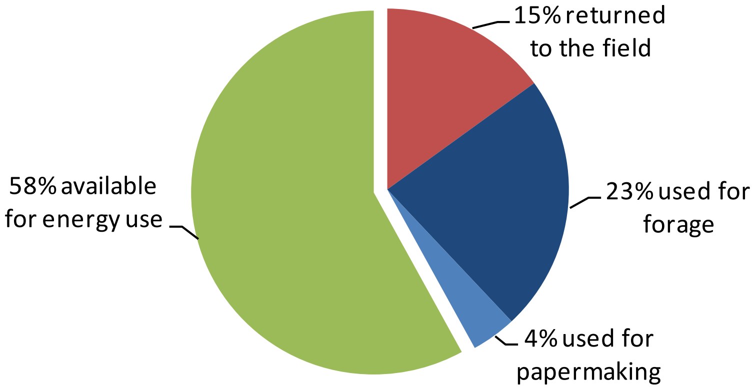
Crop residues are the byproducts of crop production, mainly include straw and stalk of crops. Fig 1.4 shows annual production of crop residues and their utilization in Japan (JMAFF, 2010a), America (Milbrandt, 2005) and China (Liu et al., 2008). As shown in this figure, massive crop residues are used diversely. Parts of them are conventionally returned to the field for soil protection, and parts of them are utilized for compost, forage, bedding or papermaking. The remaining parts are available and suitable for energy use. In rural China, during 1996−2003 about 37% of the crop residues were combusted directly as fuels for livelihood (Liu et al., 2008). Nowadays, for utilizing the energy content of crop residues with higher efficiency, a number of new conversion technologies have been developed and popularized, such as pelletizing, briquetting, gasification and pyrolysis, etc (Zeng et al., 2007). We can expect that more efficient use of crop residues will be achieved in a near future.



14 Mt/y in Japan for 2007
(JMAFF,2011)



494 dry Mt/y in America for 2002
(Milbrandt,2005)



627 Mt/y in China for 1995-2004
(Liu et al., 2008)

(Mt/y: million tons/year)

Fig. 1.4 Annual production and utilization of crop residues in Japan, America, China

1.4.2. Wood residues

Wood residues include forest residues, wood manufacturing residues and urban wood waste (Milbrandt, 2005). Depending on the difficulty in their collection, the utilization status of the residues from wood harvesting, manufacturing and utilization processes are different with each other, as shown in Fig 1.5. In Japan, about 95% of the wood manufacturing residues are recycled as fuel or fibre products, and about 90% of the wood residues generated in construction and demolition sites are generally used as feedstock for paper making or bedding materials. In the case of the forest residues, merely 1% of 8 dry Mt are collected and utilized for pulping, but as the construction of forest road networks and the integration of forestry carry out, more efficient collection and utilization of forest residues can be anticipated. Moreover, modern waste-to-energy technologies like gasification make the wooden residues being a more important biomass feedstock to produce various high-value added commodities such as bio-fuels, electricity, and fine chemicals (JMAFF, 2010a). The same trend can be seen in America (Walsh et al., 2000) and China (Liao et al., 2004; Yang et al., 2010).

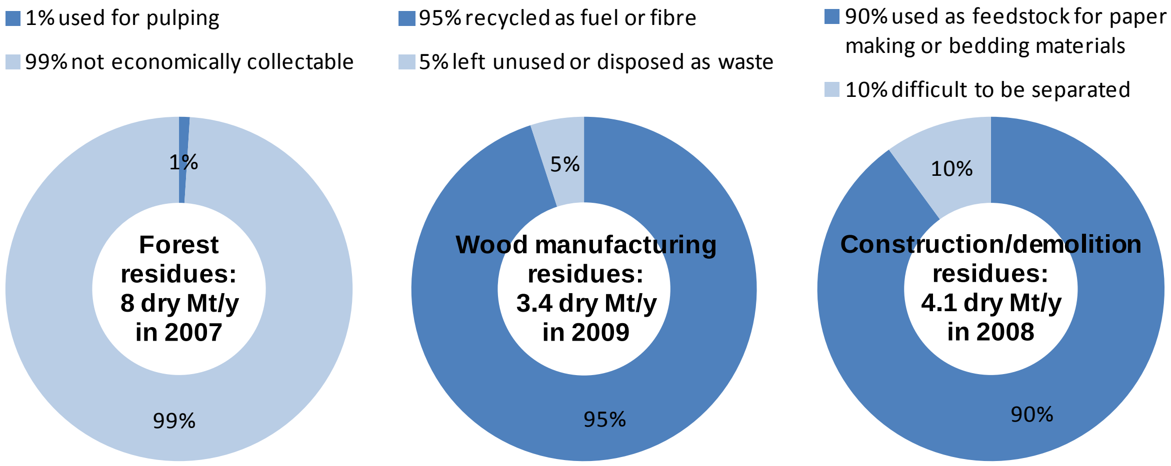


Fig. 1.5 Status-quo of utilization of wood residues in Japan

1.4.3. Paper waste

Municipal collection of consumed paper for recycling is economically and ecologically sound, and recovered paper has been utilized as a vital raw material for producing new paper. In Japan, more than 70% of the used paper has been recovered for recycling each year since 2004, and the paper recycling rate reached 79.7% in 2009 (JPA, 2012). This is considerably high because the share of non-collectable and non-recyclable paper, such as book stock, archives, sanitary paper, etc, is estimated to account for at least 19% of the total paper and board consumption, and the theoretical maximum recycling rate would be 81% rather than 100% (ERPC, 2012). The current progress of paper recycling in developed regions is shown in Table 1.2.

Table 1.2 Paper recycling rate in developed regions in recent years

	Recycling rate, %		
	2009	2010	2011
Japan [a]	79.7	78.2	77.9
EU [b]	72.3	68.9	70.4
U.S. [c]	63.6	63.0	66.8

[a] Source: JPA, 2012

[b] Source: ERPC, 2012

[c] Source: AF&PA, 2012

1.4.4. Black liquor

Black liquor is a liquid by-product from the Kraft process where wood is converted into wood pulp to make paper. It is the mixture of pulping residues (like lignin and hemicellulose) and inorganic chemicals used in the process. It is routinely burned as a liquid fuel in a recovery boiler to produce energy and recover the cooking chemicals needed for paper making (JIE, 2008). Nearly all of the black liquor, about 14 dry Mt/y in Japan, is used as a bio-energy resource at Kraft paper mills (JMAFF, 2010a). Now, new waste-to-energy methods like black liquor gasification have been developed, showing to have higher energy conversion efficiency and GHGs reduction potential than the conventional recovery boilers (Naqvi et al., 2010).

1.4.5. Food waste

According to the sources of food waste, it can be divided into three parts: food processing residues, food business waste, and household food waste. Table 1.3 shows the Japanese status-quo of management of food waste occurred at the stages of processing, marketing and consumption. Because food processing residues is concentrated at one source, relatively homogeneous and clean, up to 95.5% of the residues has been currently recycled for various purposes, such as feedstuff (69.0%), fertilizer (14.4%), biogas (3.4%) and other products. Although the recycling rate of food business waste and household food waste didn't reach such a high level, it still can be seen that the food waste is versatile and has a great utilization potential (JMAFF, 2010b).

Table 1.3 Status-quo of food waste disposal and utilization in Japan

Division	Annual generation, kt/y [a]	Annual disposal, kt/y [a]	Annual recycling, kt/y [a]	Recycling purposes	Individual recycling quantities, kt/y [a]
Food processing residues [b]	15098	674 (4.5%)	14424 (95.5%)	Feedstuff	10419 (69.0%)
				Fertilizer	2180 (14.4%)
				Biogas	520
				Heat	486
				Oil and fat products	307
				Carbonized products	37
				Ethanol	7
				Other	468
Food business waste [b]	3644	2876 (78.9%)	768 (21.1%)	Feedstuff	290
				Fertilizer	269
				Biogas	22
				Heat	1
				Oil and fat products	136
				Carbonized products	4
				Other	46
Household food waste [c]	10310	9730 (94.4%)	580 (5.6%)		

[a] kt/y: thousand tons/year

[b] Data year: 2010; Source: JMAFF, 2010b

[c] Data year: 2009; Source: JMAFF, 2009

1.4.6. Sewage sludge

Sewage sludge is the excess sludge discharged from an activated-sludge wastewater treatment process, which is usually a mixture of microorganisms and settable solids in wastewater, rich in organic contents and plant nutrients. However, sludge may also contain pathogens, organic pollutants or heavy metals. Before land application, municipal sewage sludge has to be treated with the purposes of stabilization and pasteurization via the processes such as anaerobic digestion, thermophilic aerobic digestion, lime conditioning, dewatering and storage, etc (Fytili and Zabaniotou, 2008). With different treatment processes and degrees, the sewage sludge is recycled and utilized diversely, e.g. used as fertilizer or soil amendment in agricultural applications, or used as construction materials like aggregate, tile and water-permeable block in civil engineering applications, or used as fuel gas to generate electricity or steam/hot water (JIE, 2008). In Japan, of an annual generation of 78 Mt of sewage sludge in 2008, about 77% was recycled as building materials or compost (JMAFF, 2010a). In the United States, about 50% of 7 dry Mt of sewage sludge produced in 2004 was applied to land as soil amendment or fertilizer (NEBRA, 2007).

1.4.7. Livestock excrement

Livestock excrement is one of the main waste materials in agriculture production, and it accounts for a significant part in the waste biomass. The estimated quantities of annual excretion of feces and urine for major livestock breeds in Japan are shown in Table 1.4. It indicated that in 2012 about 84.5 Mt of livestock excrement (including 26.9 Mt of urine) was generated, which included cattle excrement of about 49.3 Mt, pig excrement of about 22.5 Mt and chicken excrement of about 12.7 Mt, accounting for a quarter of biomass resources present in Japan (Minami and Saka, 2005; JMAFF, 2012).

Table 1.4 Annual generation of livestock excrement in Japan of 2012

Livestock breeds	Annual generation, Mt/y		
	Feces	Urine	Feces & urine
Dairy cattle	18.5	5.5	24.0
Beef cattle	18.5	6.8	25.3
Swine	7.9	14.6	22.5
Layers	7.7	-	7.7
Broilers	5.0	-	5.0
Total	57.6	26.9	84.5

Source: JMAFF, 2012

The large amounts of livestock excrement contain a great deal of easily biodegradable organic matter and plant nutrients, carry bacteria and insect spawn, which are liable to cause environmental pollution. For pollution prevention and material recycling, several livestock waste management methods have been developed in Japan according to the characteristics of the excrement. In chicken farms, the excrement is usually air-dried at the henhouse, and the dried manure is conventionally used as good compost. For the swine excrement, urine is more than feces. The waste discharged from pig farms is usually in the liquid form, and the most common management method in Japan is solid–liquid separation followed by aeration treatment of liquid part and composting of solid phase. In the case of cattle farming, most of the farmers have their own farmlands. The liquid fraction of cattle waste is often stored and then applied on their farmlands, and the solid fraction is generally composted. Besides, anaerobic digestion or liquid composting of cattle/pig slurry has been promoted (Harada, 1996). Through anaerobic digestion, a combustible biogas can be produced for farms energy use, and the digested liquid residue can be used as a source of fertilizer (Fujino et al., 2005). By liquid composting, the organic waste slurry can be turned into a biologically-safe fertilizer product (Urano et al., 1982). JMAFF (2012) reported that about 90% of the animal feces had been recycled in Japan since 2004.

The situation of livestock waste management in China is much worse. As shown in Table 1.5, the estimated amounts of annual generation of livestock waste are several times higher than those in Japan. It is estimated that about 1 Mt of urine and flushed water are discharged everyday and only a limited quantity is under control (Li et al., 2005). A project to assess the availability of biomass resources in China reported that only 20% of the livestock excrement and foul water was processed to different degrees and only 10% was processed by biogas projects; about 80% of the farms discharged their refuse directly to the rivers or fields (Su et al., 1998). The arbitrary discharge of the waste other than the rational recycle of the waste for production has caused not only large losses of nutrient resources but also serious environmental pollution in China.

Table 1.5 Annual generation of livestock excrement in China

Livestock types	Annual generation, Mt/y		
	1997	2005	2010
Cattle	853	1294.6	1736.2
Pig	284	506.6	729.2
Chicken	279.5	293.7	307.9
Total	1416.5	2094.9	2773.3

Source: Li et al., 2005

1.4.8. Human excreta

Management of human excreta is a critical part of our daily life (Esrey et al., 2001). It was estimated that each person would produce about 2.2 t of excrement per year (Li et al., 2005), and it would contain approximately 6.3 kg of elemental NPK which could enable to produce 250 kg of cereals that one person needs to consume in a year (Drangert, 1998). However, human excreta are seldom reused and recycled as a resource for food production. The current management systems seek to isolate them away from our life mainly because of pathogens consideration, which have led to large losses of nutrient elements (Mihelcic et al., 2011).

To be specific, the common management way found in developing regions is hiding human excreta in latrine pits, which could raise potential risks of groundwater contamination and infectious diseases outbreak (Esrey et al., 2001). The sanitation practice promoted in developed regions for centuries is based on flush toilets, it seeks to dilute and carry away the human waste by large volumes of water via a sewer collection system, then purify the massive foul water in an energy and material intensive manner, and then release them into nearby water bodies, often in large quantities that may degrade the waters (Mihelcic et al., 2011).

In Japan, the toilet system was traditionally non-water based and night soil (a euphemism for human excrement collected from cesspits) was stored for agriculture use, but the application of night soil was diminished by the popularization of chemical fertilizer after the 1950s, then the arising sanitation problems were tackled by constructing public sewerage systems and developing on-site wastewater treatment systems, so called “johkasou” systems in Japanese terminology (Magara, 2003). Up to now, almost all Japanese citizens can assess to the advanced treatment of human excreta, 92.6% of the total population is equipped with flush toilets connected to sewerage systems (70.6%) or johkasou systems (22.0%), and the remaining 7.4% is still using pit toilets (JME, 2011). The wastewater collected by the sewerage is purified at municipal wastewater treatment facilities, while the excreta in toilet pits and the sludge in johkasou are collected by vacuum trucks, then threw into sewerage systems or sent to night soil treatment facilities that also employ advanced treatment like wastewater treatment plants (Matsui et al., 2005; JME, 2011).

The conventional systems used in developed courtiers have reduced the spread of diseases, but they have consumed considerable energy and water for elements removal, and the treated water can still cause water quality problems (Muga and Mihelcic, 2008). From the viewpoint of nutrients recycling and conservation, the nutrient elements in human excreta should be flowed back to food chain, instead of discharging into water bodies or hiding in deep pits. Therefore, a renewed perspective of sanitizing and recycling human excreta for agriculture use is gaining growing interest now (Drangert, 1998; Heinonen-Tanski and van Wijk-Sijbesma, 2005; Mihelcic et al., 2011; Karak and Bhattacharyya, 2011).

1.5. The need to more efficient use of high-moisture waste biomass

The dry waste biomass with lower moisture content, like crop residues, wood residues and municipal solid waste, is normally carbon rich, can be favorably stored (Gajdoš, 1998). The dry waste biomass is more economically suited for conversion into biomass energy by gasification, pyrolysis or combustion (Demirbaş, 2001; McKendry, 2002). Nowadays, many researches and applications have been dedicated to the technologies to produce bio-fuels from herbaceous biomass (Phayom, 2012), woody biomass (Miura et al., 2004) or municipal solid waste (Psomopoulos et al., 2009). The dry waste biomass has become a more and more versatile resource to provide bio-fuels or other bio-products.

By comparison, the wet waste biomass with higher moisture content, such as livestock excrement and human excreta, is often nitrogen rich and more readily biodegradable, which is more suited for aqueous biological conversion to recycle as fertilizers (Gajdoš, 1998). However, most of the existing systems for handling it do much to remove the chemicals and pathogens in the raw material, but little to ensure the recovery and recycling of nutrients in it. This has contributed to large losses of nutrient elements, which would eventually lead to unsustainability of biomass production and utilization (Martinez et al., 2009; Mihelcic et al., 2011). Therefore, it is necessary to construct a new system which can stabilise, sanitise the high-moisture waste biomass and recycle the nutrients in it back to the agriculture production for sustainable development.

Chapter 2

Recycle high-moisture waste biomass as a fertilizer for sustainable production

High-moisture waste biomass, e.g. animal waste and human waste, is foul, stinky, perishable and troublesome in handling, which can also contain weed seeds, insect larvae and pathogenic organisms. The fresh, non-treated waste slurry could be a risk for the environment, animals and humans. These reasons have largely restricted the direct spreading of raw slurry on the land (Mohaibes and Heinonen-Tanski, 2004).

Treatment technologies can play an important role in the management of high-moisture waste biomass by improving its storage, handling and fertilizing properties and by minimizing the risks of pathogen transmission and environmental pollution. Such treatments are based on physical and biological processes, with the possible use of chemical additives (Martinez et al., 2009). The applicable technologies include drying, struvite precipitation, composting, aerobic treatment and anaerobic fermentation, etc, by which the raw waste slurry can be treated and converted to solid-type or liquid-form products that would be suited for fertilizing farmland.

2.1. General description of waste slurry treatment methods and their limitations

2.1.1. Drying

Drying refers to the removal of moisture from the raw waste slurry so that the volume, viscosity, odour and foulness associated with the raw material can be reduced, and the rate of deterioration by microbial activity can be minimized (Ghaly and MacDonald, 2012). Drying allows for easier storage and handling of the material (Bernhart and Fasina, 2009). However, the dried products still contain lots of readily biodegradable organic matter as much as the raw material. Large application of dried organic waste to the field is liable to cause nitrogen barriers for the plants (Harada, 1996). Moreover, drying a liquid waste often requires additional heat supply from electricity, fossil fuels or solar energy, and the most energy-saving way—drying by the sunlight is lack of efficiency and also gives out bad smell (ASABE, 2009).

2.1.2. Heat treatment

Heating slurry increases the slurry temperature, enhances the ammonia liberation. The high temperature and sufficient thermal exposure are responsible for effective inactivation of pathogens present in the slurry, and the ammonia liberation is also inhibitory to many enteric bacteria (Heinonen-Tanski et al., 2006). This process leads to bacterial and viral reduction, but the process is costly, and the liberated ammonia would need to be recovered by using scrubbers to reduce the ammonia emission to atmosphere (Mohaibes and Heinonen-Tanski, 2004).

2.1.3. Lime addition

Alkaline chemicals, like lime, are widely used in agriculture to counteract soil acidification, and they are used to destroy salmonella or other pathogens present in animal manure (Grabow et al., 1978; Mohaibes and Heinonen-Tanski, 2004). Addition of lime to the slurry is an exothermic reaction and the temperature increase would be experienced. This would result in inactivating microbial cells and viruses. Also, the increased pH and ammonia emission at high pH (over 10) are responsible for disinfection of the slurry (Heinonen-Tanski et al., 2006). But the intensive emission of ammonia would be a big drawback of the treatment method, and treating slurry with lime is not recommended for large-scale and long-term use.

2.1.4. Struvite precipitation

Struvite precipitation permits ammonium and phosphorus to be recovered from different wastewater effluents in the form of magnesium ammonium phosphate ($MgNH_4PO_4$, commonly known as struvite), which can be used as a valuable slow-release fertilizer (Uysal et al., 2010). On the other hand, effective struvite formation usually requires large addition of magnesium and phosphate salts, which leads to a high operation cost, hampers the widespread application of the struvite process (Huang et al., 2011).

2.1.5. Solid composting

Composting, commonly used to treat semisolid or solid organic waste, represents a thermophilic waste treatment technology (Lapara and Alleman, 1999). Within this process, the temperature could rise to 70–80 ℃ by utilizing the heat from microbial aerobic decomposition of the organic substrate, and the pathogens, parasites, insect larvae and weed seeds trapped in the raw waste could be eliminated with the high temperature (Bernal et al., 2009). A stabilised, sanitised product like soil humus can be obtained. Its fertilizing effect would be long and stable for the plants and soil, since the fraction of readily biodegradable organic matter has been decomposed (Harada, 1996). However, this process is less suited for treating liquid organic waste, because the high moisture content needs to be pre-adjusted before composting by drying or mechanical dehydrating or admixing with sawdust, rice hull or straw (Ahmad et al., 2007).

2.1.6. Wet anaerobic fermentation

Anaerobic fermentation is a collection of biological aqueous processes by which specialised bacteria break down biodegradable organic matter into carbon dioxide and methane that can be used as a gaseous fuel, while the digested slurry is nutrient-rich and can be used as a fertilizer (Wellinger, 1984). Since the methane producing bacteria are rather sensitive to environment factors, the design of anaerobic reactor system is usually complicated, and the methanogenesis process needs to operate at a specific constant temperature preferable in mesophilic (20–40 ℃) or thermophilic ranges (45–70 ℃). It is stated that thermophilic anaerobic digestion can offer a higher reaction rate and better pathogen reduction effect over mesophilic anaerobic process (Heinonen-Tanski et al., 2006). Furthermore, because the methanogenesis process itself releases a fairly small amount of heat and most of the substrate energy is retained in the form of methane, the anaerobic reactors, thermophilic or even mesophilic, often require an external heat supply to maintain reactor temperature, especially in cold areas (Heinonen-Tanski et al., 2005).

Nowadays, many anaerobic fermentation plants have been constructed for biogas production, and the effective use of the byproduct, digested slurry, is becoming more and more important to be studied. Furukawa et al. (2009) analyzed the chemical properties of the digested liquor from a methane fermentation plant at Yamaga Biomass Center, Kumamoto Prefecture, Japan, which treats raw garbage, livestock manure and sewage sludge at 37 ^{o}C. The digested liquor contained five-day biochemical oxygen demand (BOD_5) of 1000–1500 mg/L, chemical oxygen demand (COD) of 8000–10000 mg/L, total suspended solids (TSS) of 2000–3000 mg/L, total nitrogen (TN) of 1600–2200 mg/L, ammonia nitrogen (NH_4-N) of 1400–1600 mg/L, and nitrite nitrogen (NO_2-N) and nitrate nitrogen (NO_3-N) were not detectable. Phayom et al. (2012) studied the effect of application of the digested liquor for rice cultivation in Yamaga city, Kumamoto Prefecture, Japan. It showed that the grain yield of the paddy field fertilized with the digested liquor from Yamaga Biomass Center was around 6.79 t/ha, while the grain yield of the field applied with commercially available chemical fertilizer was about 6.51 t/ha. And similar rice cultivation experiments with digested slurry were carried out in Jintan city, Jiangsu Province, China. The digested slurry used there was the liquid residue from anaerobic digestion of swine waste, it contained total kjeldahl nitrogen (TKN) of 1500–2500 mg/L and NH_4-N of 1300–1900 mg/L, and it was applied to the paddy field by mixing with irrigation water. The cultivation results showed that the grain yields of the fields applied with the digested slurry or chemical fertilizer were 8.64 t/ha or 7.91 t/ha, respectively.

According to the chemical characteristics and application experiences of digested slurry, it was found that the fertilizing effect of digested slurry was readily available because the fraction of mineral NH_4-N can be absorbed by the plants immediately, and then the fertilizing effect became weaker but can still last for about a hundred days because the ammonification of organic nitrogen would occur and release NH_4-N over time in the soil. Besides, it is more important to note that large application of digested slurry may cause plant physiological disorders because anaerobic digestion usually can not break down organic matter completely, remaining many kinds of organic compounds in the digested slurry (Tanaka, 2004).

2.1.7. Aqueous aerobic treatment

Aerobic biological treatment has been used to tackle a range of the problems caused by organic waste slurry all along. Compared to anaerobic process, the relatively strong oxidizing environment leads to a more extensive biodegradation of organic matter by microorganisms, with water, carbon dioxide and other simple molecules being the products. In this way, many of the organic compounds related to offensive odours such as phenols, cresols, indoles and volatile fatty acids are removed. As the process progresses, the BOD and COD fall as the more readily degradable material is consumed, and the slurry becomes more stable. The presence of aerobic conditions also leads to a reduction in pathogen numbers. Meanwhile, aerobic treatments result in the generation of considerable amounts of microbial oxidation heat, whereas anaerobic treatments often need some warming to maintain the slurry temperature (Burton, 1992).

2.1.7.1. Conventional aerobic treatment

For many years, simple aeration process (SAP) and activated-sludge process (ASP) have been dominant in the treatment of waste slurry for the simplicity of the processes and effective reduction of odour and COD (Bernet and Béline, 2009). SAP can be easily carried out by deploying an aeration device in a storage vessel or lagoon. It is commonly used in small livestock farms for odour control and effluent disposal (Burton, 1992). ASP mainly consists of two stages, the first provides aeration of the waste slurry, and the second removes suspended solids by sedimentation. The application of ASP is widely seen in many centralised wastewater treatment plants, in which COD, N and P are removed to a very low level, and the treated effluent is discharged to nearby water courses (Wang et al., 2013a). To achieve high removal rates for meeting discharge standard, pre-treatment and post-treatment units are often added to these processes, such as dilution with water, solid−liquid separation, and membrane filtration. As a result, the treatment cost would increase as well (Béline et al., 2004).

The primary problem of the conventional treatment systems is that they treat organic waste slurry as a hazardous material rather than a useful resource, they lay too much emphasis on the removal of chemicals for final discharging, but little to ensure the nutrient elements to be recovered and recycled for agriculture production (Muga and Mihelcic, 2008). The direct consequence is that there is no payback from the costly treatment processes, since the nutrient components in the treated effluent are not enough to be recycled as a mineral liquid fertilizer (Nakamaru et al., 2003; Tanaka, 2004). The more serious consequence is the damage to the global biogeochemical cycle of nutrient elements (Mihelcic et al., 2011).

2.1.7.2. Thermophilic aerobic process

Compared to SAP or ASP that usually operates under mesophilic conditions, thermophilic aerobic process (TAP) would offer many benefits including higher biodegradation rates, more efficient pathogen inactivation, process simplicity and robustness, and the potentialities of heat recovery and nitrogen conservation (Juteau, 2006). It is reported that the high temperature would lead to elevated biodegradation rates that reduce the necessary detention time for treatment, and thus the capital cost for facility construction. In addition, high biodegradation rates may improve process stability by allowing for rapid recovery from upset conditions (LaPara and Alleman, 1999). The enhanced effect of pathogen inactivation by thermophilic conditions is also a primary positive consideration for waste slurry treatment (Martens and Böhm, 2009). Besides, the nitrification activity would be inhibited over 40 °C and thus the nitrogen removal through nitrification and de-nitrification would be restricted (USEPA ,1990; Blackburn, 2000).

The above benefits are favourable in obtaining a stable, hygienic and odour-free product, and the product would be suited to apply on the farmland as a fertilizer or soil improver with few risks for soil and plants (Skjelhaugen, 1999, Mohaibes and Heinonen-Tanski, 2004). The fertilizing properties of the treated product are similar as solid compost. It is almost free of readily biodegradable organics and pathogenic organisms, mainly contains recalcitrant organics and mineral constituents that can provide not only soil organic carbon but also plant readily available nutrients (Tanaka, 2004). However, the difficulty of raising a full-scale reactor to temperatures of 45 °C or higher sets a practical limit for application of TAP to treat liquid slurries, because the large amount of water contained in the waste slurries would need to be heated and simultaneous water evaporation would also take the heat away (Lapara and Alleman, 1999).

2.2 Importance of thermal balance analysis for high-moisture waste biomass treatment

To sum up, in order to stabilise and sanitise the high-moisture waste biomass for turning it to a fertilizer product for sustainable agriculture production, it is better to choose a wet/aqueous conversion process, especially TAP, and one of the first concerns to use TAP is to determine if self-heating phenomenon can be observed and in what conditions self-heating of an aerobic reactor to thermophilic temperatures can be achieved. The self-heating operation of TAP is referred to as autothermal thermophilic aerobic treatment (ATAT), which is cost-effective and environmental-friendly for high-moisture waste biomass treatment (Juteau, 2006).

As we know, the most important mechanism of aerobic treatment is oxidative decomposition of organic matter by microorganisms, producing decomposition heat. In the case of solid composting, this heat accumulates owing to the build-up of solids that insulate heat in the material, and thus the self-heating phenomenon is commonly observed in this process. During aqueous aerobic processes, heat is also created by the decomposition of organic matter, but simultaneous heat loss occurs more easily and the temperature in the reactor is often similar to the influent slurry temperature. It implies that if the produced heat is greater than the heat loss, the temperature of treated materials would rise or become the same as the ambient. Therefore it is important to reduce heat loss and enhance heat production for developing ATAT operation.

Efficient heat conservation is needed for decreasing heat loss. A common method for heat retention is to insulate the system exposed to air with the material of low thermal conductivity. Blackburn (2001) reported that a reactor cover insulated with a 0.1-m-thick layer of fibreglass is a critical requirement for successful TAP operation at lower concentration of solids. Another simple method is to cover the slurry surface with hollow plastic balls for reducing evaporation loss. In addition, it is important to choose a highly efficient aerator for minimizing the quantity of air that has to be injected to the reactor and conserving the sufficient amount of oxidation heat in the reactor to support self-heating (Juteau, 2006).

Besides, adequate organic loading is required to achieve self-heating. USEPA (1990) suggested a minimum content of volatile solids (VS) at 2.5% for ATAT of municipal wastewater sludge. Skjelhaugen (1999) and Ahn et al (2007) found that with higher VS content in the substrate the heat generated through VS degradation became greater. Ponti et al. (1995) studied the aerobic thermophilic treatment of sewage sludge at a pilot plant scale and they reported that high COD concentration in the feed positively influenced heat generation and degradation efficiency.

Many researchers have revealed that the heat generated from aerobic biodegradation was nearly proportional to the consumption of organic substrate. The biological heat yield in aerobic biodegradation (Q_{bio}) has been quantified as a linear function of organic removal in previous studies (Jewell and Kabrick, 1980; Vismara, 1985; Lapara and Alleman, 1999). Metcalf & Eddy Inc, 2003 suggested that an empirical value of specific biological heat yield (Y_{bh}) from degrading of 1 g VS was 20 kJ/g VS. McCarty (1972) proposed a heat generation of 14.6 kJ by consuming 1 g COD for many common organics in aerobic respiration and synthesis reactions. Messenger et al. (1993) reported that the Y_{bh} in ATAT of sewage sludge ranged from 11.4 to 14.6 kJ/g O (1 kJ/g O = 1 kJ/g COD). Gomez et al. (2007) gave a Y_{bh}

value of 13.6 kJ/g COD for estimation of Q_{bio} during ATAT of sewage sludge. The value of Y_{bh} is useful in indicating the self-heating potential of an organic substrate and estimating the level of organic removal necessary for ATAT development.

Analysis of thermal balance during aerobic processes can assist us to determine reactor heat budget and evaluate Y_{bh} value, it enable us to give effective instructions to produce, utilize and conserve the heat for self-heating operation. However, the process thermodynamics of aqueous aerobic biological treatment was rarely analyzed and not well-defined, especially for ATAT.

In this study, thermal balance analyses were investigated during aqueous aerobic processes including ASP and ATAT to elucidate the thermodynamic process with exothermic reaction and to explore the potential of autothermal thermophilic operation for efficient recycling the high-moisture waste biomass as a fertilizer product, then a simple method based on thermal balance modelling was developed to determine Y_{bh} in aerobic biodegradation on a VS or COD removal basis with the purpose to explore a practical ATAT condition by necessary organic removal level.

First step in Chapter 3, thermal balance was examined over an activated-sludge reactor (ASR) to simulate the heat budget on the reactor and to investigate the potential for autothermal operation of the ASR to higher temperatures; Secondly in Chapter 4 thermal balance was examined over an ATAT reactor to simulate the thermal dynamic changes in the reactor and to elucidate the principle of autothermal thermophilic operation of the reactor; then in Chapter 5, to confirm the application potential of thermal balance modelling method, a lab-scale aerobic system was developed and this modelling method was examined to estimate Q_{bio} and Y_{bh} during aerobic treatment of artificial wastewater.

Chapter 3

Thermal balance analysis of activated-sludge process for pig slurry treatment

3.1. Introduction

The inappropriate management of pig slurry resulting from intensive livestock production has caused many environmental problems and health hazards because of the point-source accumulation of pig slurry, which often exceeds local farmland nutrient demand (Mart ńez-Suller et al., 2008). In-place storage and spread of large quantities of pig slurry causes over fertilisation of the soil, and high levels of chemicals such as N and P contained in pig waste affect adjacent watercourses (Juteau et al., 2004). Moreover, direct transportation and distribution of pig slurry to a wider area is not feasible owing to problems related to malodour, foulness and pathogens (Nicholson et al., 2005). Thus, pig slurry must be stabilised and sanitised on the spot before reuse and recycle.

The waste discharged from pig farm is usually in the form of liquid slurry, because the quantities of excreted urine would be much greater than that of pig feces. Currently, the most widespread treatment method for its management is solid−liquid separation followed by activated-sludge treatment of liquid slurry and composting of solid phases (Wakiya et al., 2000).

Both the composting and ASP are the biological aerobic treatment processes that make use of microorganisms to degrade organic compounds into carbon dioxide, water and other simpler products, producing oxidative decomposition heat. Composting accumulates the heat and it represents a thermophilic aerobic treatment process, while ASP usually operates under mesophilic or ambient temperature conditions. During summer, there is no problem, but during winter, purification performance is often degraded by the lower temperatures because it influences the microbial activity, and thus heat insulation is required for maintaining treatment efficiency in ASP (Wakiya & Sakai, 2002). Besides, unlike the compost of pig-feces that can be recycled as a fertiliser or soil additive, the treated liquid effluent by ASP is seldom reused as any useful materials, and it is often discharged to nearby watercourses, although the treatment cost of ASP is much higher than that for composting (Bernet & Béline, 2009).

Thermophilic aerobic operation can be induced by effective heat insulation and adequate organic loading. The applicability of thermophilic aerobic slurry treatment has been reported in the literature, e.g. the biodegradation of swine waste (Juteau et al., 2004), the bio-oxidation of cattle slurry and food waste (Heinonen-Tanski et al., 2005), the biodegradation of potato slops (Cibis et al., 2006) and the bioremediation of palm oil mill effluents (Chan et al., 2011). It implies that the conventional ASP for pig slurry treatment could be modified and operated under thermophilic, or at higher temperatures, and a beneficial product using as a nutrient solution could be expected. Determining the heat budget during ASP at full scale is extremely important for maintaining its biodegradation ability and for analysis enabling proper guidance for its modification.

Previous studies on aerated basin temperature modelling were reviewed by Talati and Stenstrom (1990); their model accounted for the heat exchanges through surface evaporation, surface convection, atmospheric radiation, aeration, tank wall conduction and convection, solar radiation, power input and biological reaction. It improved the temperature prediction accuracy to ± 1.24 °C. Sedory and Stenstrom (1995) extended Talati and Stenstrom's steady-state model to a dynamic model, which could predict aerated basin temperature within ± 1 °C. Makinia et al. (2005) described the transport of heat in an ASR by using a one-dimensional, advection-dispersion equation, adopting Sedory and Stenstrom's expressions for most heat flux components except for solar radiation and biological reaction heat. Escalas-Cañellas et al. (2008) adapted the model of Sedory and Stenstrom to estimate the temperature of the aerated lagoon, with two modifications on solar radiation energy input and latent heat loss resulting from aeration.

The thermal balance model developed here was adapted from previously published models (Sedory & Stenstrom, 1995) for simplicity in calculations but without deterioration in accuracy. One main modification was incorporation of heat exchange through surface evaporation, convection and atmospheric radiation into the overall heat transfer from the slurry surface, where the overall heat transfer coefficient was determined using Gunneberg empirical formula (Gunneberg, 1978; Yu & Yu, 2009). A further modification was to calculate the heat of biological oxidation using the approach proposed by la Cour Jansen et al. (1992), which considered the heat released from biological processes including carbon oxidation, nitrification and de-nitrification, not only depending on organic substrate removal. A third modification was to estimate solar radiation energy using observed insolation data which could avoid complicated calculations using complex equations.

In this study, a detailed heat balance model was designed, as explained above, and applied to a conventional ASR, and its heat budget and temperature changes were simulated. The results were validated by comparison with measured thermal changes in actual full-scale reactor.

3.2. Materials and methods

3.2.1. Experimental site

This study was performed at a pig slurry treatment plant at Saga Prefecture Livestock Experiment Station, located 33°12′ North and 129°55′ East (Yamauchi-cho, Takeo city, Saga Prefecture, Japan). Using a continuous-flow ASP, this plant treated slurry from a pig-rearing facility with a breeding stock of more than 600 pigs. The daily slurry loading was about 13 m^3/d during the experimental period (September 2011).

As shown in Fig. 3.1, raw slurry in a storage tank was mechanically separated into liquid and solid phases by a screen, and the solid phase was transferred to a composting plant while the liquid phase was balanced. The liquid was then sent to a series of ASRs and aerated periodically. The aerobically treated slurry was then sent to a membrane filtration tank and other treatment units for advanced treatment.

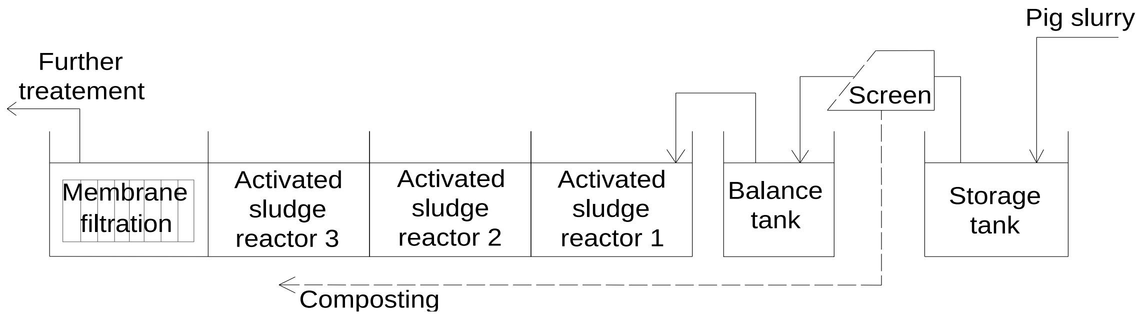


Fig. 3.1 Process flowchart of pig slurry treatment plant at Saga Prefecture Livestock Experiment Station

The design and operation data of the three in-line ASRs are shown in Table 3.1. These reactors were constructed from cement block and mortar, with a thermal conductivity of 0.546 W/m/℃ (JSCE, 1999). They were embedded in the ground to a depth of 1.5 m, while the upper part was exposed to air. The working volumes of the reactors are 32.8 m^3, 31 m^3 and 30 m^3. In each reactor, 15 m^3 of the volume was filled with porous ceramic (Bio-carrier SI type, Iwao Jiki, Saga, Japan) as a bio-carrier. Its porosity, density and specific heat are 77%, 340 kg/m^3 and 0.8 kJ/kg/℃, respectively. The diffused aeration system for the ASRs comprises a roots-type blower (BH65, Anlet, Aichi, Japan) placed outside the reactors and ceramic tube diffusers (Iwao Jiki, Saga, Japan) uniformly installed at the bottom of the reactors. Aeration was intermittent during experimental days, with the blower being alternately switched on for 2 h and off for the next 2 h. The working power was 2200 W with a mechanical efficiency of 81%. The blower could deliver 81 m^3/h of air which was supplied evenly to the 3 reactors through uniformly arranged ceramic tube diffusers, so that the air delivery rate for each reactor was 27 m^3/h.

Table 3.1 Design and operational specifications of ASRs

Design or operation parameters		ASRs		
		ASR1	ASR2	ASR3
Building material		Cement block with mortar		
Dimensions	Length	3275 mm	3200 mm	3200 mm
	Width	6550 mm	6550 mm	6550 mm
	Height	2020 mm	2020 mm	2020 mm
Wall thickness		450 mm	450 mm	450 mm
Working volume		32.8 m^3	31 m^3	30 m^3
Volume of bio-carrier		15 m^3	15 m^3	15 m^3
Diffused aeration system	Daily working hours	12 h/d		
	Working voltage	200 V		
	Working current	11 A		
	Mechanical efficiency	81%		
	Total air delivery rate	81 m^3/h		
	Air delivery rate for reactor	27 m^3/h	27 m^3/h	27 m^3/h

3.2.2. Slurry sampling and analysis

To investigate the performance of carbon oxidation, nitrogen nitrification and de-nitrification in the continuous-flow ASRs, multiple point samplings of raw slurry and aerobically treated slurry were taken from the storage tank and ASRs, respectively, on 14, 20, 27, September and 3 October 2011. The analysed parameters were COD, total organic carbon (TOC), TKN, NH_4-N and NO_x-N. COD was analysed using the dichromate reactor digestion method proposed by HACH Company. TOC was measured by the dichromate redox titration method (Skejemstad & Baldock, 2008). TKN, NH_4-N and NO_x-N were measured according to the testing methods published by FAMIC, 2008. The sum of the results of TKN and NO_x-N gives a value of TN.

3.2.3. Temperature monitoring and meteorological data collection

Onsite temperature measurement was performed from 20 to 30 September, 2011. The slurry temperatures in the storage tank, ASR1, ASR2 and ASR3 (T_{stt}, T_{ASR1}, T_{ASR2}, T_{ASR3}) were measured by Pt100 thermal resistor mounted in the liquid phase of each tank, and data were recorded by thermo-recorders (TR-81, T&D, Matsumoto, Japan) every 10 min. The ambient air temperature (T_{air}) and surrounding soil temperature (T_{soil}) were measured and recorded every 10 min by thermo-sensors (TR-0106, T&D) coupled with recorders (RT-11, ESPEC, Osaka, Japan), whose sensors were installed in an open cabin and in the soil (50 mm deep and 2000 mm from the reactor wall) around the reactors, respectively. The layout of the installation of the sensors is shown in Fig. 3.2, and the picture of a sensor mounted in an ASR is shown in Fig. 3.3.

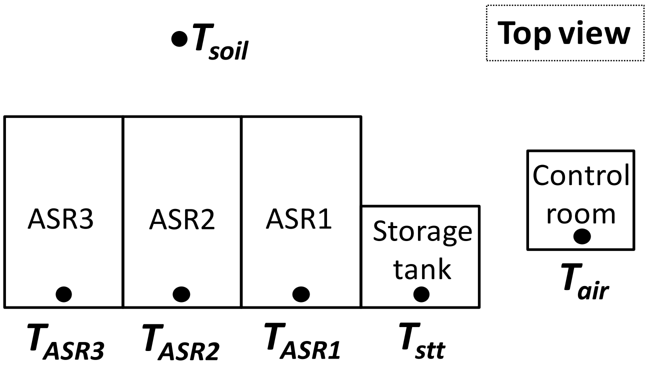


Fig. 3.2 Layout diagram for installation of temperature sensors (●)

Fig. 3.3 Picture of sensor installation in the liquid phase of an ASR

Meteorological data, such as mean daily values of wind velocity, ambient air relative humidity and daily global solar radiation were obtained from Saga local meteorological observatory (JMA, 2011).

3.2.4. Theoretical thermal balance

Estimation of theoretical thermal balance targeted ASR2, which is considered as a completely mixed tank. This implies that the net heat gain or loss (Q_{net}) is equivalent to the enthalpy change (ΔH) in ASR2.

$$Q_{net} = \Delta H \qquad (3.1)$$

For further model development, unifying the division of time interval on each time-dependent variable, such as slurry temperatures and heat exchanges, is essential. Assuming that the time interval is 1 day (from 00:00 one day to 00:00 the next day) and the unit for heat exchanges is MJ/d, the daily enthalpy change in ASR2, ΔH can be written as

$$\Delta H = C_{ASR2} \cdot \Delta T_{ASR2} \cdot 10^{-3} \quad (3.2)$$

$$C_{ASR2} = \rho_{sl} \cdot V_{sl} \cdot C_{p,sl} + \rho_{bc} \cdot V_{bc} \cdot C_{p,bc} \quad (3.3)$$

$$V_{sl} = V_{ASR2} - V_{bc} + V_{bc} \cdot p \quad (3.4)$$

where C_{ASR2} indicates the heat capacity of ASR2 (kJ/ ℃); ΔT_{ASR2} is the daily temperature change in ASR2 (℃), as the daily mean temperature difference with slurry the previous day; ρ_{sl} is the slurry density, considered equal to that of water (1000 kg/m^3); ρ_{bc} is the bio-carrier density (kg/m^3); V_{sl} is the slurry volume in ASR2 (m^3); V_{bc} is the bio-carrier filling volume in ASR2 (m^3); $C_{p,sl}$ is the specific heat of the slurry, considered equal to that of water (4.2 kJ/kg/ ℃); $C_{p,bc}$ is the specific heat of the bio-carrier (kJ/kg/ ℃); V_{ASR2} is the working volume of ASR2 (m^3); p is the bio-carrier porosity (%).

Q_{net}, the net heat gain or loss in ASR2, can be determined by Eq. 3.5 and 3.6, the relationship between each index is described in Fig. 3.4. It is the sum of the components that represent the heat exchanges through solar radiation, biological reaction, blower inefficiency, slurry flow, diffused air aeration, slurry surface and reactor wall. Note that the heat exchange occurring from rainfall is not considered in this study.

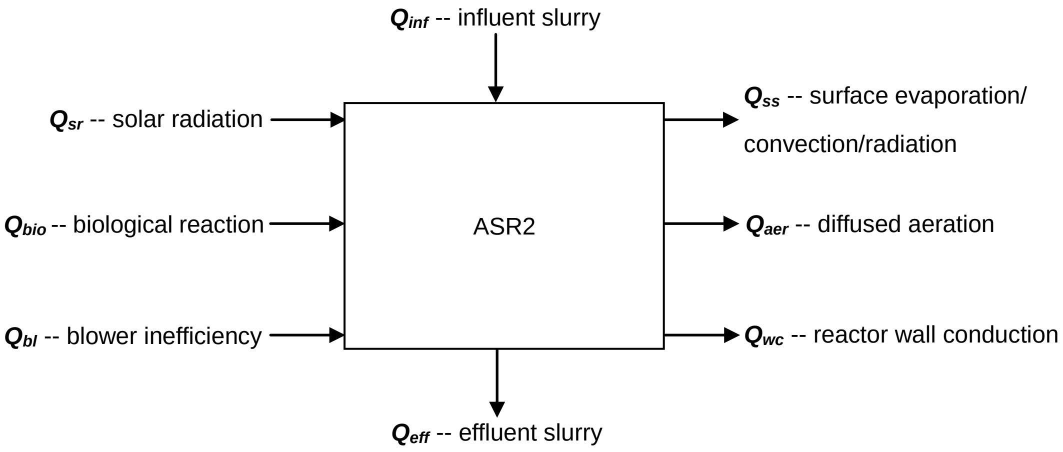


Fig. 3.4 Heat inputs and outputs of ASR2

$$Q_{net} = Q_{sr} + Q_{bio} + Q_{bl} + Q_{sf} + Q_{ss} + Q_{aer} + Q_{wc} \qquad (3.5)$$

$$Q_{sf} = Q_{inf} - Q_{eff} \qquad (3.6)$$

where Q_{sr} is the heat gained from solar radiation (MJ/d); Q_{bio} is the heat generated in biological reactions (MJ/d); Q_{bl} is the heat input due to blower inefficiency (MJ/d); Q_{sf} is the heat flux through slurry flow (MJ/d); Q_{inf} is the heat input with influent slurry (MJ/d) while Q_{eff} is the heat output with effluent slurry (MJ/d); Q_{ss} is the overall heat transfer from slurry surface (MJ/d); Q_{aer} is the heat exchange by diffused aeration (MJ/d); Q_{wc} is the heat transfer from reactor wall (MJ/d).

Q_{sr} was estimated by

$$Q_{sr} = \phi_{sr} \cdot A_{ss} \cdot (1 - \alpha) \qquad (3.7)$$

where Φ_{sr} is the daily global solar radiation (MJ/m^2/d); A_{ss} is the slurry surface area of ASR2 (m^2); α is the albedo of the slurry, dimensionless, assumed equal to that of municipal wastewater, 0.2 (Vassallo et al., 2009).

Q_{bio} was determined by the following equation proposed by la Cour Jansen et al. (1992)

$$Q_{bio} = -(\Delta G_1 \cdot \frac{\Delta COD}{32} + \Delta G_2 \cdot \frac{8 \cdot \Delta TN}{14} + \Delta G_3 \cdot \frac{5 \cdot 0.8 \cdot \Delta TN}{14}) \cdot q_{sl} \cdot 10^{-3} \quad (3.8)$$

where ΔG_1 is Gibb's free energy for aerobic respiration (-110 kJ/mol); ΔG_2 is Gibb's free energy for nitrification (-43 kJ/mol); ΔG_3 is Gibb's free energy for de-nitrification (-104 kJ/mol); ΔCOD is the removal of COD (mg/L), calculated as the difference in the levels of COD concentrations in raw slurry and aerobically treated slurry; ΔTN is the removal of TN (mg/L), determined as the difference in the levels of TN concentrations in raw slurry and aerobically treated slurry; q_{sl} is the daily slurry loading (m^3/d).

Q_{bl} was estimated using

$$Q_{bl} = 3.6 \times 10^{-3} \cdot t \cdot U \cdot I \cdot (1 - \eta) \quad (3.9)$$

where t is aerating hours per day (h/d); U is the blower working voltage (V); I is the blower working current (A); η is the blower mechanical efficiency (%).

Q_{sf} was estimated as

$$\begin{aligned} Q_{sf} &= Q_{inf} - Q_{eff} \\ &= \rho_{sl} \cdot q_{sl} \cdot C_{p,sl} \cdot (T_{ASR1} - T_{ASR2}) \cdot 10^{-3} \quad (3.10) \end{aligned}$$

where T_{ASR1} is the daily mean slurry temperature in ASR1 (℃); T_{ASR2} is the daily mean slurry temperature in ASR2 (℃).

Q_{ss}, accounting for slurry surface evaporation, convection and radiation heat, was estimated as

$$Q_{ss} = 8.64 \times 10^{-2} \cdot K_s \cdot (T_{air} - T_{ASR2}) \cdot A_{ss} \quad (3.11)$$

$$K_s = 4 \cdot \varepsilon \cdot \sigma \cdot \left(T_{ASR2} + 273.15\right)^3 + \left(1.5 \times 10^{-3} + 1.12 \times 10^{-3} \cdot W\right) \cdot \left[\left(2501.7 - 2.366 \cdot T_{ASR2}\right) \cdot \frac{25509}{\left(T_{ASR2} + 239.7\right)^2} \cdot 10^{\frac{7.56 T_{ASR2}}{T_{ASR2} + 239.7}} + 1621\right] \quad (3.12)$$

where K_s is the overall heat transfer coefficient from slurry surface (W/m^2/ ℃), determined by the Gunneberg empirical formula (Eq. 3.12; Gunneberg, 1978); ε is the slurry surface emissivity, dimensionless, considered as equal as water surface emissivity, 0.97; σ is Stefan Boltzman constant (5.67×10^{-8} W/m^2/ ℃^{4}); W is the daily mean wind velocity (m/s); T_{air} is the daily mean ambient air temperature (℃).

Q_{aer} consists of two components: sensible heat exchange in the form of convection (Q_{as}) and latent heat exchange in the form of evaporation (Q_{al}). They were estimated as

$$Q_{aer} = Q_{as} + Q_{al} \quad (3.13)$$

$$Q_{as} = \rho_{air} \cdot q_{air} \cdot t \cdot C_{p,air} \cdot \left(T_{ia} - T_{ea}\right) \cdot 10^{-3} \quad (3.14)$$

$$Q_{al} = \frac{M_{H_2O} \cdot q_{air} \cdot t}{10^6 R} \cdot \left\{ \frac{P_{sat(ia)} \cdot RH}{\left(T_{ia} + 273.15\right)} - \frac{P_{sat(ea)} \cdot \left[RH + h_f \cdot \left(1 - RH\right)\right]}{\left(T_{ea} + 273.15\right)} \right\} \cdot H_{vap}(T_{ASR2}) \quad (3.15)$$

$$P_{sat(xa)} = 10^{\lg\frac{101325}{760} + 8.07131 - \frac{1730.63}{233.426 + T_{xa}}} \quad (3.16)$$

$$H_{vap}(T_{ASR2}) = H_{vap}(100) - C_{p(H_2O(l))}(T_{ASR2} - 100) - C_{p(H_2O(g))}(100 - T_{ASR2}) \quad (3.17)$$

where ρ_{air} is the air density (1.293 kg/m^3); q_{air} is the rate of air delivery to ASR2 (m^3/h); $C_{p,air}$ is the specific heat of air (1.0 kJ/kg/ ℃); T_{ia} is the daily mean inflow-air temperature (℃), considered equal to T_{air}; T_{ea} is the daily mean exit-air temperature (℃), assumed as equal as T_{ASR2}; M_{H_2O} is the water molecular weight (18 g/mol); R is universal gas constant (8.314

m^3 ·Pa/mol/ ℃); $P_{sat(ia)}$ and $P_{sat(ea)}$ are the inflow-air and exit-air saturation vapour pressure (Pa), respectively, determined by Antoine equation, i.e. Eq. 3.16; *RH* is the daily mean ambient air relative humidity (%); h_f is exit-air humidity factor, dimensionless, assumed as 1.0 for subsurface aeration system; $H_{vap}(T_{ASR2})$ is the vaporisation heat at certain reactor temperature (kJ/kg H_2O); $H_{vap}(100)$ is the vaporisation heat at 100 ^oC (2263 kJ/kg H_2O); $C_{p(H_2O(l))}$ and $C_{p(H_2O(g))}$ are the specific heat of water (4.2 kJ/kg/ ℃) and water vapour (1.84 kJ/kg/ ℃), respectively.

Q_{wc} is the sum of heat conduction from ASR2 to ASR1 (Q_{21}) and to ASR3 (Q_{23}) as well as to the surrounding soil (Q_{2s}). They were described by

$$Q_{wc} = Q_{21} + Q_{23} + Q_{2s} \qquad (3.18)$$

$$Q_{21} = 8.64 \times 10^{-2} \cdot \frac{\left(T_{ASR1} - T_{ASR2}\right)}{S_{21}},\ S_{21} = \frac{L}{k_{wall} \cdot A_{21}} \qquad (3.19)$$

$$Q_{23} = 8.64 \times 10^{-2} \cdot \frac{\left(T_{ASR3} - T_{ASR2}\right)}{S_{23}},\ S_{23} = \frac{L}{k_{wall} \cdot A_{23}} \qquad (3.20)$$

$$Q_{2s} = 8.64 \times 10^{-2} \cdot \frac{\left(T_{soil} - T_{ASR2}\right)}{S_{2s}},\ S_{2s} = \frac{L}{k_{wall} \cdot A_{2s}} + \frac{D}{k_{soil} \cdot A_{2s}} \qquad (3.21)$$

where T_{ASR3} is the daily mean slurry temperature in ASR3 (℃); T_{soil} is the daily mean surrounding soil temperature (℃); S_{21} or S_{23} is the conduction resistance of reactor wall (℃/W); S_{2s} is the sum of conduction resistances of reactor wall and surrounding soil (℃/W); *L* is the thickness of reactor wall (m); *D* is the distance of soil temperature measurement point to the reactor wall (m); k_{wall} is the thermal conductivity of reactor wall (W/m/ ℃); k_{soil} is the thermal conductivity of soil (W/m/ ℃); A_{21} is the area of the wall between ASR2 and ASR1 (m^2); A_{23} is the area of the wall between ASR2 and ASR3 (m^2); A_{2s} is the area exposed to the soil for ASR2 (m^2).

The data required for calculation are summarised in Table 3.2.

Table 3.2 Data collection and processing for thermal balance analysis

Term	Eq.	Symbol	Value	Source
ΔH	3.2	C_{ASR2}	119790 kJ/℃	Calculation
		ΔT_{ASR2}	Variable, ℃	Measurement data
	3.3	ρ_{sl}	1000 kg/m^3	Approximate estimation
		$C_{p,sl}$	4.2 kJ/kg/℃	Approximate estimation
		ρ_{bc}	340 kg/m^3	Design data
		$C_{p,bc}$	0.8 kJ/kg/℃	Design data
		V_{sl}	27.55 m^3	Design data
		V_{bc}	15 m^3	Design data
	3.4	V_{ASR2}	31 m^3	Design data
		p	77 %	Design data
Q_{sr}	3.7	A_{ss}	20.96 m^2	Design data
		Φ_{sr}	Variable, MJ/m^2/d	Meteorological data
		α	0.2	Vassallo et al., 2009
Q_{bio}	3.8	ΔG_1	-110 kJ/mol	la Cour Jansen et al. (1992)
		ΔG_2	-43 kJ/mol	
		ΔG_3	-104 kJ/mol	
		ΔCOD	See analysis results	Wastewater analysis
		ΔTN	See analysis results	Wastewater analysis
		q_{sl}	Variable, m^3/d	Design data
Q_{bl}	3.9	t	12 h/d	Design data
		U	200 V	Design data
		I	11 A	Design data
		η	81 %	Design data
Q_{sf} Q_{inf}	3.10	T_{ASR1}	Variable, ℃	Measurement data
Q_{eff}		T_{ASR2}	Variable, ℃	Measurement data
Q_{ss}	3.11	T_{air}	Variable, ℃	Measurement data
		K_s	Variable, W/m^2/℃	Gunneberg equation
	3.12	W	Variable, m/s	Meteorological data
		ε	0.97	Theoretical value
		σ	5.67×10^{-8} W/m^2/℃^{4}	Theoretical value
Q_{aer} Q_{as}	3.14	ρ_{air}	1.293 kg/m^3	Theoretical value
		$C_{p,air}$	1.0 kJ/kg/℃	Theoretical value
		q_{air}	27 m^3/h	Design data
		T_{ia}	Variable, ℃	Assumed equal to T_{air}
		T_{ea}	Variable, ℃	Assumed equal to T_{ASR2}

	Q_{al}	3.15	M_{H_2O}	18 g/mol	Theoretical value
			R	8.314 m^3·Pa/mol/℃	Theoretical value
			RH	Variable, %	Meteorological data
			h_f	1	Talati & Stenstrom, 1990
		3.16	$P_{sat(ia)}$	Variable, Pa	Antoine equation
			$P_{sat(ea)}$	Variable, Pa	
		3.17	$H_{vap}(100)$	2263 kJ/kg H_2O	Theoretical value
			$C_{p(H_2O(l))}$	4.2 kJ/kg/℃	Theoretical value
			$C_{p(H_2O(g))}$	1.84 kJ/kg/℃	Theoretical value
			$H_{vap}(T_{ASR2})$	Variable, kJ/kg H_2O	Calculation
Q_{wc}	Q_{21}	3.19	S_{21}	0.062 ℃/W	Calculation
			k_{wall}	0.546 W/m/℃	JSCE, 1999
			L	0.45 m	Design data
			A_{21}	13.231 m^2	Design data
	Q_{23}	3.20	A_{23}	13.231 m^2	Design data
			S_{23}	0.062 ℃/W	Calculation
			T_{ASR3}	Variable, ℃	Measurement data
	Q_{2s}	3.21	T_{soil}	Variable, ℃	Measurement data
			S_{2s}	0.122 ℃/W	Calculation
			k_{soil}	0.686 W/m/℃	JSCE, 1999
			D	2.0 m	Experimental design
			A_{2s}	30.56 m^2	Design data

3.2.5. Model calculation and validation

The procedures for model calculation and validation are shown in Fig. 3.5. After data collection, the daily heat exchanges were determined by Eq. 3.5~3.21. Assuming that the daily enthalpy change ΔH equals Q_{net}, the daily temperature change ΔT_{ASR2} was estimated from Eq. 3.2~3.4, and T_{ASR2} was predicted by giving a slurry temperature of previous day. By comparing the estimated ΔT_{ASR2} and T_{ASR2} to the measured ΔT_{ASR2} and T_{ASR2}, the model was validated.

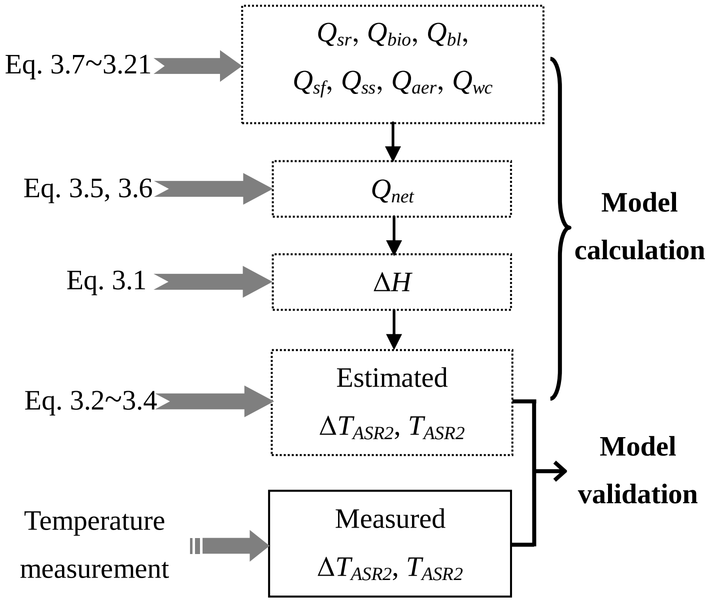


Fig. 3.5 Flowchart of model calculation and validation

3.3. Results and discussion

3.3.1. Slurry characterisation

Table 3.3 shows the average levels and variation ranges of COD, TOC, TKN, NH_4-N, NO_x-N and TN for raw slurry from the storage tank and treated slurry sampled from the ASRs during the experimental period. By comparing the treated slurry qualities with those of raw slurry, an obvious decrease in the organic substrate and nitrogen source was revealed. COD and TOC were reduced by 71% and 84%, respectively, and the removal efficiency of TKN and NH_4-N was better than 90%. This implied that the biological processes in the ASRs such as carbon oxidation, nitrogen nitrification and de-nitrification were performing well during our experimental period.

Table 3.3 Characteristics of raw slurry and aerobically treated slurry during the experimental period

Slurry samples	COD (mg/L)	TOC (mg/L)	TKN (mg/L)	NH_4-N (mg/L)	NO_x-N (mg/L)	TN (mg/L)
Raw	311.1 ± 73.3	167.7 ± 52.9	374.70 ± 31.64	339.23 ± 18.41	11.12 ± 7.40	385.82 ± 35.24
Treated	90.8 ± 41.0	27.0 ± 21.2	22.71 ± 33.26	17.01 ± 18.11	67.28 ± 26.29	89.99 ± 38.23
Difference *	220.3	140.7	351.99	322.22	-56.16	295.83

* difference between raw slurry and aerobically treated slurry

3.3.2. Characterisation of state variables

To complete the model, several state variables were prepared, such as daily slurry loading, several physicochemical parameters, process performance parameters and meteorological conditions. The results of these variables are shown in Table 3.4. A comparison of measured temperature data from 20 to 30 September between rainy and fine days showed that T_{soil} and T_{air} on rainy days were relatively high, and the slurry temperatures in the ASRs (T_{ASR1}, T_{ASR2}, T_{ASR3}) and the storage tank (T_{stt}) might be affected by the surroundings, thus these slurry temperatures were also higher on rainy days. In addition, by comparing the slurry temperature in ASRs with that in the storage tank, we found that the temperature in the ASRs was higher for most of the days. This can probably be attributed to the exothermic biological reactions involved in the ASRs, since other ambient conditions including solar radiation were the same for these tanks. Moreover, T_{ASR2} was higher than T_{ASR1}, T_{ASR3}. This was possibly attributed to the position of the reactor.

Table 3.4 Investigated results on plant operation, physicochemical properties, process performance and weather conditions for thermal balance analysis

Date (day/ month)	Weather	q_{sl} (m^3/d) [a]	K_s (W/m^2/ ℃) [b]	$P_{sat(ia)}$ (Pa) [b]	$P_{sat(ea)}$ (Pa) [b]	$H_{vap}(T_{ASR2})$ (kJ/kg H_2O) [b]	T_{stt} (℃) [c]	T_{ASR1} (℃) [c]	T_{ASR2} (℃) [c]	T_{ASR3} (℃) [c]	T_{air} (℃) [c]	T_{soil} (℃) [c]	Φ_{sr} (MJ/m^2/d) [d]	W (m/s) [d]	RH (%) [d]
20/9	Rainy	12.984	40.093	2591.198	3556.243	2435	22.55	23.97	27.01	21.84	21.73	28.54	5.82	3.2	81
21/9	Rainy	12.995	43.399	2251.836	3295.509	2438	21.18	22.06	25.72	20.10	19.45	31.09	5.55	3.9	72
22/9	Fine	12.957	36.281	2171.741	3302.714	2438	19.56	21.61	25.75	20.00	18.87	29.75	22.27	2.9	58
23/9	Fine	12.968	25.182	2078.919	3219.954	2439	18.31	21.32	25.33	19.53	18.18	25.88	22.71	1.4	63
24/9	Fine	12.989	24.935	2109.283	3160.023	2440	19.02	20.94	25.01	19.13	18.41	24.56	22.48	1.4	56
25/9	Fine	12.976	22.796	2273.756	3147.476	2440	19.13	20.89	24.94	19.05	19.61	24.28	13.53	1.1	59
26/9	Fine	12.787	28.130	2539.338	3246.801	2439	20.40	21.53	25.47	19.66	21.40	25.06	15.00	1.8	65
27/9	Fine	12.993	33.660	2640.953	3346.916	2438	20.24	22.20	25.98	20.26	22.04	25.73	14.90	2.5	69
28/9	Fine	12.979	27.691	2824.210	3471.384	2436	20.71	22.98	26.60	21.04	23.14	26.23	17.87	1.6	69
29/9	Rainy	12.959	27.939	2806.213	3902.478	2432	21.90	25.11	28.60	23.29	23.04	27.98	9.84	1.4	79
30/9	Rainy	12.991	33.290	2465.302	3852.735	2432	22.48	25.25	28.38	23.11	20.92	31.24	3.83	2.1	82
Mean	-	12.962	31.218	2432.068	3409.294	2437	20.50	22.53	26.25	20.63	20.61	27.30	13.98	2.1	68
Max.	-	12.995	43.399	2824.210	3902.478	2440	22.55	25.25	28.60	23.29	23.14	31.24	22.71	3.9	82
Min	-	12.787	22.796	2078.919	3147.476	2432	18.31	20.94	24.94	19.05	18.18	24.28	3.83	1.1	56
SD [e]	-	0.059	6.653	270.101	262.003	2.90	1.43	1.58	1.26	1.50	1.80	2.55	7.02	0.9	9

[a] plant information, [b] estimated values for some physicochemical parameters,
[c] measured temperature behaviour, [d] investigated meteorological data,
[e] standard deviation

3.3.3. Thermal balance analysis

3.3.3.1. Simulation of heat exchanges

Since the presented model did not account for the heat exchange from rainfall, the model calculation and validation were carried out during days without precipitation from 22 to 28 September. The simulated heat input (Q_{sr}, Q_{bio}, Q_{bl}), heat output (Q_{sf}, Q_{ss}, Q_{aer}, Q_{wc}) and net heat change (Q_{net}) for ASR2 are shown in Table 3.5. The table shows that Q_{sr} was the largest energy source to the reactor, and the second largest source was Q_{bio}. Energy loss mainly occurred through Q_{ss}, and then Q_{sf}. The other heat exchanges were relatively small. These results bear some similarity to those reported by Makinia et al. (2005).

Table 3.5 Simulated heat exchanges for ASR2 by the developed model (MJ/d)

Date (day/month)	Q_{sr} ^{i}	Q_{bio} ^{i}	Q_{bl} ^{i}	Q_{sf} $^{\circ}$	Q_{ss} $^{\circ}$	Q_{aer} $^{\circ}$	Q_{wc} $^{\circ}$	Total input	Total output	Q_{net}
22/9	373.423	217.893	18.058	−225.387	−451.977	−14.403	−10.907	609.374	−702.674	−93.300
23/9	380.801	218.078	18.058	−218.278	−325.969	−13.761	−13.210	616.937	−571.218	45.719
24/9	376.945	218.431	18.058	−221.890	−298.119	−13.972	−14.119	613.434	−548.100	65.334
25/9	226.871	218.213	18.058	−221.100	−220.228	−12.464	−14.276	463.142	−468.068	−4.926
26/9	251.520	215.034	18.058	−211.577	−207.273	−10.719	−13.797	484.612	−443.366	41.246
27/9	249.843	218.498	18.058	−206.231	−240.143	−10.227	−13.346	486.399	−469.947	16.452
28/9	299.644	218.263	18.058	−197.189	−173.217	−9.999	−12.977	535.965	−393.382	142.583
Mean	308.435	217.773	18.058	−214.522	−273.847	−12.221	−13.233	544.266	−513.822	30.444

^{i} heat input, $^{\circ}$heat output

3.3.3.2. Simulation of temperature changes

From Eq. 3.2, based on the daily total heat input or output in Table 3.5, the temperature increment or decrement in the reactor resulted from the heat gain or loss was estimated, and the results are shown in Table 3.6. It shows that the total heat gain ranged from 460 MJ/d to 620 MJ/d which would produce a temperature increase of 3.87–5.15 ^{o}C/d (mean 4.54 ^{o}C/d), whereas a total heat loss of the same magnitude would result in a decrease at the same level. Thus it is important to reduce the heat loss for operating ASP at higher temperatures.

The above results indicated that the largest heat loss was Q_{ss}, which was positively correlated to the temperature difference between the slurry and air, wind velocity and slurry surface area (see Eq. 3.11 and 3.12). To reduce Q_{ss}, covering ASRs by a greenhouse, vinyl sheeting or fibreglass panels can be one of effective and practical methods, because they can guard against wind exposure, increase the ambient air temperature, and the temperature difference between the slurry and air can be reduced. Wakiya and Sakai (2002) reported that reactor temperature was raised by 3.8 ℃ by covering it with a plastic tent during the winter. In addition, an underground tank is also a useful thermal insulation method, which protects the reactor wall from exposure to the flowing air, and thus the convective heat transfer from the wall to the air can be reduced.

Table 3.6 Simulated contributions to temperature changes in the ASR2 by heat input or heat output

Date (day/month)	Increment caused by heat input (℃)	Decrement caused by heat output (℃)
22/9	5.09	−5.87
23/9	5.15	−4.77
24/9	5.12	−4.58
25/9	3.87	−3.91
26/9	4.05	−3.70
27/9	4.06	−3.92
28/9	4.47	−3.28
Mean	4.54	−4.29

3.3.4. Model validation

A comparison between the estimated T_{ASR2}, ΔT_{ASR2} and measured T_{ASR2}, ΔT_{ASR2} which was made for model validation, is shown in Table 3.7. The estimated and measured T_{ASR2} from 22 to 28 September were plotted in Fig. 3.6, which indicated that the estimated T_{ASR2} by this model agreed well with the measured T_{ASR2}. The model estimated reactor temperatures with a root mean square error (RMSE) of 0.60 ^{o}C, which is smaller than that reported by Talati and Stenstrom (1990) or Sedory and Stenstorm (1995), and comparable to that reported in the work of Escalas-Cañellas et al. (2008). Besides, the RMSE of 0.60 ^{o}C was judged applicable in simulating the temperature increment of (4.54 ± 0.57) ^{o}C by heat input or the temperature decrement of (4.29 ± 0.86) ^{o}C by heat output. On the other hand, the relative root mean square error (R-RMSE) in estimating T_{ASR2} from 24.94 ℃ to 27.17 ℃ was about 2.3%. It indicated that the model is suitable as an analytical tool for simulating heat exchanges and predicting ASR temperatures for process design and control.

Table 3.7 Comparison between estimated T_{ASR2}, ΔT_{ASR2} and measured T_{ASR2}, ΔT_{ASR2} (℃)

Date (day/month)	Measured T_{ASR2}	Estimated T_{ASR2}	Estimated ΔT_{ASR2}	Measured ΔT_{ASR2}	Difference [b]
21/9 [a]	25.72				
22/9	25.75	24.94	-0.78	0.03	0.81
23/9	25.33	26.13	0.38	-0.42	-0.80
24/9	25.01	25.88	0.55	-0.32	-0.87
25/9	24.94	24.97	-0.04	-0.07	-0.03
26/9	25.47	25.28	0.34	0.53	0.19
27/9	25.98	25.61	0.14	0.51	0.37
28/9	26.60	27.17	1.19	0.62	-0.57

[a] T_{ASR2} at 25.72 °C on 21/9 was set as initial temperature.

[b] difference was calculated by the subtraction of estimated values from measured data.

RMSE between estimation and measurement was calculated at 0.60 ℃.

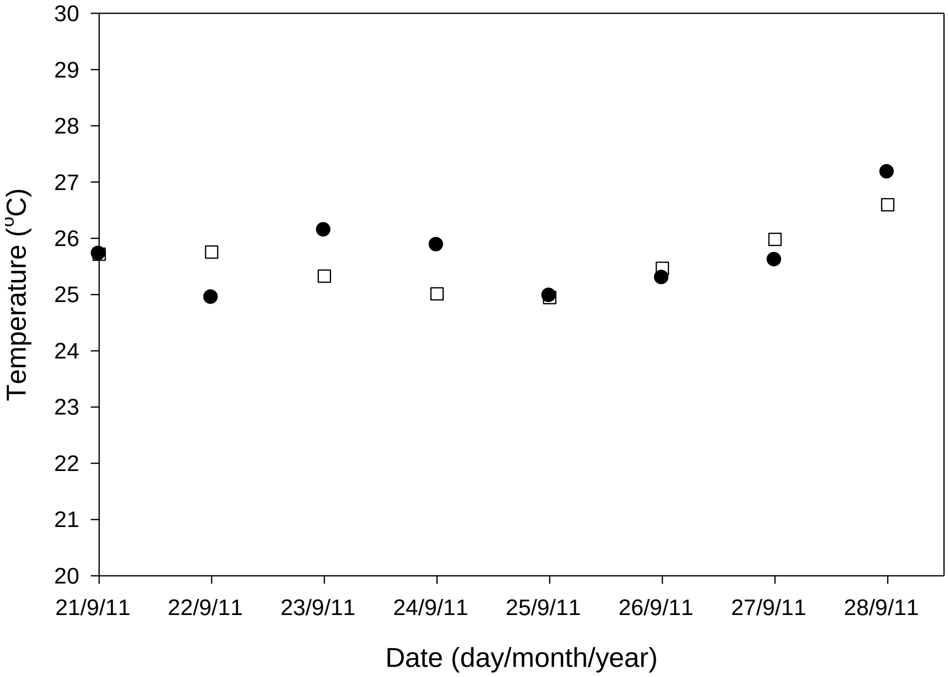


Fig. 3.6 Comparison between estimated (●) and measured (□) temperatures in ASR2

3.4. Conclusions

A heat balance model for application to ASRs was developed and validated to simulate thermal changes in the reactor. The model estimation results matched in situ temperatures data with a RMSE of 0.60 ℃, and this model can be used as an analytical tool for ASP design and modification. The simulation results showed that heat was mostly gained from solar radiation and biological reaction, while heat was mainly lost through the slurry surface and slurry flow. The model indicated that the total heat gain resulted in a rate of temperature increase of 4.54 ℃/d in the reactor, but this effect was offset by the heat loss. To operate an ASP at higher temperatures, it is recommended to insulate the system either by placing it in a greenhouse, or covering the slurry surface with vinyl sheeting or fibreglass panels—a method that can be easily installed in existing facilities. Moreover, using an underground tank is useful when implementing an ASP.

Chapter 4

Thermal balance analysis of autothermal thermophilic aerobic treatment of human excreta

4.1. Introduction

Human excreta contains high percentages of N, P and K. According to the principles of sustainable development, human excreta should be recycled for use as fertilisers in agriculture in order to properly utilise their nutrients, instead of simply being subject to removal by wastewater treatment systems. The recirculation of such nutrients in agriculture would also decrease the consumption of chemical fertilisers (Tidåker et al., 2007). However, direct land application should be avoided because of concerns over the potential risks of groundwater contamination and pathogen dissemination (Esrey et al., 2001). In addition, easily decomposable organic matter in raw human excreta can cause physiological disorders in plants (Inbar et al., 1993). Therefore, the conversion of human excreta into a hygienic and stable fertiliser is a major concern.

In Japan, many recycling facilities convert organic waste such as livestock manure, food waste and crop residue into compost, feedstuff or bioenergy (JIE, 2008). However in Japan, human excreta is seldom reused, because most of it is presently being purified at centralised treatment facilities or treated by on-site wastewater treatment systems (johkasou in Japanese) (Gaulke, 2006). One particular, rare case can be seen in Chikujo town, Fukuoka Prefecture, Japan, where human excreta with johkasou sludge are co-digested by ATAT and turned into a hygienic and stable liquid fertiliser for crops. Annually, a total of 9000 t of liquid fertiliser is produced and returned to the 20.8 km^2 farmland in this town.

ATAT is operated on the same principle as solid composting: it utilises the heat released from microbial aerobic metabolism to raise the temperature in an insulated system to a thermophilic range. By this process, biodegradable organics are reduced and pathogens trapped in the raw materials are destroyed at high temperatures (Juteau, 2006). Metcalf and Eddy (2003) stated that aerobic respiration, biosynthesis and endogenous respiration are involved in the aerobic stabilisation process. McCarty (1972) found that the heat released from aerobic respiration and synthesis reactions was nearly proportional to the consumption of organic substrates, estimated as 14.6 kJ by consuming 1 g of COD.

ATAT is also called liquid composting because the water content in the feed is generally higher than 90% (Layden et al., 2007). For instance, the water content of collected human excreta in the Chikujo ATAT facility is about 99%. Such liquid organic waste mostly contains water; hence, its energy content is generally lower compared to that of solid or semisolid organic waste. Moreover, water evaporation during the ATAT removes large amounts of heat. As a result, self-heating to thermophilic levels in the ATAT of liquid organic waste is difficult to achieve, and thus relatively few full-scale ATAT facilities are currently in operation (Juteau, 2006).

Self-heating is a primary criterion when designing or maintaining an ideal ATAT system. To this end, thermal balance analysis is important for understanding reactor thermodynamics and providing proper process control. Previous studies presented several heat balance models: Messenger et al. (1990) developed a mathematical model to simulate the heat balance in aerobic reactors; Gomez et al. (2007) developed a dynamic energy model for ATAT to predict the process temperature. Subsequently, Rojas et al. (2010) simplified the Gomez et al. energy model for improved temperature simulation and process optimisation. These models were primarily developed for thermophilic aerobic sludge digestion in semi-batch processes and seldom validated or applied to a full-scale ATAT reactor. Although the thermodynamic processes were explored through virtual scenarios, the principle of autothermal thermophilic operation was only explored qualitatively. Moreover, energetics of the ATAT of human excreta has not been studied in detail.

Toward the quantitative interpretation of autothermal thermophilic operation, this study aims to establish a detailed heat balance model for a batch ATAT of human excreta digestion, validate the model reproducibility by simulating the heat transfer process in a full-scale ATAT reactor and determine the specific biological heat yield from degrading 1 g COD of human excreta.

4.2. Materials and methods

4.2.1. Site description

4.2.1.1 Facility profile

The target ATAT facility is located in Chikujo town, Fukuoka Prefecture, Japan. The process flowchart for this ATAT facility is shown in Fig. 4.1. On each workday, about 30 t of human excreta and johkasou sludge (raw materials) are transported to the facility by a vacuum car. After the raw materials are fed into the first separator, foreign materials (stone, plastic, metal etc.) are removed and the raw slurry is sent to a balance tank for enzymatic hydrolysis. After further solid–liquid separation, the slurry is pumped and stored in the feed storage tank until the next batch of raw materials is fed into the ATAT reactor. In the reactor, the slurry is aerated for three to four weeks. The final product from the ATAT reactor is stored in the fertiliser storage tank for farmland application. The picture of the facility is shown in Fig. 4.2.

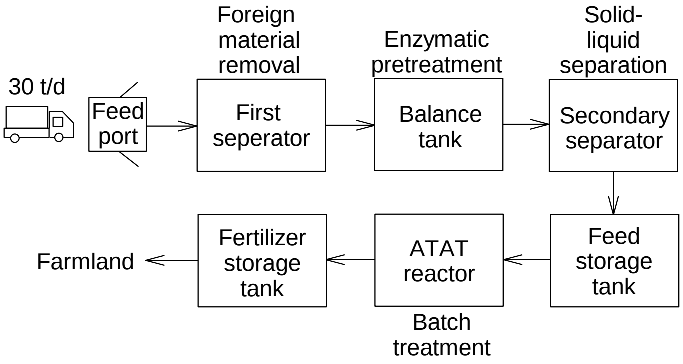


Fig. 4.1 Process flowchart of the Chikujo ATAT facility

Fig. 4.2 Picture of the Chikujo ATAT facility

4.2.1.2 ATAT reactor

A schematic diagram of the cross-section of the ATAT reactor is shown in Fig. 4.3. This semi-underground reactor covered above ground by soil, is made of concrete, and has a working volume of 150 m^3. It has a wall thickness of 300 mm, and the external surface area in contact with the soil is 220.6 m^2. Two alternately running aerators bring about simultaneous aeration and mixing in the reactor with each aerator working 12 h per day. The working power is adjusted to 6.6 kW and the airflow rate is approximately 120 m^3/h. The two aerators are completely submerged in the slurry; hence, all the heat from the motors is transferred to the slurry. A foam cutter with its motor outside the reactor is installed in the upper part of the reactor.

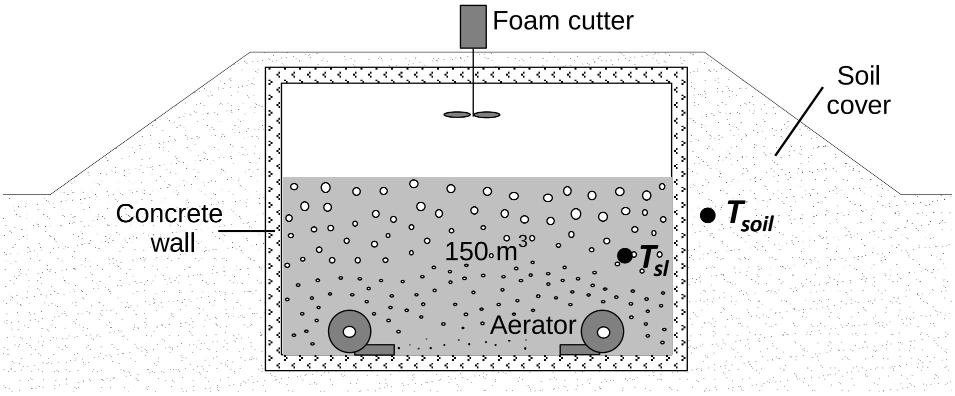


Fig. 4.3 Diagram of the ATAT reactor (●: location of temperature measurements)

To avoid short circuit and contamination of the treated product by the untreated feed, the ATAT reactor is operated in a batch mode consisting of a sequence of discharging, feeding and aeration. The steps of discharging and feeding by pumps require approximately 10 h in total. During these steps, the entire treated product of the last batch is removed; following this, 150 t of untreated feed is introduced. Subsequently, the untreated product is aerated. The total processing time for the reported batch was 23 days, which was the same as the designed value.

4.2.2. Chemical analysis

To determine ATAT batch performance, slurry samples were taken from the liquid phase of the reactor on 2, 8, 17 and 25 March 2011 by temporarily stopping aeration. Chemical analyses were performed on the mixed liquid state within 48 h after sampling. Each sample was homogenised for 2 min with a blender (TESCOM, Japan) before starting analysis. COD analysis was performed by the Hach dichromate reactor digestion method. Total solids (TS), VS and TKN were measured in accordance with the standard methods recommended by APHA (1999). NH_4-N and NO_x-N were determined by steam distillation methods proposed by Bremner and Keeney (1965).

4.2.3. Temperature measurement

Batch temperatures were measured from 2 to 25 March 2011. The slurry temperature in the ATAT reactor (T_{sl}) and the raw slurry temperature in the feed storage tank (T_{feed}) were measured by Pt100 thermocouples inserted in the liquid phase. The soil temperature around the ATAT reactor (T_{soil}) was measured using an underground Pt100 thermocouple buried 1000 mm deep and 250 mm away from the reactor wall (measurement locations for T_{sl} and T_{soil} are indicated in Fig. 4.3). Those temperatures were recorded by thermo recorders (TR-81, T&D) every 30 min. The ambient air temperature and relative humidity were also measured by a thermo-hydro sensor (TR-3110, T&D) coupled with a recorder (TR72U, T&D) installed in the facility's garage.

4.2.4. Theoretical calculations

Based on the first law of thermodynamics, the heat generated in the reactor is equal to the heat lost and the heat transformed in the reactor. This relationship is shown in Fig. 4.4. The heat generated from biological reactions in the reactor (Q_{bio}) can be estimated by Eq. 4.1,

$$Q_{bio} + Q_{mec} - Q_{loss} = Q_{sl} \qquad (4.1)$$

where Q_{sl} is the heat or enthalpy change of the slurry in the reactor, Q_{loss} is the heat loss from the reactor and Q_{mec} is the mechanical heat input by the submerged aerators.

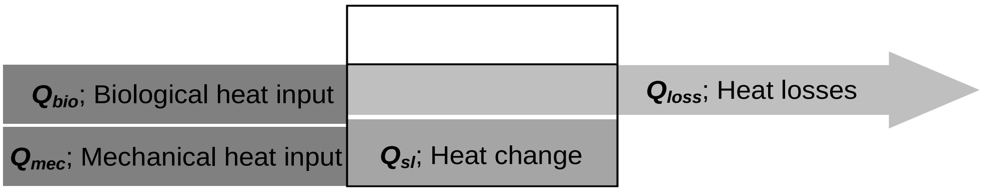


Fig. 4.4 Heat transfer in the ATAT reactor

Q_{sl} can be calculated as follows,

$$Q_{sl} = m_{sl}\, C_{p,sl}\, \mathrm{d}T_{sl}/\mathrm{d}t \qquad (4.2)$$

where m_{sl} is the mass of the fed slurry; assumed constant during aeration, $C_{p,sl}$ is the specific heat of the slurry; considered equal to that of water and dT_{sl}/dt is the temperature change rate of the slurry; assuming that T_{sl} is uniform throughout.

Q_{loss} is treated as the sum of the sensible heat loss with airflow (Q_{air}), the latent heat loss by evaporation (Q_{vap}) and the heat conduction through the reactor wall and surrounding soil (Q_{con}):

$$Q_{loss} = Q_{air} + Q_{vap} + Q_{con} \qquad (4.3)$$

Q_{air} is estimated as,

$$Q_{air} = \rho_{air} \cdot q_{air} \cdot C_{p,air} \cdot (T_{ea} - T_{ia}) \qquad (4.4)$$

where ρ_{air} is the density of air, q_{air} is the designed airflow rate, $C_{p,air}$ is the specific heat of air, T_{ia} is the inflow air temperature, considered equal to the ambient air temperature and T_{ea} is the exit air temperature, assumed to be equal to the slurry temperature.

Q_{vap} is given by the following equation;

$$Q_{vap} = m_{vap} \cdot H_{vap}(T_{sl}) \qquad (4.5)$$

where m_{vap} is the evaporated water from the reactor, $H_{vap}(T_{sl})$ is the vaporisation heat at a certain slurry temperature (T_{sl}) and m_{vap} is given by the following equations,

$$m_{vap} = \rho_{air} \cdot q_{air} \cdot (w_{ea} - w_{ia}) \qquad (4.6)$$

$$w_{xa} = 18.015P_{xa}/28.9632(P_{atm} - P_{xa}) \qquad (4.7)$$

$$P_{xa} = P_{xa}^{sat} \cdot RH_{xa} \qquad (4.8)$$

$$P_{xa}^{sat} = 10^{8.07131-1730.63/(233.426+T_{xa})} \qquad (4.9)$$

where w_{xa} is the specific humidity of the inflow (w_{ia}) or exit air (w_{ea}), P_{atm} is the standard atmospheric pressure, P_{xa} is the actual water vapour pressure of the inflow (P_{ia}) or exit air (P_{ea}), P_{xa}^{sat} is the saturation vapour pressure of the inflow (P_{ia}^{sat}) or exit air (P_{ea}^{sat}) calculated by the Antoine equation, T_{xa} is the temperature of the inflow (T_{ia}) or exit air (T_{ea}), RH_{xa} is the relative humidity of the inflow (RH_{ia}) or exit air (RH_{ea}); assuming that RH_{ia} is equal to the ambient air relative humidity and RH_{ea} is 100%.

$H_{vap}(T_{sl})$ is given by

$$H_{vap}(T_{sl}) = H_{vap}(100) - C_{p(H_2O(l))}\,(T_{sl} - 100) - C_{p(H_2O(g))}\,(100 - T_{sl}) \qquad (4.10)$$

where $H_{vap}(100)$ is the vaporisation heat at 100 ℃, $C_{p(H_2O(l))}$ is the specific heat of water and $C_{p(H_2O(g))}$ is the specific heat of water vapour.

Q_{con} is calculated using Eq. 4.11,

$$Q_{con} = (T_{wall} - T_{soil})/S \qquad (4.11)$$

where T_{wall} is the temperature of the reactor wall and is considered equal to T_{sl}, T_{soil} is the soil temperature around the reactor and S is the sum of the conduction resistances of the reactor wall and surrounding soil, given by Eq. 4.12.

$$S = L/k_{wall} \cdot A + D/k_{soil} \cdot A \qquad (4.12)$$

In this equation, L is the reactor wall thickness, D is the distance between the soil temperature measurement location and the reactor wall, k_{wall} is the thermal conductivity of the reactor wall,

k_{soil} is the thermal conductivity of the soil and A is the external surface area of the reactor covered with soil.

Q_{mec} is given by Eq. 4.13,

$$Q_{mec} = U \cdot I \cdot \eta \tag{4.13}$$

where U is the aerator working voltage, I is the aerator working current and η is the heat utilisation factor, which is 100% for fully submerged aerators.

Table 4.1 summarises the data required for the calculation of heat transfer for 150 t slurry in the reactor. For all time-dependent variables (temperature and heat), the time interval was set to 1 h.

Table 4.1 Data required for thermal balance analysis

Term	Eq.	Parameter			
		Symbol	Meaning	Value	Unit
Q_{sl}	4.2	m_{sl}	Slurry mass	1.50×10^5	kg
		$C_{p,sl}$	Slurry specific heat	4.2	kJ/kg/℃
		dT_{sl}/dt	Slurry temperature change rate	Variable	℃/h
Q_{air}	4.4	ρ_{air}	Air density	1.293	kg/m^3
		q_{air}	Airflow rate	120	m^3/h
		$C_{p,air}$	Air specific heat	1.0	kJ/kg/℃
		T_{ia}; T_{ea}	Inflow or exit air temperature	Variable	℃
Q_{vap}	4.5 ~ 4.10	m_{vap}	Evaporated water	Variable	kg H_2O/h
		$H_{vap}(T_{sl})$	Vaporisation heat	Variable	kJ/kg H_2O
		w_{ia}; w_{ea}	Inflow or exit air specific humidity	Variable	kg H_2O/kg air
		P_{atm}	Standard atmospheric pressure	760	mmHg
		P_{ia}; P_{ea}	Inflow or exit air water vapour pressure	Variable	mmHg
		P_{ia}^{sat};P_{ea}^{sat}	Inflow or exit air saturation vapour pressure	Variable	mmHg
		RH_{ia}	Inflow-air relative humidity	Variable	%
		RH_{ea}	Exit-air relative humidity	100	%
		T_{sl}	Slurry temperature	Variable	℃
		$H_{vap}(100)$	Vaporisation heat at 100 ℃	2263	kJ/kg H_2O
		$C_{p(H_2O(l))}$	Water specific heat	4.2	kJ/kg/℃
		$C_{p(H_2O(g))}$	Vapour specific heat	1.84	kJ/kg/℃
Q_{con}	4.11 ~ 4.12	T_{wall}	Wall temperature	Variable	℃
		T_{soil}	Soil temperature	Variable	℃
		S	Conduction resistance	9.44×10^{-4}	℃·h/kJ
		L	Wall thickness	0.30	m
		D	Distance to wall	0.25	m
		k_{wall}	Concrete thermal conductivity	2.80 (JSCE, 1999)	kJ/h/m/℃
		k_{soil}	Soil thermal conductivity	2.47 (JSCE, 1999)	kJ/h/m/℃
		A	Conduction area	220.6	m^2
Q_{mec}	4.13	U	Aerator working voltage	200	V
		I	Aerator working current	33	A
		η	Heat utilisation factor	100	%

4.3. Results and discussion

4.3.1. Treatment performance

Table 4.2 shows the change in chemical properties of the slurry in the ATAT reactor from 2 to 25 March. The COD rapidly decreased from 15.68 to 3.87 g/L during the first few days (from 2 to 8 March) and then increased in the following days. A similar tendency was also observed with TS and VS. The initial decrease is probably attributed to the microbial oxidation of the biodegradable organic substrate (Ugwuanyi *et al.*, 2005). The increase in constituents after the initial decrease is seldom reported in previous studies. Abeynayaka and Visvanathan (2011) observed a COD and nitrogen re-increase after maximal removal in a lab-scale thermophilic aerobic reactor under substrate-limited conditions. This was attributed to the dominance of endogenous respiration. During this process, part of the biomass was oxidised to generate energy, and the cellular constituents were released back into the wastewater; other parts were accumulated in the reactor (Metcalf and Eddy, 2003). Presumably, the late increase in COD and solids in the slurry after their rapid initial drop possibly resulted from the oxidation of the sludge accumulated in the reactor when there was a lack of organic substrate. In addition, Table 4.2 indicates that nitrogen was mainly in the form of NH_4-N and organic nitrogen, and the concentration of NO_x-N was relatively low. During 2 to 25 March, TKN decreased from 2.048 to 1.544 g/L with a removal rate of about 25%. Compared with the nitrogen removal of 90% in activated-sludge treatments (Osada *et* *al.*, 1991), the nitrogen loss was considerably low and appropriate for making fertiliser products; thus resulting in nutrient conservation.

Table 4.2 Chemical properties of the slurry taken from the ATAT reactor during the batch

Sampling time	COD (g/L)	TS (g/L)	VS (g/L)	TKN (g/L)	NH_4-N (g/L)	NO_x-N (mg/L)
3/2 17:00*	15.68 ± 0.35	10.783 ± 0.226	7.210 ± 0.198	2.048 ± 0.088	1.208 ± 0.002	83.3 ± 14.8
3/8 11:00	3.87 ± 0.07	6.007 ± 0.021	3.005 ± 0.015	1.426 ± 0.235	1.093 ± 0.021	36.8 ± 1.5
3/17 11:00	6.69 ± 0.22	9.068 ± 0.043	5.395 ± 0.063	1.431 ± 0.080	1.025 ± 0.021	38.5 ± 0.0
3/25 11:00	11.14 ± 0.41	11.530 ± 0.095	7.586 ± 0.130	1.544 ± 0.025	1.086 ± 0.001	28.0 ± 7.9

* On 3/2, sampling was done after discharging treated product of the last batch and feeding raw slurry of the new batch (n = 3).

4.3.2. Temperature changes

The measured temperature changes during the batch treatment are shown in Fig. 4.5. From 2 to 25 March, the shape of the temperature curve of the slurry in the ATAT reactor changed to resemble the radical sign ($\sqrt{\quad}$). On 3/2, the slurry temperature rapidly decreased to 30 ℃ because of the replacement of the warm treated slurry of the last batch with the cold raw slurry of the new batch. Aeration was started in the morning on 3/3, and the slurry temperature further dropped to 22.1 ℃ during the first 4 h because of the cold air (<10 ℃). Then, the temperature gradually increased to thermophilic ranges and finally reached a maximum of 52.5 ℃ after 120 h. From then on, only a slight decrease in slurry temperature was noted until the end of the batch. The raw slurry temperature in the feed storage tank remained stable at about 12 ℃—slightly higher than the ambient air temperature (averaged at 9.4 ℃) but significantly lower than the slurry temperature in the ATAT reactor.

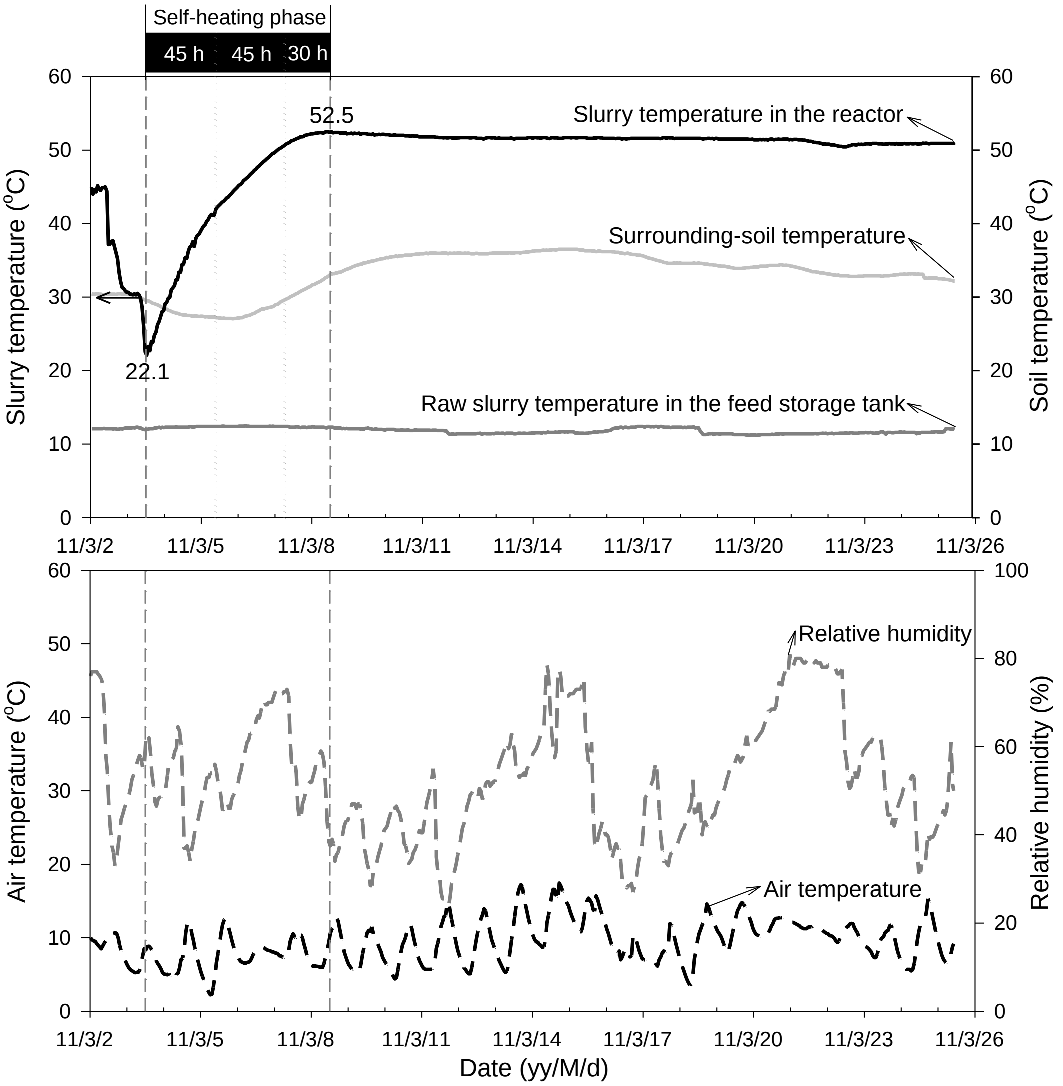


Fig. 4.5 Temperature changes during the batch

On comparing Table 4.2 and Fig. 4.5, we found that the period of COD removal corresponded to the slurry temperature rise. This indicated that the microbial oxidation of bio-available organic compounds released heat, causing temperature rise (Heinonen-Tanski *et al.*, 2005). During the 120 h (from 12:00 on 3/3 to 12:00 on 3/8), the slurry temperature increased from 22.1 to 52.5 ℃. We called this period as ‘self-heating phase’, since there was no external heating device employed during the thermophilic operation of the reactor.

4.3.3. Thermal balance analysis

4.3.3.1. Simulation of heat transfers

All the heat transfer terms (Q_{sl}, Q_{air}, Q_{vap}, Q_{con}, Q_{mec} and Q_{bio}) were calculated from the start of the self-heating phase to the end of the batch. Q_{mec} was calculated to a constant value of 23.76 and the calculated Q_{sl}, heat losses, Q_{bio} are displayed in Figs. 4.6, 4.7 and 4.8, respectively. The unit for the heat transfer terms of the reactor with 150 t slurry is MJ/h/reactor.

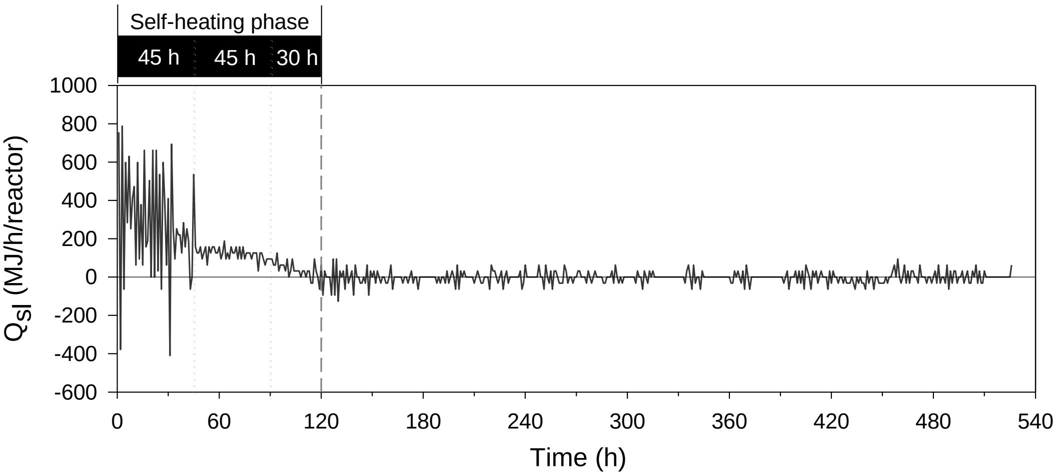


Fig. 4.6 Simulation of Q_{sl} in the ATAT reactor from the start of the self-heating phase to the end of the batch

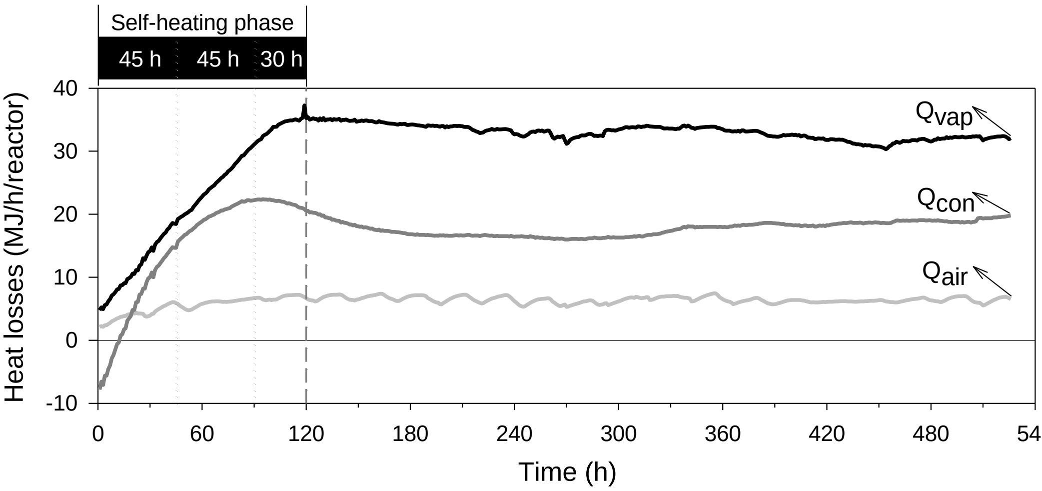


Fig. 4.7 Simulation of Q_{vap}, Q_{con}, Q_{air} in the ATAT reactor from the start of self-heating phase to the end of the batch

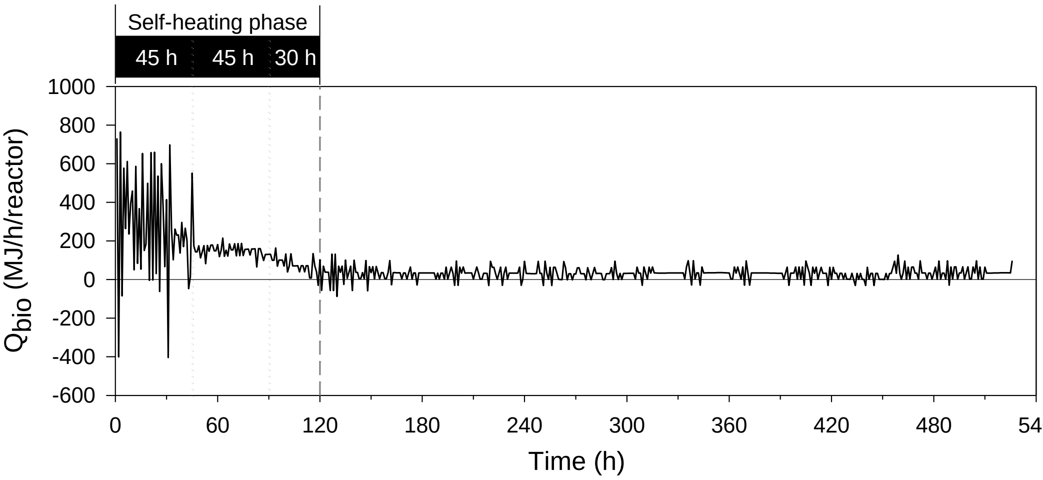


Fig. 4.8 Simulation of Q_{bio} in the ATAT reactor from the start of the self-heating phase to the end of the batch

At the beginning of the self-heating phase (first 45 h), most of the heat was trapped in the slurry, whereas a smaller quantity of the heat was lost and Q_{mec} was insufficient to cause large changes in the slurry enthalpy. Q_{bio} generated from the microbial oxidation of the organic substrate significantly contributed to the slurry heating up to the thermophilic level. Both Q_{sl} and Q_{bio} in this period were significant and constantly changing. In the next 45 h, Q_{sl} slowed down and Q_{bio} reduced to about 150 MJ/h/reactor. Subsequently, Q_{sl} decreased as the slurry temperature reached a maximum of about 50 ℃ and Q_{bio} decreased to a magnitude comparable with Q_{mec} or other heat losses. The variation in the magnitude of Q_{bio} may be attributed to the change in the dominant reactions in ATAT. In the self-heating phase, the decomposition of organic matter was probably dominated by aerobic respiration and biosynthesis. Then, endogenous respiration became dominant when there was a lack of organic substrate.

4.3.3.2. Heat budget

Because the thermal behaviour during and after the self-heating phase was distinct, the heat budgets of these two periods were separately analysed. For each period, the calculated heat transfer terms were added to the accumulated ones in the reactor. Then, the specific mean heat transfer rates (MJ/h/t) were calculated by dividing the respective values by slurry mass and time. The results are shown in Table 4.3.

Table 4.3 Heat budgets during and after the self-heating phase

Term	During the self-heating phase		After the self-heating phase	
	Cumulative heat transfer (MJ/120h/reactor[*])	Specific mean transfer rate (MJ/h/t)	Cumulative heat transfer (MJ/406h/reactor[*])	Specific mean transfer rate (MJ/h/t)
Q_{sl}	19152.0	1.064	−945.0	−0.016
Q_{air}	656.0	0.036	2612.6	0.043
Q_{vap}	2670.5	0.148	13421.4	0.220
Q_{con}	1785.1	0.099	7210.7	0.118
Q_{mec}	2851.2	0.158	9646.6	0.158
Q_{bio}	21412.4	1.190	12653.1	0.208

[*] Slurry mass in the reactor is 150 t.

In the self-heating phase (120 h), 19152.0 MJ of heat was trapped in the 150 t slurry and the total heat loss from the reactor was estimated at 5111.6 MJ, in which Q_{vap} was the highest (52.2%), followed by Q_{con} (34.9%) and then Q_{air} (12.8%). The total heat generation in the reactor was estimated at 24263.6 MJ, in which Q_{bio} accounted for a significant contribution of 88.2%. This indicates that substrate oxidation was the principal factor influencing the heat production in the reactor.

After the self-heating phase, the total heat loss was estimated at 23244.7 MJ, which was 945.0 MJ higher than the total heat gain. The heat lost through the airflow was 2612.6 MJ (11.2%), the heat used for water evaporation was 13421.4 MJ (57.7%) and the heat conducted via the reactor wall was 7210.7 MJ (31.0%). The heat lost from the reactor with the warm and moist exit air (Q_{air} and Q_{vap}) was significant and amounted to 16034.0 MJ, which was 69.0% of the total heat loss. This suggests that the airflow rate was the primary factor affecting heat retention in the reactor. To reserve more heat in the reactor after self-heating, application of a smaller airflow rate in this period than in the self-heating phase was suggested.

On comparing specific mean heat transfer rates in the self-heating phase with those after the self-heating phase, we concluded that the specific mean generation rate of Q_{bio} at 1.190 MJ/h/t in the self-heating phase was 5.7 times greater than that after the self-heating phase estimated at 0.208 MJ/h/t. Furthermore, the specific mean transfer rates of Q_{con}, Q_{vap} and Q_{air} after the self-heating phase were 1.2, 1.5 and 1.2 times, respectively, which were higher than those during the self-heating phase. The specific mean transfer rate of Q_{vap} after the self-heating phase at 0.220 MJ/h/t was higher than that of Q_{bio} at 0.208 MJ/h/t in this period. Therefore, more effective heat conservation measures after the self-heating phase were needed to maintain the reactor temperature in the thermophilic range.

4.3.4. Specific biological heat yield by degrading 1 g COD

For guiding the autothermal operation, the organic substrate heating potential of human excreta was evaluated on a COD basis as follows: noting that the Q_{bio} generated in the reactor at the self-heating phase (from 12:00 on 3/3 to 12:00 on 3/8) amounted to 21412.4 MJ and the decrease in COD from 17:00 on 3/2 to 11:00 on 3/8 was 11.81 g/L, we converted the COD removal of 1771.5 kg for this period by multiplying the working volume of 150000 L. Considering that the COD mass removed during the self-heating phase was 1771.5 kg, the specific biological heat yield from degrading 1 g COD of human excreta was estimated at 12.1 kJ/g COD.

Previous studies presented several results for the specific biological heat yield in substrate oxidation. For instance, on the basis of thermodynamic principles, McCarty (1972) proposed a heat generation of 14.6 kJ by oxidising 1 g COD in aerobic respiration and synthesis reactions for many common organics. Messenger et al. (1993) reported that the specific biological heat yield in aerobic thermophilic sludge digestion ranged from 11.4 to 14.6 kJ/g O. Gomez et al. (2007) recommended a value of 13.6 kJ/g COD for estimating the biological heat generation in the ATAT of sewage sludge. The estimated value of 12.1 kJ/g COD was within the range of literature data, which means that the model is reasonable and this value can be used for guiding the process operation.

4.4. Conclusions

We developed and validated a mathematical model for thermal balance analysis of a full-scale ATAT reactor. The results showed that Q_{bio} was primarily responsible for self-heating and was greater at the beginning of self-heating, then decreased and approached Q_{vap}. The specific mean generation rate of Q_{bio} during self-heating was estimated at 1.190 MJ/h/t, which was 5.7 times higher than that after self-heating. The heat was mainly lost with the exit air. After self-heating, higher heat losses from the exit air led to a decrease in temperature and thus, reduction in the airflow rate was recommended. The specific biological heat yield of human excreta was estimated at 12.1 kJ/g COD.

Chapter 5

Development of a method based on thermal balance analysis to estimate specific biological heat yield in aerobic treatment

5.1. Introduction

As we know, the raw waste biomass with high moisture content, such as livestock excrement, human excreta, should be stabilised and sanitised before land application. Suitable treatment process can convert the raw waste slurry into a stable, hygienic and odour-free product that would be suited for fertilizing cereals and grass farmland. Such treatment can be conducted in thermophilic aerobic reactors, in which thermophilic aerobic conditions lead to rapid biodegradation rate and efficient pathogens inactivation (Mohaibes and Heinonen-Tanski, 2004).

However, the cost of heating a full-scale reactor up to thermphilic temperatures would set a limit for practical application of thermophilic aerobic slurry treatment unless this heat can be obtained virtually free (Lapara and Alleman, 1999). The heat generated in aerobic biodegradation can provide such a free heat source for raising the reactor temperature. It is important to estimate the magnitude of biological decomposition heat needed for complete self-heating to develop an autothermal thermophilic aerobic slurry treatment process (Wang et al., 2013b).

Theoretically, the biological heat yield in aerobic decomposition, Q_{bio}, is proportional to the amount of organic matter degraded (Ahn et. al., 2007), and the specific biological heat yield, Y_{bh}, is generally defined as the heat generation from degrading of 1 g VS or COD in the substrate. The earlier results of Y_{bh} are shown in Table 5.1, which can be used in estimation of the level of organic removal necessary for self-heating of an aerobic reactor to thermophilic temperatures.

Table 5.1 Previous studies about determination of Y_{bh} in aerobic treatment

Previous studies	Y_{bh}	Substrate	Remark
Metcalf & Eddy Inc, 2003	20 kJ/g VS	N.S.*	Empirical value
McCarty, 1972	14.6 kJ/g COD	N.S.*	Thermodynamic analysis
Messenger et al., 1993	11.4–14.6 kJ/g COD	Sewage sludge	Full-scale operation result
Gomez et al., 2007	13.6 kJ/g COD	Sewage sludge	
Wang et al., 2013b	12.1 kJ/g COD	Human excreta	
Riley & Forester, 2002	15.68 kJ/g COD	Sewage sludge	Pilot test result

* N.S.: Not specified

The previous studies to determine Y_{bh} were mostly based on theoretical analysis or engineering calculation and the studies carried on full-scale or pilot-scale ATAT reactors were costly and time-consuming to obtain such a result, and the results repeatability and accuracy were difficult to be ensured and validated. Therefore, it is needed to develop a simple method for determination of Y_{bh} by using a laboratory scale system. By doing so, the Y_{bh} can be estimated repeatedly at the stage of process design. It is favorable for efficient development and optimization of a full-scale ATAT process. This study was aimed to develop a simple method by thermal balance modelling to determine Y_{bh} in aerobic biodegradation on a VS or COD removal basis for exploring a practical ATAT condition by necessary organic removal level.

5.2. Materials and methods

5.2.1. Lab-scale system set-up

Fig. 5.1 illustrates the experimental setup of lab-scale system. A culture vessel (model2600, NALGENE, USA) with the brim capacity of 15 L (O.D. = 289 mm) was used as the main reactor, which was insulated by a glass wool layer of 105 mm (Polycut430, MAG, Japan) all over the body (see Fig. 5.2). Aeration to the reactor was conducted by an air pump (4.5 W, β6000, NISSO, Japan) connected with 2 stone air diffusers by 300 mm silicone air tube, its length was determined from preliminary experiments which indicated that by this length the mechanical heat input from the air pump could be almost dissipated. The air flow rate was monitored by a mass flow meter (SEF-51, HORIBASTEC, Japan), and the ambient air temperature as well as the reactor temperature were monitored by thermo-sensors coupled with recorders (T&D, Japan).

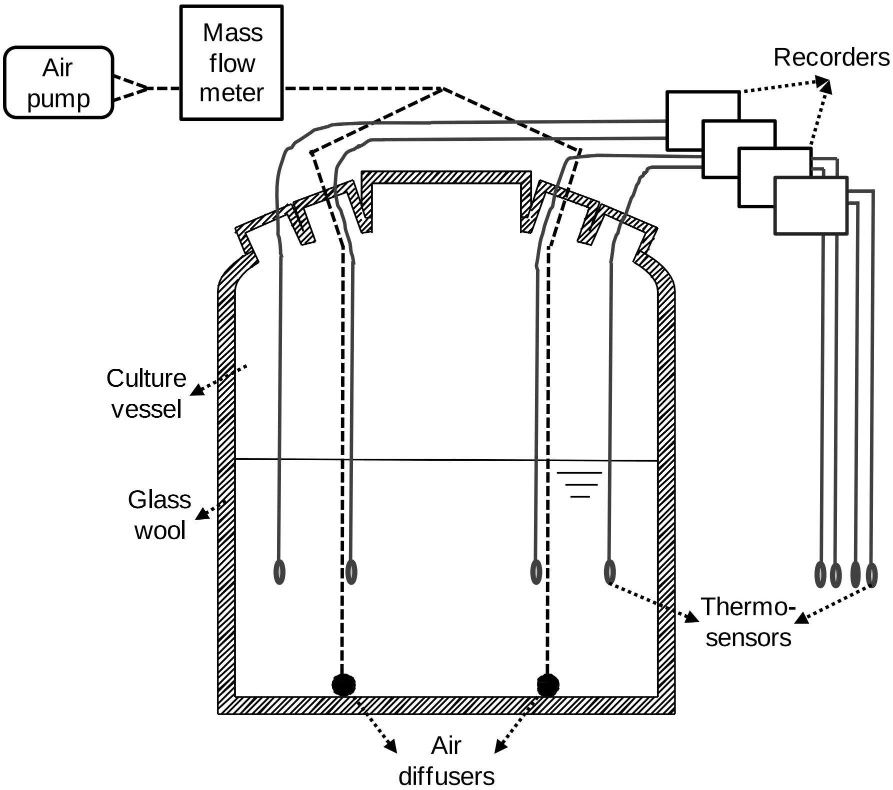


Fig. 5.1 Schematic diagram of lab-scale system

Fig. 5.2 Picture of insulated reactor

5.2.2. Theoretical calculation method

5.2.2.1 Determination of heat transfer characteristics of the insulated reactor

To determine the heat transfer characteristics of the insulated reactor, heat balance analyses (30 times) were conducted during natural cooling of 10 kg warm tap water in a room temperature controlled at 20 ^{o}C (see Fig. 5.3).

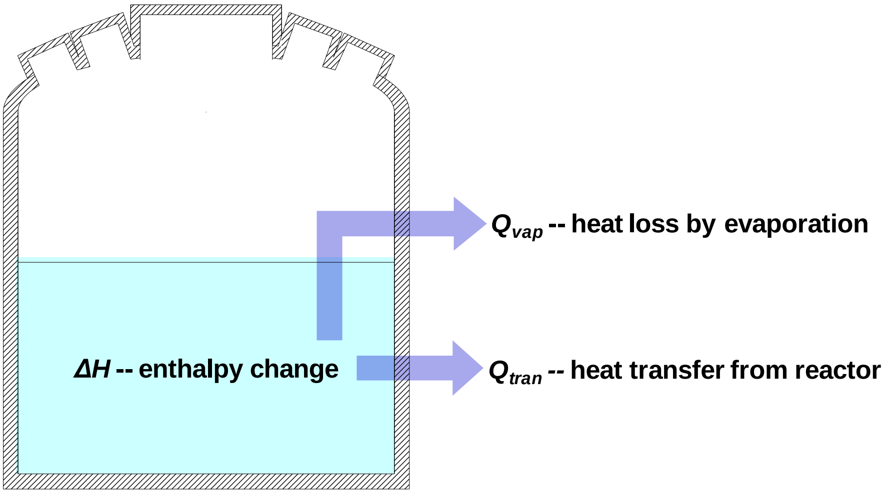


Fig. 5.3 Heat balance during natural cooling of 10 kg warm tap water in a room temperature of 20 ^{o}C

During the natural cooling, the enthalpy change in the reactor (ΔH, kJ) was attributed to the heat loss by evaporation (Q_{vap}, kJ) and the heat transfer from the reactor (Q_{tran}, kJ).

$$\Delta H = Q_{vap} + Q_{tran} \qquad (5.1)$$

in which the ΔH can be calculated as

$$\Delta H = m_{r,t} \cdot C_{p(H_2O)} \cdot T_{r,t} - m_{r,0} \cdot C_{p(H_2O)} \cdot T_{r,0} \qquad (5.2)$$

where $m_{r,t}$ and $m_{r,0}$ are the mass of reactor content of final state and initial state, respectively (kg); $C_{p(H_2O)}$ is the specific heat of water (4.2 kJ/kg/℃); $T_{r,t}$ and $T_{r,0}$ are the reactor temperature of final state and initial state, respectively (℃).

Q_{vap} can be estimated by

$$Q_{vap} = \Delta m_r \cdot H_{vap}(\overline{T_r}) \qquad (5.3)$$

$$H_{vap}(\overline{T_r}) = H_{vap}(100) - C_{p(H_2O(l))} \cdot (\overline{T_r} - 100) - C_{p(H_2O(g))} \cdot (100 - \overline{T_r}) \qquad (5.4)$$

where Δm_r is the mass of evaporated water (kg), determined as the decreased mass of reactor content; $H_{vap}(\overline{T_r})$ is the vaporisation heat at a mean reactor temperature (kJ/kg H_2O), determined from Eq. (5.4), in which $H_{vap}(100)$ is the vaporisation heat at 100 ^{o}C (2263 kJ/kg H_2O); $C_{p(H_2O(l))}$ and $C_{p(H_2O(g))}$ are the specific heat of water (4.2 kJ/kg/ ℃) and water vapour (1.84 kJ/kg/ ℃), respectively; $\overline{T_r}$ is the mean reactor temperature for the time (℃).

Then Q_{tran} can be determined from the difference between Q_{vap} and ΔH.

$$Q_{tran} = \Delta H - Q_{vap} \qquad (5.5)$$

On another hand, based on the following equation to calculate Q_{tran}

$$Q_{tran} = -K \cdot A \cdot \int_0^t (T_r - T_a) dt \qquad (5.6)$$

where the integration of temperature differences between reactor temperature and ambient temperature ($\int_0^t (T_r - T_a) dt$) can be calculated from the measured curves of temperature variation, the product of heat transfer coefficient (K, kJ/m^2/min/ ℃) and heat transfer area (A, m^2) for the experimental reactor can be therefore determined together.

5.2.2.2 Validation of heat loss calculation during tap water aeration

In order to validate the calculation of heat loss during aeration time, heat balance analyses were conducted during aeration of 10 kg warm tap water in a room temperature controlled at 20 ^{o}C (see Fig. 5.4).

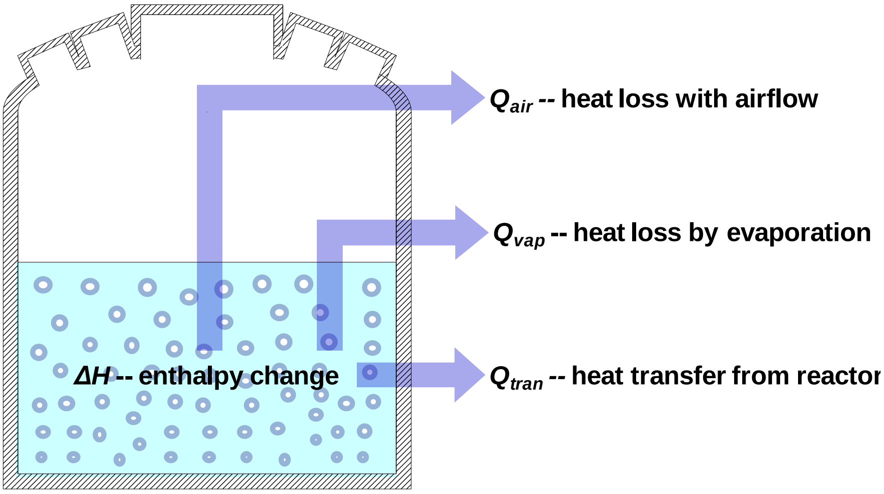


Fig. 5.4 Heat balance during aeration of 10 kg warm tap water in a room temperature of 20 ^{o}C

During the tap water aeration, there was no biological heat generation and other heat gain, the ΔH was equal to the gross heat loss (Q_{loss}, kJ), which was considered as the sum of Q_{tran}, Q_{vap} and the heat loss with airflow (Q_{air}, kJ).

$$Q_{loss} = Q_{tran} + Q_{vap} + Q_{air} \qquad (5.7)$$

where Q_{tran} was calculated by using Eq. 5.6; Q_{vap} was calculated from Eqs. 5.3 and 5.4; Q_{air} was calculated as

$$Q_{air} = \rho_a \cdot V_a \cdot C_{p,a} \cdot (\overline{T_a} - \overline{T_r}) \qquad (5.8)$$

in which ρ_a is the air density (1.293 kg/m^3); V_a is the volume of air supply in the aeration time (m^3); $C_{p,a}$ is the specific heat of air (1.0 kJ/kg/℃); $\overline{T_a}$ and $\overline{T_r}$ are the mean values of ambient temperature and reactor temperature for the aeration time, respectively (℃).

If the sum of Q_{tran}, Q_{vap} and Q_{air} can match ΔH estimated from Eq. 5.2, it indicates that the calculation of Q_{loss} during the aeration is appropriate.

5.2.2.3 Calculation of Q_{bio} and Y_{bh} during wastewater aeration

For determining Q_{bio} and Y_{bh} in wastewater aerobic treatment, heat balance analyses were conducted during aeration of 10 kg warm wastewater in a room temperature controlled at 20 °C (see Fig. 5.5).

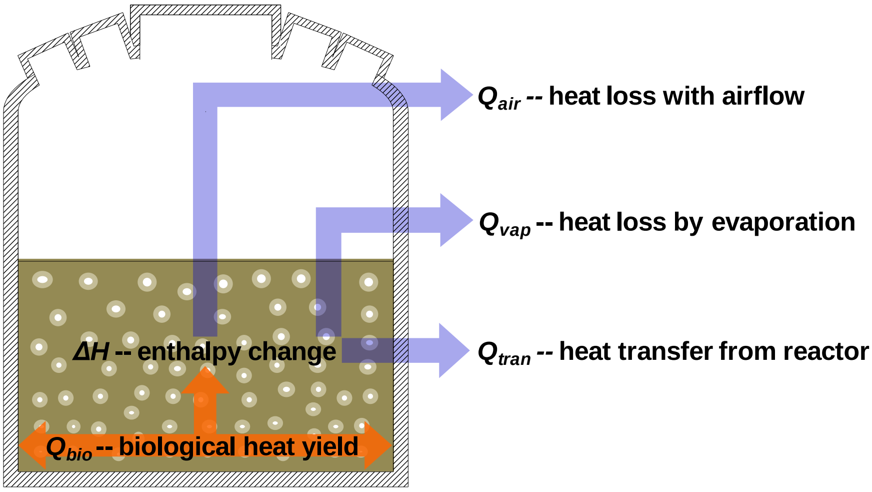


Fig. 5.5 Heat balance during aeration of 10 kg warm wastewater in a room temperature of 20 °C

During the wastewater aeration, Q_{bio} generated while Q_{loss} occurred, and the ΔH was attributed to Q_{bio} and Q_{loss}

$$\Delta H = Q_{bio} + Q_{loss} \qquad (5.9)$$

in which the ΔH can be calculated by Eq. 5.2, and Q_{loss} during the wastewater aeration can be calculated according to the same method used in tap water aeration.

Then Q_{bio} can be calculated as the difference between the ΔH and Q_{loss} for the wastewater aeration, and the specific biological heat yield by degrading 1g of VS ($Y_{bh,VS}$, kJ/g VS) or COD ($Y_{bh,COD}$, kJ/g COD) was estimated respectively as

$$Y_{bh,VS} = \frac{Q_{bio}}{\Delta VS} \qquad (5.10)$$

$$Y_{bh,COD} = \frac{Q_{bio}}{\Delta COD} \qquad (5.11)$$

where ΔVS and ΔCOD were the mass of removed VS and COD during the wastewater aeration (g), respectively, determined as per

$$\Delta VS = m_{r,0} \cdot VS_0 - m_{r,t} \cdot VS_t \qquad (5.12)$$

$$\Delta COD = V_{w,0} \cdot COD_0 - V_{w,t} \cdot COD_t \qquad (5.13)$$

in which VS_0 and VS_t are the VS content in the wastewater of the initial state and final state, respectively (g/kg); COD_0 and COD_t are the COD concentration of the wastewater at the initial state and final state, respectively (g/L); $V_{w,0}$ and $V_{w,t}$ are the wastewater volume at the initial state and final state, respectively (L), converted as equal as $m_{r,0}$ and $m_{r,t}$ by considering that the wastewater density was 1000 kg/m^3.

5.2.3. Preparation of organic model wastewater

A commercial dog food named of Aijou Monogatari (YEASTER, Japan) was used as organic substrate to prepare model wastewater, because the composition of same dog food was shown to be homogeneous. The component analysis of the dog food showed that the moisture content was 7.92 ± 0.03 %, the ash content was 6.37 ± 0.12 %, and the content of TOC, TKN, and total phosphorus (TP) were 473.93 ± 22.71 g TOC/kg dry material, 49.56 ± 0.16 g TKN/kg dry material, and 7.36 ± 0.17 g TP/kg dry material, respectively.

One preparation mixed 500 g of dog food, 4.6 g of hydrolysis promoter (Asahikasei

Clean Chemical, Japan) into 15 L of hot water. After cooling and storing overnight, about 10 L of the upper liquid was collected and preconditioned to 47 ^{o}C in a water bath before being used for aeration trials. The remaining prepared wastewater was used for cultivation of seed sludge without temperature preconditioning.

5.2.4. Cultivation of seed sludge

Seed sludge was collected from ASRs treating swine wastewater at Saga Prefecture Livestock Experimental Station, and the collected sludge was aerobically cultured in a cylindrical plastic drum of 180 L (Harcostar, UK) at university's greenhouse with the model wastewater supplied as nutriment. A fraction of the cultured sludge was then intermittently transferred into an aerobic culture vessel with the working volume of 10 L for successive aerobic cultivation. The vessel was set in the water bath for sludge temperature acclimation to 47 ^{o}C.

5.2.5. Aeration trials

The operating conditions of aeration trials with the temperature preconditioned tap water or wastewater for heat balance analyses are shown in Table 5.2. In tap water aeration trials, 10 kg of tap water of about 47 ^{o}C was initially filled into the insulated reactor and then aerated for 2666 min. In wastewater aeration, the insulated reactor was firstly filled with about 1 L of seed sludge acclimated to 47 ^{o}C or so, and then about 9 L of model wastewater preconditioned at this temperature was added in (at this time the reactor content mass was a bit higher than 10 kg). After mixing them well, three replicate samples were collected from the reactor for analyzing initial organic content, and the mass of the mixture was adjusted to 10 kg, then aeration was started and operated for 2666 min. Thereafter, we collected three replicate samples again to determine final organic content in the wastewater.

Table 5.2 Operating conditions of aeration trials

	Tap water aeration	Wastewater aeration
Reactor content	10 kg tap water	10 kg mixture of seed sludge and model wastewater (about 1:9)
Ambient air temperature	Controlled at 20 ^{o}C	
Initial reactor temperature	Preconditioned to about 47 ^{o}C	
Aeration time	Set as 2666 min	

5.2.6. Measurement and analysis

The weight of reactor and reactor content was measured by a balance (FG-30KBM, A&D, USA) immediately before and after the natural cooling or forced aeration. The reactor temperature and ambient air temperature were measured and recorded by thermo-sensors coupled with recorders (TR-0106, T&D, Japan). During each aeration trial, the volume of air supply was monitored by a mass flow meter (SEF-51, HORIBASTEC, Japan).

The samples collected right before and after wastewater aeration were analyzed for determining COD and VS. COD was analysed using the dichromate reactor digestion method proposed by HACH Company. VS was measured according to the standard methods recommended by APHA (1999).

5.3. Results and discussion

5.3.1. Heat transfer characteristics of the insulated reactor

The products of heat transfer coefficient and heat transfer area ($K\,A$) for the insulated reactor were determined from the heat balance analyses during natural cooling of 10kg of warm tap water while the reactor was allowed to cool without aeration (30 times). The results of $K\,A$ were plotted versus $\overline{T_r}$ in Fig. 5.6. It indicated an approximate linear relationship between $K\,A$ and $\overline{T_r}$ ($K\,A = 0.0002\overline{T_r} + 0.02$, $R^2 = 0.87$), and the relationship was used to estimate the $K\,A$ and Q_{tran} during the aeration trails.

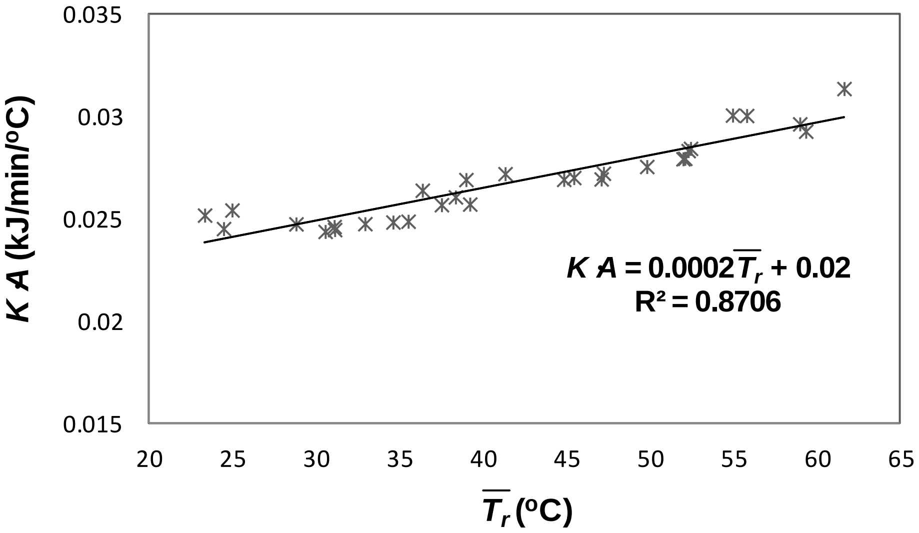


Fig. 5.6 The product of heat transfer coefficient and heat transfer area ($K\,A$) at different mean reactor temperature ($\overline{T_r}$)

5.3.2. Heat balance during tap water aeration

Heat balance analyses during the tap water aeration trials were conducted to validate the simulation of heat loss for the aeration time. The measured data and simulated heat exchanges are shown in Table 5.3. Based on the measured $m_{r,0}$, $m_{r,t}$, $T_{r,0}$ and $T_{r,t}$, the ΔH for a tap water aeration trial was calculated by using Eq. 5.2, and the Q_{loss} was estimated as the sum of Q_{vap},

Q_{tran} and Q_{air}, which were simulated by Eq. 5.3, 5.6 and 5.8, respectively. The sum of Q_{vap}, Q_{tran} and Q_{air} can match the ΔH, which indicates that the heat loss simulation for the aeration time by this method is appropriate.

Table 5.3 Heat balance analyses during the tap water aeration (TWA) trails

Items		TWA 1	TWA 2	TWA 3
Measured performance parameters	$m_{r,0}$ (kg)	10.000	10.000	10.000
	$m_{r,t}$ (kg)	9.810	9.815	9.820
	Δm_r (kg)	0.190	0.185	0.180
	$T_{r,0}$ (°C)	46.73	46.33	46.66
	$T_{r,t}$ (°C)	20.29	20.17	20.05
	$\overline{T_r}$ (°C)	27.42	27.52	27.69
	$\overline{T_a}$ (°C)	19.65	19.61	19.56
	V_a (m^3)	10.96	10.84	10.87
Simulated heat exchanges	ΔH (kJ)	-1126.67	-1114.39	-1132.78
	Q_{tran} (kJ)	-554.27	-563.29	-580.48
	Q_{vap} (kJ)	-462.51	-450.30	-438.06
	Q_{air} (kJ)	-110.11	-110.87	-114.27
	Q_{loss} (kJ)	-1126.89	-1124.46	-1132.81

5.3.3. Heat balance during wastewater aeration

The measured performance and simulated heat exchanges in the wastewater aeration trials are shown in Table 5.4. By comparing evaporated water mass and reactor temperature reduction between the tap water aeration and wastewater aeration, it shows that the water evaporation from the reactor tend to be higher in the wastewater aeration, while the temperature reduction and enthalpy loss in the reactor were smaller during the wastewater aeration. This was because that there was a heat generation from microbial aerobic degradation of organic content in the wastewater aeration. By analysing the heat balance during the wastewater aeration, the Q_{bio} was determined as the difference between the ΔH and Q_{loss}.

Table 5.4 Heat balance analyses during the wastewater aeration (WWA) trails

Items		WWA 1	WWA 2	WWA 3
Measured performance parameters	$m_{r,0}$ (kg)	10.000	10.000	10.000
	$m_{r,t}$ (kg)	9.770	9.790	9.810
	Δm_r (kg)	0.230	0.210	0.190
	$T_{r,0}$ (^{o}C)	46.71	46.59	45.57
	$T_{r,t}$ (^{o}C)	22.70	23.69	23.83
	$\overline{T_r}$ (^{o}C)	29.40	29.86	29.78
	$\overline{T_a}$ (^{o}C)	19.45	20.07	19.47
	V_a (m^3)	10.55	10.26	9.49
Simulated heat exchanges	ΔH (kJ)	-1030.35	-982.69	-932.10
	Q_{tran} (kJ)	-707.29	-697.06	-731.12
	Q_{vap} (kJ)	-558.81	-509.99	-461.46
	Q_{air} (kJ)	-135.73	-129.88	-126.51
	Q_{loss} (kJ)	-1401.83	-1336.93	-1319.09
	Q_{bio} (kJ)	371.48	354.24	386.99

5.3.4. Estimated specific biological heat yield

The measured changes of VS and COD in the wastewater aeration trials are given in Table 5.5. By dividing the Q_{bio} by ΔVS or ΔCOD for each trial, the $Y_{bh,VS}$ or $Y_{bh,COD}$ was estimated out, respectively. The results are also shown in Table 5.5. For this study using dog food as organic substrate to prepare model wastewater, the Y_{bh} on a VS removal basis was 16.98 ±1.53 kJ/g VS, and the Y_{bh} on a COD removal basis was 16.40 ±0.56 kJ/g COD. They are comparable to the literature values shown in Table 5.1, which indicates that the developed simple method can be used as a theoretical basis to calculate Y_{bh} in aerobic biodegradation of organic wastewater.

Table 5.5 Estimated Y_{bh} based on measured removal of VS and COD

Items		WWA 1	WWA 2	WWA 3
Organic content change	VS_0 (g/kg)	8.043	6.205	6.094
	VS_t (g/kg)	5.846	4.112	4.107
	ΔVS (g)	23.31	21.79	20.65
	COD_0 (g/L)	8.883	8.121	7.958
	COD_t (g/L)	6.633	6.154	5.721
	ΔCOD (g)	23.53	20.96	23.46
$Y_{bh,VS}$ (kJ/g VS)		15.94	16.26	18.74
$Y_{bh,COD}$ (kJ/g COD)		15.79	16.90	16.50

5.4. Conclusions

A simple method by using a lab-scale system for determination of Q_{bio} and Y_{bh} in aerobic biodegradation of organic wastewater was developed. By this method, the $Y_{bh,VS}$ and $Y_{bh,COD}$ for the model wastewater prepared using dog food as organic substrate were estimated at 16.98 ± 1.53 kJ/g VS and 16.40 ± 0.56 kJ/g COD, respectively. This method is suitable as a theoretical basis to calculate Y_{bh} in aerobic biodegradation, which is useful in process design to develop an aerobic system with high efficiency.

Chapter 6

General conclusions

Finally, we summarized the general conclusions as follows:

This study developed and validated two heat balance models for ASP and ATAT. By each model, the heat transfer processes and temperature dynamic changes in ASP and ATAT can be simulated and reproduced. The proposed ASP energy model can estimate ASR temperatures with a RMSE of 0.60 ℃, and the model simulation results indicated that heat was mainly gained from solar radiation and biological reaction, while mostly lost through slurry flow and slurry surface in ASP. The total heat gain resulted in a temperature increase of 3.87–5.15 ℃/d in the reactor, but this effect was offset by the heat loss. These results implied the need for effective thermal insulation to be installed if an ASP was to be operated at higher temperatures. On another hand, the results of thermal balance analysis of ATAT showed that Q_{bio} was the principal heat source for autothermal thermophilic operation. The specific mean generation rate of Q_{bio} during self-heating was estimated at 1.190 MJ/h/t and 5.7 times higher than that after self-heating at 0.208 MJ/h/t. The heat loss from airflow, including sensible heat loss by airflow and latent heat loss by evaporation, was primarily responsible for the decrease in reactor temperature after self-heating. The heating potential of human excreta on a COD basis was estimated at 12.1 kJ/g COD. From these results, we found that effective thermal insulation and adequate organic matter degradation are primary aspects inducing the autothermal thermophilic operation.

Then in order to confirm the application potential of thermal balance modelling method, a lab-scale aerobic system was developed and this method was examined to estimate Y_{bh} during aerobic treatment of artificial wastewater. In this work, the estimated results of $Y_{bh,VS}$ and $Y_{bh,COD}$ for the model wastewater prepared using dog food as organic substrate were 16.98 ± 1.53 kJ/g VS and 16.40 ± 0.56 kJ/g COD, respectively. This method was suitable as a theoretical basis for calculating Y_{bh} in aerobic biodegradation on a VS or COD removal basis, which is useful in estimation of the level of organic removal necessary for ATAT development.

These study results were successful in providing an important insight into the process thermodynamics of aqueous biological aerobic treatment and were available as effective guidance for ATAT development, which is conducive to efficient recycling of the high-moisture waste biomass as a fertilizer product for sustainable agriculture production.

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