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Integrated Waste Management

Volume II

Edited by Sunil Kumar

INTEGRATED WASTE MANAGEMENT – VOLUME II

Edited by **Er. Sunil Kumar**

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Meet the editor

Er. Sunil Kumar is an expert in Environmental Engineering and Management with specialization in Solid and Hazardous Waste Management. He is a member of Editorial Boards of several international journals, such as Environmental Monitoring and Assessment, The Netherlands, and Open Waste Management Journal, UK. He is the Regional Editor for Asian countries in the International Journal of Process Wastes Treatment, Serial Publications, New Delhi, and is an expert member in Development of a Fact Base of Greenhouse Gas Emissions from Waste Sector in India leading up to 2030. He has served as Editor for several international publications, among which the Special Issue for Environmental Monitoring and Assessment, The Netherlands; Journal of Air and Waste Management Association, USA and International Journal of Environment and Pollution, and others.

Contents

Preface

The quantum of wastes generated in urban centres has become one of the difficult tasks for those responsible for their management. The problem is becoming acute specially in economically developing countries, where there is a financial crunch, and other resources are scarce.

Although there are varieties of publications dealing with various topics of solid waste management, most of these documents have been published addressing the needs of developed nations. Only a few documents have been specifically written to provide the type of information that is required by those in developing countries. In addition, most of the documents are not accessible to all the readers, and there is also a strong need to update the published documents once again in view of globalization. To maximize the use of limited available resources, it was decided to combine information gathered from both developed and developing countries on all the elements of solid waste management under the title "Integrated Waste Management". Due to overwhelming response from authors all around the world, the book has been divided into two parts, *i.e.* Volume I and II, and the chapters have been grouped under different sub-headings.

This publication has been prepared primarily for researchers, engineers, scientists, decision-makers and policy makers involved in the management of solid wastes. The information provided in both the volumes would also be useful to students studying environmental science and engineering.

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Part 1

Planning and Social Perspectives Including Policy and Legal Issues

1

Operationalising Municipal Solid Waste Management

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1. Introduction

A municipality is an administrative division composed of a defined territory and population (Al-Salem and Lettieri, 2009). While there are many varieties of municipalities, most fall into one of two categories: a single settlement and a land area similar to a township that may contain multiple settlements, or even just part of one, such as a city's municipality. Municipal Solid Waste (MSW) can be defined as solid waste which includes all domestic refuse and non-hazardous wastes such as commercial and institutional wastes, street sweepings and construction debris (Magutu et al., 2010). The major types of MSW are food wastes, paper, plastic, rags, metal and glass, with some hazardous household wastes such as electric light bulbs, batteries, discarded medicines and automotive parts. MSW is thus seen as primarily coming from households but also includes wastes from offices, hotels, shopping complexes/shops, schools, institutions, and from municipal services such as street cleaning and maintenance of recreational areas. In some countries the solid wastes management system also handles human wastes such as night-soil, ashes from incinerators, septic tank sludge and sludge from sewage treatment plants. The complexities and enormity of the challenges become evident when considering other waste types to be managed and these include industrial solid waste, municipal wastewater, industrial wastewater, storm water and hazardous waste.
This chapter will focus on the major ways of managing the Municipal Solid Waste, especially through the proper long-term strategies by looking at the following four key areas: Formulation of the municipal solid waste management strategy; Objectives of municipal solid waste management; Waste management strategies used in municipal solid waste management; and lastly the challenges facing the implementation of sound municipal solid waste management strategies

2. Municipal solid waste

Municipal Solid Waste (MSW) can be defined as solid waste which includes all domestic refuse and non-hazardous wastes such as commercial and institutional wastes, street sweepings and construction debris. In some countries the solid wastes management system also handles human wastes such as night-soil, ashes from incinerators, septic tank sludge and sludge from sewage treatment plants. If these wastes manifest hazardous characteristics they should be treated as hazardous wastes (UNEP, 2005).

Waste management practices differ for developed and developing nations, for urban and rural areas, and for residential and industrial producers. MSW is thus seen as primarily coming from households but also includes wastes from offices, hotels, shopping complexes/shops, schools, institutions, and from municipal services such as street cleaning and maintenance of recreational areas. Residential and commercial types of solid waste include food wastes, paper, cardboard, plastics, textiles, glass, metals, and ashes, special wastes like bulky items, consumer electronics, batteries, oil, tires and household hazardous wastes. Institutions types of solid waste include paper, cardboard, plastics, wood, food wastes, glass, metals, special wastes, hazardous wastes. Municipal services types of solid waste include Street sweepings, landscape and tree trimmings, general wastes from parks, beaches, and other recreational areas. Therefore, the major types of MSW are food wastes, paper, plastic, rags, metal and glass, with some hazardous household wastes such as electric light bulbs, batteries, discarded medicines and automotive parts (UNEP, 2005; UNEP, 2004).

In recent years the volume of waste has been increasing at an alarming rate, posing a formidable challenge to governments (Magutu et al., 2010). The complexities and enormity of the challenges become evident when considering other waste types to be managed and these include industrial solid waste, municipal wastewater, industrial wastewater, storm water and hazardous waste. Often, different government agencies are mandated to manage different waste sectors. This fragmented approach to waste management, coupled with a lack of clear definition and delineation of the different waste types, makes an assessment of current waste management practices in most countries difficult (UNEP, 2005).

2.1 Waste management strategies used in municipal solid waste management

Waste management is the collection, transport, processing, recycling or disposal, and monitoring of waste materials. Operations strategy can be viewed as part of a planning process that coordinates operational goals with those of the larger organization. Since the goals of the larger organization change over time, the operations structure must be designed to anticipate future needs. The operations capabilities of a firm can be viewed as a portfolio best suited to adapt to the changing product and service needs of a firm's customers (Hayes, 1985).

The costs for solid waste management are high especially for collection, transportation, treatment and disposal, which are largely borne by city councils. Methods of collection of waste are either door-to-door or using containers or communal bins. All medium and large cities have administrative structures for providing collection services but often, cities in developing countries use non-compaction trucks for daily collection, with a few cities using compaction trucks and hauling trucks. The most common municipal waste management practices include: recycling/recovery, composting, incineration and land filling/open dumping. The operations strategy is a very important tool in the solid waste management practices and processes (Peters, 1984).

MSW may contain the following materials, which are considered recyclables: ferrous and non-ferrous metals, construction debris, scrap tires, paper/cardboard, plastics, textiles (including cloth and leather), glass, wood/timber, animal bones/feathers, waste oil and grease, cinders/ashes. In the middle-to-low-income cities, there exists a long-standing practice of informal source separation and recycling of materials (Magutu et al., 2010). This has led to the development of enterprises for the gathering, trading and reprocessing of materials. For example Mukuru Recycling project which started in 1991 to help men and women scavengers sell recyclable waste to industries. The national ministries support waste recovery and recycling activities at city level although many of these are family businesses. However, since industries would only be interested to use recycled materials when they cost less than the virgin materials, the practice of recycling is so market-driven that recycling has become selective. The disposal of those unselected recyclables remains a problem.

Informal waste separation or waste picking takes place in three ways: At source - this is in large urban areas, e.g., commercial areas or residential areas with apartments/high-rise buildings for high income earners. Here waste pickers sort out the waste before the authorized collection vehicle arrives. During collection, when the collectors segregate recyclable materials during loading and store them inside the truck or on the sides of the vehicles. At the disposal site – where the waste pickers often live on or near the dumps. However, they risk the danger of potential slides and fires. While waste picking means survival for waste pickers the methods of uncontrolled waste picking can reduce the efficiency of the formal collection system and can be detrimental to health due to exposure to biological pathogens.

Composting is not well practiced. Waste materials that are organic in nature, such as plant material, food scraps, and paper products, can be recycled using biological composting and digestion processes to decompose the organic matter (Al-Salem and Lettieri, 2009). The resulting organic material is then recycled as mulch or compost for agricultural or landscaping purposesHousehold organic wastes, including wastes from the restaurants, are often collected for animal feed. But these are either not working or are not operating at full capacity for a number of reasons, such as: High operating and maintenance costs, poor maintenance and operation of facilities, Incomplete separation of non-compostables, such as, plastics and glass, high cost of compost compared to commercial fertilizers.

Another waste treatment method that is practiced is incineration where 90 percent of non-recyclable municipal solid waste is incinerated. Final disposal of waste is at landfills where 10 percent of non-recyclable municipal solid waste is deposited (Al-Salem and Lettieri, 2009). Singapore has four government-owned and operated incinerators for the disposal of solid waste that is not recycled. However, controversy remains over the soundness of incineration as a waste treatment technology because of greenhouse gas emissions from incinerators. Incineration has been completely banned under the new law on solid waste management (Rio de Janeiro, 1992). The practice of informal incineration or open burning is, however, still prevalent, not only in the rural areas where waste collection is rare but also in peri-urban and urban areas.

The popular meaning of 'recycling' in most developed countries refers to the widespread collection and reuse of everyday waste materials such as empty beverage containers. These are collected and sorted into common types so that the raw materials from which the items are made can be reprocessed into new products. Material for recycling may be collected separately from general waste using dedicated bins and collection vehicles, or sorted directly from mixed waste streams.

Landfills are generally the cheapest and most common disposal method for municipal solid waste (Al-Salem and Lettieri, 2009). Disposing of waste in a landfill involves burying the waste, and this remains a common practice in most countries. Landfills were often established in abandoned or unused quarries, mining voids or borrow pits. A properly designed and well-managed landfill can be a hygienic and relatively inexpensive method of disposing of waste materials. Older, poorly designed or poorly managed landfills can create a number of adverse environmental impacts such as wind-blown litter, attraction of vermin, and generation of liquid leachate. An exception is a large city like Singapore, which faces rising disposal costs due to exhaustion of traditional disposal sites, stricter environmental controls and greater waste quantities, thus requiring other methods like incineration to reduce the volume of waste for final disposal. In the other developing countries, open dumping is the common practice, i.e., municipal solid waste is dumped on swamplands and low-lying areas, which are eventually reclaimed for development. The problems associated with landfills, even with those that are clay-lined, include high water table, groundwater contamination and gas migration.

Incineration is a disposal method in which solid organic wastes are subjected to combustion so as to convert them into residue and gaseous products. This method is useful for disposal of residue of both solid waste management and solid residue from waste water management (Al-Salem and Lettieri, 2009). This process reduces the volumes of solid waste to 20 to 30 percent of the original volume. Incineration and other high temperature waste treatment systems are sometimes described as "thermal treatment". Incinerators convert waste materials into heat, gas, steam and ash. Incineration is common in countries such as Japan where land is more scarce, as these facilities generally do not require as much area as landfills. Waste-to-energy (WtE) or energy-from-waste (EfW) are broad terms for facilities that burn waste in a furnace or boiler to generate heat, steam and/or electricity.

2.2 Formulation of the municipal solid waste management operations strategy

Operations strategy is the “HOW” in any corporate and market strategy. Operations strategy is no longer a tool for continuous improvement and sustainable competitive advantage in the manufacturing sector only, since it can be now applied in the service industry and public organizations.

The operation strategies used in solid waste management can be modeled using a process chart as follows:

From the model, the formulation of organizational strategy must be done by the CEO and the employees through selected committees. The formulation of organizational strategy should be followed by setting of Annual Objectives in Solid Waste Management. The annual objectives includes to: improve public health of the people; improve the environment; and maintain public cleanliness in order to keep public places aesthetically acceptable; by ensuring the proper storage, collection, transportation, safe treatment and disposal of solid waste. This driven by the annual departmental objectives designed according to the department of environment’s major mandates. They are derived from annual departmental objectives especially by the departmental heads and the employees. This is operational Level (origination of Annual Objectives in Solid Waste Management). The policies adequately support the institutions strategic plan: the departmental organizational structure support implementation of strategy; and the procedures/regulations followed by the departments are supportive of change implementation. This should be documented in the current strategic plan (Magutu et al., 2010).

There are so many factors that can enable an organization to take a fresh look at its operations Strategy. The different factors that impact on the operations strategy are: most managers felt that the emergence of aggressive and highly competent competitors, demanding and environmentally conscious customers. Other secondary factors include: advances in production and information technology, global business operations, business process re-engineering techniques and the enormous opportunities for operational

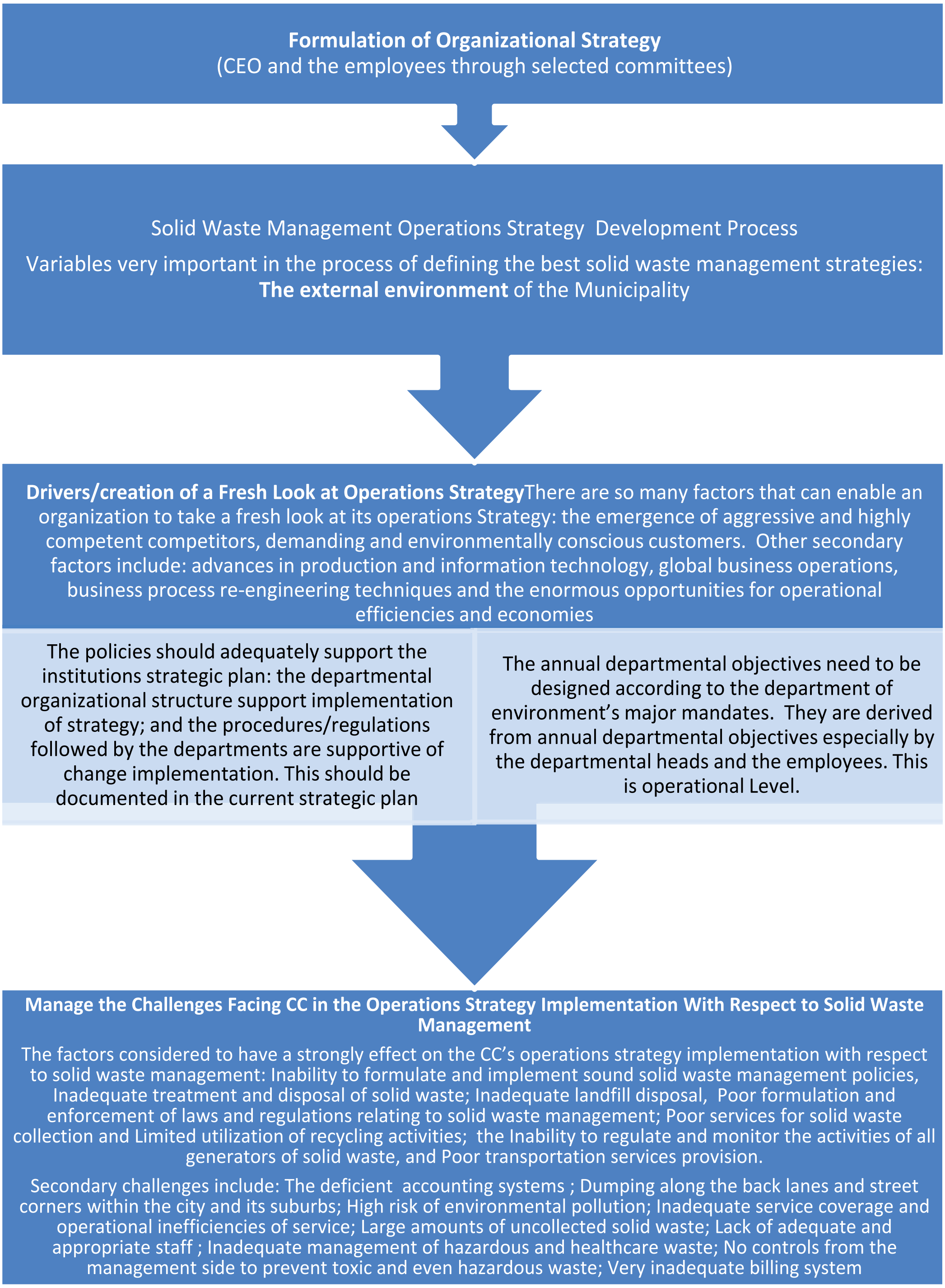

Fig. 1. Formulation of Strategies Used In Municipal Solid Waste Management

efficiencies and economies. The external environment of the municipality's trends in the market; changes in customer wants and expectations (demanding speed of delivery, high quality, and low price); Identifying the company's strengths, special skills of workers, such as expertise in providing customized services or knowledge of information technology; the trends in the political environment changes in the political climate—local, national, and international) and forming partnerships with international firms(Magutu et al., 2010).
Investment on scanning the environment of operation before developing the solid waste management operations strategy is important. As one invests on time and effort in implementing its operations strategy after the environmental scanning and formulation of operations strategy, there is need to invest in its implementation. There are a number of challenges which needs to be managed during the implementation of operations strategy with respect to solid waste management. The factors considered to have a strongly effect on the municipal solid waste strategy implementation include: Inability to formulate and implement sound solid waste management policies, Inadequate treatment and disposal of solid waste; Inadequate landfill disposal, Poor formulation and enforcement of laws and regulations relating to solid waste management; Poor services for solid waste collection and Limited utilization of recycling activities; the Inability to regulate and monitor the activities of all generators of solid waste, and Poor transportation services provision. The secondary challenges include: The deficient accounting systems ; Dumping along the back lanes and street corners within the city and its suburbs; High risk of environmental pollution; Inadequate service coverage and operational inefficiencies of service; Large amounts of uncollected solid waste; Lack of adequate and appropriate staff ; Inadequate management of hazardous and healthcare waste; No controls from the management side to prevent toxic and even hazardous waste; Very inadequate billing systems (Magutu et al., 2010).
This proper management of the challenges finally leads to proper solid waste management.

3. Success stories in solid waste management

Rapid urbanization and the associated growth of industries and services is an essential feature of economic and demographic development in most developing countries. Cities are currently absorbing two-thirds of the total population increase throughout the developing world (UNCHS, 1993). Another striking growth is the steady growth in size of cities. One of the most important environmental consequences of urbanization is the amount of solid waste that is generated. These wastes have fast outstripped the ability of natural environment to assimilate them and municipal authorities to dispose of them in a safe and efficient manner. The resulting contamination affects all environmental media and has a direct negative effect on human health and the quality of urban life.
Most governments all over the world where waste management services have successfully been done subsidizes the budgets for solid waste management up-to over 60 percent. In Japan for example before privatization of solid waste management services, government subsidy to SWM used to be 80 percent while in Sweden it is 70 percent despite residents still paying an equivalent of kshs 800 per month for the solid waste management services. Accra in Ghana, residents pay up to Kshs 700 per month for the solid waste management services. Singapore has a collection rate of more than 90 percent while in Bangkok, Jakarta and Kuala Lumpur the rate is more than 80 percent. In Indonesia, collection rates have been improved through a pre-collection system at villages, which deposit their municipal solid waste at transfer or temporary storage facilities (Rio de Janeiro, 1992).

In Dar es Salaam in Tanzania, the government made a bold step in 1994 to privatize the waste collection and transportation aspects where the city was zoned and different private companies were given areas of operation while collecting waste management charges approved by the various municipalities. Different municipalities enacted their own by-laws to govern and guide the operations of the private sector. The City only manages the disposal site but this again, the city of Dar-es-salaam has partnered with a strategic investor from Germany to develop a sanitary landfill site as for a long time the city has operated with a controlled disposal site. The private companies collect waste management charges from the citizens and only approved rates by the council are applied. The city has a department for solid waste disposal, which only develops policies, rules governing the private sector operation, supervision and the management of the disposal site. The private companies contracted are locals and sometimes they get a back- up from the city council whenever they cannot deliver. In this case, the council has to have what to fall back to and therefore the council cannot afford at any time to have no fleet of vehicles (Rio de Janeiro, 1992).
In Cairo Egypt, the Government decided to invite international bidders for the solid waste management services when the council failed to provide the required services and the city was dirty while the residents were not agreeable to pay for services, which were hardly there in 2002. The Giza region in Cairo, which has a population of 6.5 million was divided into three zones and contracted to three different companies. Jacorossi Impresse is one of the companies managing cleansing services from a population of about 1.2 million under a 15 yr period contract (Rio de Janeiro, 1992).

4. Conclusion

Solid waste management or municipal solid waste management varies widely among different countries and regions. Most of the management services are often provided by local government authorities, or by private companies in the industry. This can be done through The application of waste hierarchy which refers to the "3 Rs" reduce, reuse and recycle. This hierarchy classifies waste management strategies according to their desirability in terms of waste minimization aimed at extracting the maximum practical benefits from products and to generate the minimum amount of waste. The Extended Producer Responsibility (EPR) is a strategy designed to promote the integration of all costs associated with products throughout their life cycle including end-of-life disposal costs into the market price of the product. The other strategy is Polluter Pays Principle, where the polluting party pays for the impact caused to the environment, which implies that a waste generator pays for appropriate disposal of the waste.

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2

Status of Waste Management

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1. Introduction

Solid waste management problem appears to be the most prominent in urban cities and large towns across the world due to the huge quantity of solid waste generated from domestic and commercial activities. In most cities and large towns of the world, solid waste is not only heaped in huge quantities on refuse dumps but also thrown and made to lie around in piles in the street and in small illegal dump on any piece of unused land. Most third world countries have worst cases than industrialised countries which have the money, technical knowhow and public attitudes to control and manage their waste to some degree. This chapter presents a practical approach to the assessment of the Status of Solid waste management with application in Nigeria. The objectives of the chapter are discussed as follows:

1.1 Solid waste characterisation

Waste characterization is a waste stream analysis which involves a logical and systematic approach to obtaining and analyzing data on one or more waste streams or sub-streams. Waste characterization provides an estimate of solid waste quantity and composition. Two commonly use method of waste characterization are – material flow approach and site-specific study. However there is currently no agreed international standard for waste stream analysis or waste characterisation although many countries have national procedures for analyzing their waste. Waste characterization has been developed and discussed in this chapter.

1.2 Assessment of the solid waste chain

Waste management in all ramifications, is simply a planned system aimed at effectively controlling the production, storage, collection, transportation, processing and disposal of waste. Waste management is an important element of environmental protection. Its purpose is to provide hygienic, efficient and economic solid waste storage, collection, transportation and treatment or disposal of waste without polluting the atmosphere, soil or water system. The various steps involved in the management of solid waste from generation to the point of sanitary disposal are referred to as solid waste chain. It therefore means that the solid waste chain is the path trace by solid waste from generation to the final disposal point. The solid waste chain has been extensively discussed in this work

1.3 Assessment of the status of source segregation

The process of recovery is the main task in the Solid waste Management Mix that is sustainable. Waste sorting is a major part of sustainable solid waste management process because if the wastes are sorted correctly, about 30 - 50% of the work is done. The solid waste source sorting process has also been presented in this work.

1.4 Existing legislation for waste management

Various policy initiatives are required in alleviating the urban solid waste management problems. Policy initiative has been review as it applies in Nigeria

1.5 Review of the institutional and funding aspect of waste management

The institutional framework and funding are aspects of the most important requirements for improved solid waste management. The status of institutional framework and funding of solid waste management has been examined and presented with respect to Nigeria.
This presentation will not be meaningful without feasible suggestions of strategies for improved urban solid waste management system. Suggestions have therefore been presented in this work.

2. Definitions of waste

Waste is a term generally used to describe the materials we throw away. In the United state of America it includes objects the lay audience commonly calls garbage, refuse and trash (Davis and Masten, 2004). The oxford advance dictionary defined waste as unused materials or substance produced while making something. Another dictionary – the word net dictionary defined waste as any material unused and rejected as worthless or unwanted. Waste may also be defined simply as left-over, or already used items waiting for reuse or disposal (Audu, 2007). In addition another definition state thus, waste is any unwanted material intentionally thrown away for disposal (Hoornweg, 1999).
The problem with these definitions is that for any material to be a waste it must be thrown away for disposal. Not all wastes are thrown away for disposal. A lot of wastes are kept by the owner for sale which becomes useful raw materials to other persons. The dictionary meaning and other definitions stated above when carefully examined, rely too much on other terms (such as garbage, trash, refuse etc), which do not provide a means to determine whether a given particle, material or item is not a waste based on its composition and its instantaneous relationship to an owner, a generator, a recycler or a legal designation (Palmer, 1992).
As a follow-up to this, Palmer (1992) proposed the following prescriptive definition of waste which expressed in concise form the critical properties of waste which make it a subject of importance by presenting the instantaneous relationship to the owner. *"Waste is any object whose owner does not want to take responsibility for it"*. (Palmer, 1992)
A careful examination of this definition will show that this definition is of the type, which depends on the relationship with an owner. Without an identified owner an object may not be considered in terms of waste. This definition shows that the first step in discussing waste is to locate and identify an owner. This implies that anything without an owner is a waste. If it is potentially valuable an owner will emerge Palmer (1992). However any object which acquires a new owner, who wishes to take responsibility for it, is not a waste, no matter what history.
Another definition states thus "waste is a material which has served its original intended use and sometime discarded." The problem with this kind of definition is that it is

unforgiving. Any material which once had another use is now a waste no matter that it is brand new, clean and valuable, much less perfectly recyclable if given a little attention.
Waste is also seen as "any material which the holder discards, is obliged to discard or intends to discard". It has an objective element in the sense that a material becomes waste by virtue of a circumstance which is outside the control of the owner or holder of the material; namely the fact of abandoning the material or the fact that there are provisions which determine that certain material is to be classified as waste. (Melissa, 2005)
The USEPA, regulatory definition is broader in scope. It defines solid waste thus "solid waste include any discarded item, things destined for reuse, recycling, or reclamation, sludge and hazardous waste. The regulatory definition specifically excludes radioactive waste and in situ mining waste (Davis and Masten, 2004).

2.1 Proposed definition

Having examined the definition given by various authors as shown above it became clear that they are not all encompassing hence there is need to propose a definition that can define waste more broadly as follows: "Any object that may or may not have served its intended use and the owner is not ready to continue to take responsibility for ownership and or continuing to keep it and he or she is ready to discard it if possible is a waste".
Waste may be asset or a liability depending on the management system applied on it. To the developed countries wastes are asset because they have the technology and public attitude that help the nations to reverse the state of their waste which make it become assets instead of liability. Take for example, the recycling of waste paper save trees use for making fresh pulps.
In this chapter the status of municipal solid waste management has been discussed with a case study in developing country - Nigeria. The study covered the nature and quantities of municipal waste generated, the availability of technology for handling and processing, the degree of industrialisation in terms of extent of mechanisation and availability of technological resources, perception and attitude to solid waste management.

3. Municipal solid waste

Municipal solid waste (MSW) is defined Cointreau (1982) as non-air and sewage emissions created within and disposed of by a municipality, including household garbage, commercial refuse, construction and demolition debris, dead animals, and abandoned vehicles. Municipal solid waste is generally made up of paper, vegetable matter, plastics, metals, textiles, rubber, and glass (USEPA 2002). Municipal solid waste disposal is a major concern in developing countries across the world, as high poverty, population growth, and high urbanization rates combine with ineffectual and under-funded governments hampers efficient management of wastes (Doan 1998, Cointreau 1982). In most cities and large towns of developing countries, solid waste is not only heaped in huge quantities on refuse dumps but also thrown and made to lie around in piles in the street and in small illegal dump on any piece of unused land. Most third world countries have worst cases than industrialised countries which have the money and technical know now and public attitudes to control and manage their waste to some degree.

3.1 Characterisation of municipal solid waste (MSW)

Municipal solid waste characterization is a waste stream analysis which involves a logical and systematic approach to obtaining and analyzing data on one or more waste streams or sub-streams. The analysis usually provides - the composition of the waste stream and an estimate of the quantity of the waste stream (EPA Ireland, 1996). There is currently no agreed international standard for waste stream analysis or waste characterization although many countries have national procedures (EPA, Ireland, 1996). However there are two basic approaches to estimating quantities of municipal solid waste - Site-Specific Study and material flow approach (USEPA, 2006).

3.2 Site-specific study

This method involves sampling, sorting, and weighing the individual components of the waste stream. The method is useful in defining a local waste stream. The site-specific study requires a large numbers of samples to be taken over several seasons. Large sample ensure that the results are not skewed and misleading. This method is best applied in the characterization of a solid waste stream that has components such as food and yard trimmings. A study that involves the use of site-specific study is usually preceded by survey. Solid waste survey is a statistical study of a sample population which involves asking questions about age, income, opinions, size of family, and other aspects of people's lives with respect to solid waste. Usually survey research method is employed when research is to be carried out in a large population. Random sampling method is commonly used to observe and collect data from the population. The sample size required for a survey partly depends on the statistical quality needed for the survey findings; this, in turn, relates to how the results will be used (Haruna, 2004, Scheuren, 2004). Two methods commonly applied in Surveys research are the questionnaire and interview methods. Questionnaires are usually paper-and-pencil instruments that the respondent completes. Interviews are completed by the interviewer based on what the respondent says. Sometimes, it's hard to tell the difference between a questionnaire and an interview. The procedure for MSW Characterization using site-specific study is discussed in the following sections.

3.2.1 Selection of a representative sample

Selecting a representative Sample is one of the most difficult tasks associated with waste stream analysis (EPA, Ireland, 1996). It is of critical importance that a sample be collected that is representative of the waste management unit under study. The first step in good sample design is to ensure that the specification of the target population is as clear and complete as possible to ensure that all elements within the population are represented. Several sampling techniques exist - Cluster sampling, Multi-stage sampling, Quota sampling, Simple random sampling, Stratified sampling, Systematic sampling etc. As you can see listed above there are many methods available for use with varying degrees of complexity. Certain methods suit circumstances better than others.

3.2.2 Sampling

The most convenient way to select a representative sample is to use the social class grouping. The population is group into three major social classes – Upper social class, Middle social class and Lower social class (Lindsey and Beach, 2000). The forth is the Underclass.

The upper social class consists less than 5 percent of the population and is group into two sub-classes: upper-upper and lower-upper social classes. The middle social class is also broken into three sub-classes - upper-middle, average or middle-middle and lower middle class. The membership of the sub-class is determined by educational background and earning.

In Nigeria it is assumed that over 70% belong to the lower social class. This class is made up of those who barely half manage to complete secondary school and less than about 25% are able to get university education (macionis, 2002). They own their houses in least desirable neighborhood. Society segregates the lower social class especially when no education at all.
The forth is the Underclass. Very little percentage of the population is locked up in this class. The members of this group lack employable skill and have little or no experience in the job market. Unless given extensive training, they are virtually unemployable. Sociologists disagree about what to call this class. Some use the term "Underclass". In America some argue that this word is stigmatizing, a real concern given the negative classist attitude of most Americans (Lindsey and Beach, 2000)

3.2.3 Sample size

The size of sample to be taken is dependent on the number of solid waste generation units in the sampling area. The following procedure may be employed in selecting the sample size.
A breakdown of social class groups in the sampling area is obtained from the census figures. The number of sampling units (households for domestic waste) to be surveyed is determined. The minimum number of sampling units is 50 per 500 households. For domestic waste this will result in a sample of approximately 1,000 kg, assuming a waste generation rate of 20 kg /household/ week (EPA, 1996). For practical purposes, the weight of the sample for a single survey should be kept below about 5,000 kg, which is roughly equivalent to the waste collected from about 250 households. The recommended range for a survey therefore, is, roughly, 50 – 250 households. However in larger areas, where the sample size will be greater than 250 households, it is recommended that a survey be split into several sub-surveys.

3.2.4 Sample collection

In Nigeria and in most other industrialized countries such as United States, solid waste collection is by trucks fig 1. The trucks are usually parkers, tippers and trucks that carry hydraulic rams to compact the waste to reduce its volume and thus can carry larger quantity. The sample should be collected from the selected sampling units on the same day as normal collection.
The vehicle are weighed prior to and after sample collection so that the total weight of the collected sample can be obtained by determining the deference between the weight of the collection vehicle before and after collection of the waste. Occupants of households and operators of firms chosen for the survey should not be informed about the survey so that any bias that may be created by a temporary change in habits can be eliminated. However in developing countries the people show indifference to solid waste issues and as such the approach above is often not easily applicable. The weight of the total sample should be obtained before sorting and the number of sampling units (households or firms) included in the survey recorded so that the average weight of waste per household per week can be determined.

3.2.5 Sample analysis

The samples are then sorted into the types and classes of solid waste and the weight of each type and class determined and recorded. The moisture content and the bulk density of the sample should be measured. This information will help comparison of results of different surveys as large fluctuations in either moisture content or bulk density will normally reflect a significant difference in waste composition.

Fig. 1. Solid waste collection vehicles with hydraulic ram

3.2.6 Safe disposal of sampled waste

After the analysis arrangements should be made for appropriate and safe disposal of the waste to an authorised site having completed a waste composition survey.

3.3 Materials flow approach

In the material flow methodology production data (by weight) are collected for the materials and products in the waste stream. Waste generation data is obtained from the data collected by making specific adjustments to the production data for each material and product category. Adjustments are made for imports and exports and materials diverted from the municipal solid waste (MSW) stream (e.g., for building materials made of plastic and paperboard). Adjustments are also considered for the life spans of various products. One major disadvantage is that the materials flows methodology requires additional sampling study to determine food wastes, yard trimmings and a small amount of miscellaneous inorganic wastes. This method is widely used in the United States of America (USEPA, 2006).

4. The solid waste chain

Waste management in all ramifications is simply a planned system aimed at effectively controlling the production, storage, collection, transportation, processing and disposal of waste. Waste management is an important element of environmental protection. Its purpose is to provide hygienic, efficient and economic solid waste storage, collection, transportation and treatment or disposal of waste without polluting the atmosphere, soil or water system. The path trace by solid waste in the management of solid waste from generation to the point of disposal is referred to as solid waste chain. The solid waste management is a complex process, involving multiple steps (solid waste chain) shown in fig. 1.

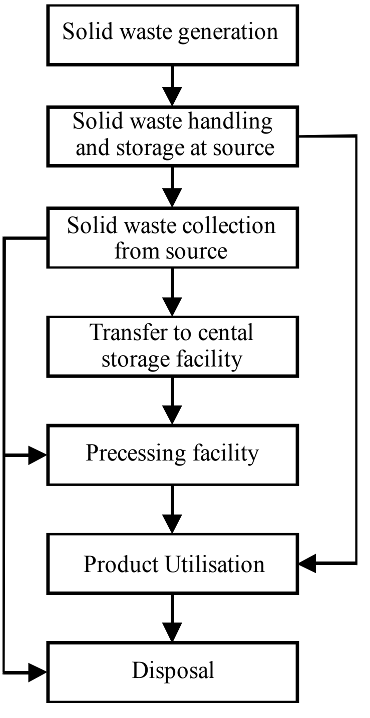

Fig. 1. The solid waste Chain

As indicated in fig.1, the first step in the solid waste chain is the generation of solid waste. Once a material no longer has value to the owner, and owner does not want to take responsibility for it, it is considered to be a waste (Palmer, 1992). The generation of waste varies by country, socioeconomic status and as a result of many other practices (Davis, eta al 2004). Once solid waste is generated, it must be handled properly and stored at source for collection by solid waste disposal agents. The processing and handling may include sorting or segregation, washing and storage so as to ensure recycling of some portion of the waste. Other steps included in the solid waste chain are collection, transfer to central storage facility, final processing facility, product utilization and disposal.

5. Solid waste source segregation

There are four common methods of solid waste Disposal – Landfill, incineration, composting and anaerobic digestion and Recycling (Rao, 2006, Audu, 2007). Incineration, composting and anaerobic digestion are volume reducing technologies; however, residues from these methods must be land filled (Seo et.al 2004). Recycling is one of the waste management techniques that can ensure sustainability of any solid waste management strategy. Recycling is a waste minimization strategy which can be used to divert or prevent discarded material from the waste stream. It consists of a series of activities that are preceded by waste segregation/sorting, processing and manufacturing into new products. Recycling is one of the solid waste management that can ensure sustainability of solid waste management as it converts the waste from liability to asset.

In most large towns and cities in Nigeria solid wastes including medical waste are commingled and disposed at the solid waste dump site. Thereafter open air incineration without pollution control is carried out on the waste. This is not sustainable as it does not bring financial return at the end point of the waste. If a sustainable solid waste management must be realized recycling which ensure financial returns at the end point of the waste must be included in the waste management designed. The first step in solid waste management which will include recycling is segregation/sorting. If solid waste is sorted about 30% of the work is done (Chidubem, 2008). Several methods of sorting solid waste exist.

6. Solid waste management policies and institutional framework

In order to handle growing volumes of wastes generated in a country, proper policies need to be enacted and implemented. These policies are formulated based on the provision of the constitution. A country's constitution is the most important legal document from which all laws and regulations are derived pertaining to the authority of every governmental unit (federal, state, or municipal). For instance the constitution of Federal Republic of Nigeria (F.R.N), 1999 defines the sphere of action and constraints on government by granting certain inalienable rights and obligations to every citizen such as freedom of speech and freedom of movement as well as freedom of association. The Environmental Objectives and Directive of State Policy on the Environment contained in the Constitution of F.R.N,1999 state that, "the state shall protect and improve the environment and safeguard the water, air and land, forest and wildlife of Nigeria" Sec 20 of the Nigeria Constitution (Nigeria 1999). In an effort to develop a framework within which the objectives of protecting and improving the environment can be realized, the constitution allocates certain legislative competencies to each of the three tiers of government. The responsibility for applying the legislation falls to the judiciary. The constitutional obligations are: That everyone in Nigeria has the right to have an Environment that is not harmful to his or her health well being, have the environment protected, for the good of present and future generations, through reasonable laws etc.

Based on the above several agencies have been created at the state level and local Government levels that are involved at least partially in solid waste management. Such agencies include – State waste management board (SWMB), Operation Cleanup, Special Environmental task force, local government environmental department etc. However, there are often no clear roles/functions of the various state and local government agencies defined in relation to solid waste management and also no single agency or committee designated to coordinate their projects and activities. As a matter of fact the government has not been able to develop administrative infrastructure to regulate the management of solid waste and establish pollution control measures in most large cities and metropolis. These therefore results in the lack of coordination among the state and Local Government agencies. The lack of coordination among the relevant agencies often results in duplication of efforts, wastage of resources, and un-sustainability of overall solid waste management programmes.

7. Solid waste management funding

Funding is one of the most importance requirements for the success of solid waste management. In developed countries such as the United States of America (USA) waste management funding policies are well in place, but in developing countries solid waste management is given very low priority, very limited funds are therefore provided to the solid waste management sector by the government. The waste management agencies are therefore left with options of coping with the little fund raised from the user service charge. The waste disposal service charges collected by the disposal agents is too little to make any meaningful impact on solid waste management and users' ability to pay for the services is very limited by their income, and their willingness to pay for the services which are irregular and ineffective is not high either. More so the end point of the solid waste does not provide financial reward to waste disposal agents hence the only source of finance to the disposal agents is the disposal service charges.

8. Case study

Case studies have been carried out in Nigeria. Results obtained from these case studies have been compared with results obtained by other researcher in and outside Nigeria. The multi-stage, stratified-random sampling techniques were applied in these case studies. The study involved survey and solid waste sampling. In carrying out these surveys and sampling, some major elements that helped to cut across the population using the representative samples were considered. Each study unit chosen was based on some major elements - Type of housing/house, size of household and social economic status for domestic solid waste, type of firm, size of firm and nature of products for commercial solid waste. The case studies presented in the following sections were carried out in Lagos and Benin metropolises in Nigeria.

8.1 Background to study sites

Lagos is a city in Lagos state, Nigeria, undergoing accelerated urbanization with influx of people on a daily basis from other parts of the country and neighbouring countries. According to the 2006 National census figure, Lagos State has a population of 9,013,534. However, based on a U.N study and the State Regional Master Plan, the State is estimated to have above 12 million inhabitants. Out of this population, Lagos metropolitan area is occupied by over 85 percent on an area that is about 37 percent of the land area of Lagos State. The rate of population growth is about 300,000 persons per annum with a population density of about 1,308 persons per square kilometer. In a recent UN study the city of Lagos is expected to hit the 24.5 million-population mark and thus be among the ten most populous cities in the world by the year 2015 (Lagos State Government, 2004).

Management of solid waste did not become a phenomenon in Lagos until the 1970s when due to oil boom Lagos witnessed massive influx of people as a result of industrial growth and urbanization from less developed part of Nigeria and West Africa. Management of solid waste started in Lagos state with the establishment of Lagos state Refuse Disposal Board in 1977. It was later changed to Lagos state waste disposal board due to added responsibilities of commercial and industrial solid waste collection and drain cleaning. Collection and disposal of scraps and derelict vehicles were added in 1981. Lagos State Waste Management Authority (LAWMA) was established in December 1991 with the responsibility of collection and disposal of municipal waste in the state (Ola, 2006). Since creation LAWMA has been faced with serious problems on the issues of waste management. These include limited budget, inadequate tools and waste data, poor public attitude, etc.

Mushin Local Government is located at the geometrical centre of Lagos metropolis. It is one of the oldest local governments in the state. Mushin is an entirely urban local government Area though its level of urbanization is not as rich as other local governments in the metropolis. Nevertheless, the vibrancy of its population is a sign that given adequate encouragement, the level could still be improved considerably. According to the final figures of the 2006 National Census, Mushin Local Government has a population of 633,009 people. The Local Government area is highly residential and there are no farmlands available for farming.

Benin Metropolis has similar characteristic with Lagos metropolis. It encompasses Benin City the capital City of the ancient Bini kingdom and it is made up of three local Government areas – Oredo, Egor and Ikpoba-Okha local government areas. These local government Areas are located within the three geographical zone of Benin metropolis – the traditional core zone, the transitional zone and the outer zone (Ikelegbe and Ogeah, 2003). The total Population in Benin metropolis is made up of about 1,085,676 persons (National Population Commission, 2006).

Solid waste management is in crisis stage in the Lagos and Benin metropolis and indiscriminate dumping of solid waste around homes, market places and around the street corners has almost become an acceptable life style in the metropolis as solid waste is allowed to lie around without serious consideration even by the government authorities. What is done to solid waste is simply solid waste relocation and not management in the real sense of it as waste is collected from source of generation and dumped on any un-used land by the disposal agents. Worst still the government waste management authorities do not have a complete record of the waste management agents in the metropolis and hand carts are largely patronised by households and small and medium scale enterprises for the disposal of their solid waste.

8.2 Methodology

Mushin local Government Area (LGA) of Lagos metropolis and Oredo LGA of Benin metropolis were selected for the study. The total Population in the metropolises was obtained from the 2006 Population census figure (National Population Commission, NPC, 2006). The population in the metropolises is not defined in terms of social class. The multi-stage stratifies random sampling technique was therefore design and applied for the study. The population was broken down into household unit (a house hold is a group of people living together in a house as a unit). The average household size in Nigeria is 7 (NPC, 1991), (NPC, 2007). Based on NPC submission, the total households was determined. A listing of the houses in the study area was done. A representative sample size was then calculated using a confidence level of 95% and confidence interval (margin of error) of 4%. The minimum number of households in any solid waste survey is 50 per 500 households and a maximum of 250 households in the sampling area (EPA, 1996). Due to the largeness of the sample size calculated the study was broken down into three sub-surveys each in the two metropolises. 250 households each were further selected from the selected sub-survey area. Each selected household was visited several times. In the first visit, contact was made and participation consent requested. Upon approval, a second visit was made to distribute the questionnaires and moderate size solid waste storage bags. The next visits were made at regular interval to retrieve the questionnaires and collect the solid waste generated over 7 days. Interviews were also conducted during the visits.

8.3 Solid waste generation

As detailed in the methodology, the multi-stage, stratified-random sampling technique was applied. The Mushin LGA and Oredo LGA located at about the geogrphical center of Lagos and Benin metropolis was chosen for the survey. A total weekly average of 2263.2Kg of domestic solid waste was measured in Lagos metropolis within the period of study. Base on these figures a daily generation rate of 0.57kg per person per day (ppd.) was calculated as shown in table 1 for Lagos metropolis. Analysis of the data measured showed that about 47.86% of food waste, 46.47% of rubbish and 4.92% ash were generated. The 46.47% of rubbish was made up of 12.63% of rubber and Plastic materials, 18.16% of other combustible material (paper, cloth, foam, etc.) and 15.685% of non-combustible materials (Metal, glass, ceramics, etc).

S/N	Component of Solid Waste	% of total waste measured	Waste generated Kg– ppd
1	Organic Food waste	47	0.27
2	Combustible waste Rubbish		
	a. Plastic	12.63	0.07
	b. *Others	18.16	0.11
	Non-Combustibles	15.68	0.09
3	Wood Ash	4.92	0.03
	Total		0.54

*Others - This include Paper, Clothes etc

Table 1. Waste generation Data for Lagos metropolis

In addition a total 5373.61Kg of domestic solid waste was measure within the period of study from Oredo LGA. Base on this figure a daily generation rate of 0.425kg per person per day (ppd.) was also calculated as shown in table 2. The results shown in the table revealed that 0.335kg ppd. (78.67%) of food waste, 0.037kg ppd. (8.64%) of plastic and rubber, 0.016kg ppd. (3.66%) of paper, 0.017kg ppd. (4.10% of metal, 0.012kg ppd. (2.82%) glass and 2.10% of other waste is generated in the metropolis.

The value of food waste, the highest from the study in Lagos metropolis consist mainly of vegetables and meal scraps and scraps associated with preparation of food compared favourably with residential waste for some large towns and cities with similar residential characteristics in developing countries such as Gwadalajara, in Mexico with about 52.9% (Gerardo, Et al, 2001) and Nsuka in Enugu state of Nigeria with about 56% (Ogweleka, 2003). In addition when the value obtained in Lagos is compared with the value obtained in Benin metropolis and values obtained by other researchers in some other cities in developing countries such as Guyana in southern America with about 72.8% (Zavodska, A

(2003), Katmandu, Capital City of Nepal with about 70-80% (Alam, Et.al. 2006), and Mumbai in Indian with about 70% (Beukering, Et.al., 1996), (Sashi, 2003) the value is a little lower. The food waste content of residential waste realised from this study is very high because of the heavy dependent on home prepared meals. When compared to the food waste found in the USA of about 11% (USEPA, 2000) the cultural difference stands out (Zavodska, 2003).

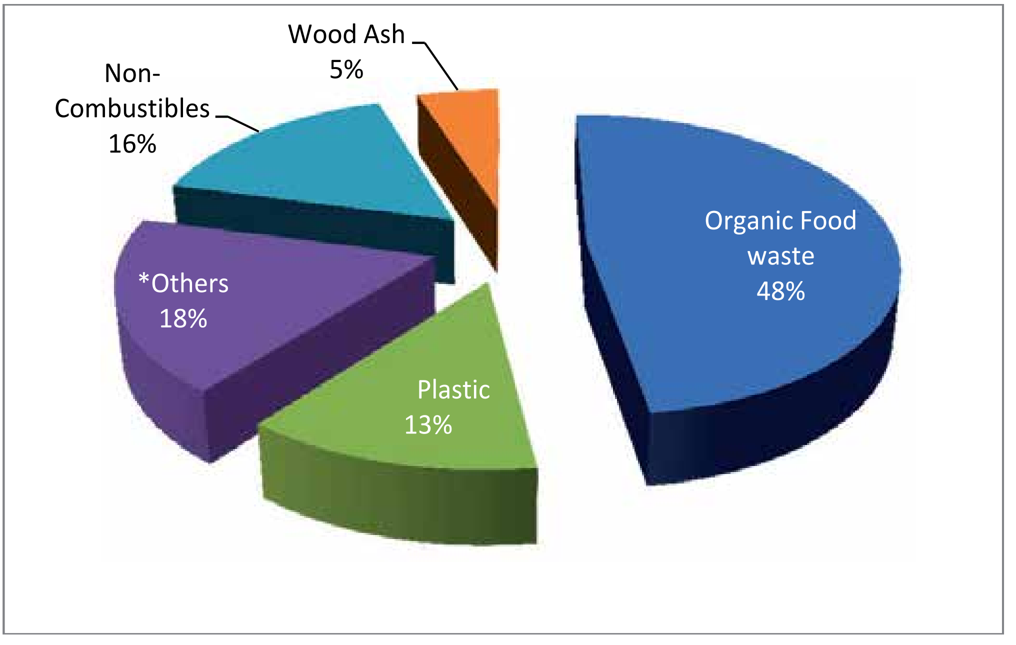

Fig. 2. Waste generation Data for Lagos metropolis

Type of solid waste	Weight (Kg)	% Components
Food waste	0.334	78.59
Plastic/Rubber	0.037	8.65
Paper	0.016	3.67
Metal Waste	0.017	4.11
Glass	0.012	2.83
Unclassified Combustibles	0.003	0.78
Special Waste (Ash)	0.003	0.67
Unclassified Incombustibles(Ceramics)	0.003	0.65
Total Soild Waste Ppd	0.425	

Table 2. Average component of solid waste generated per person per day in Oredo LGA

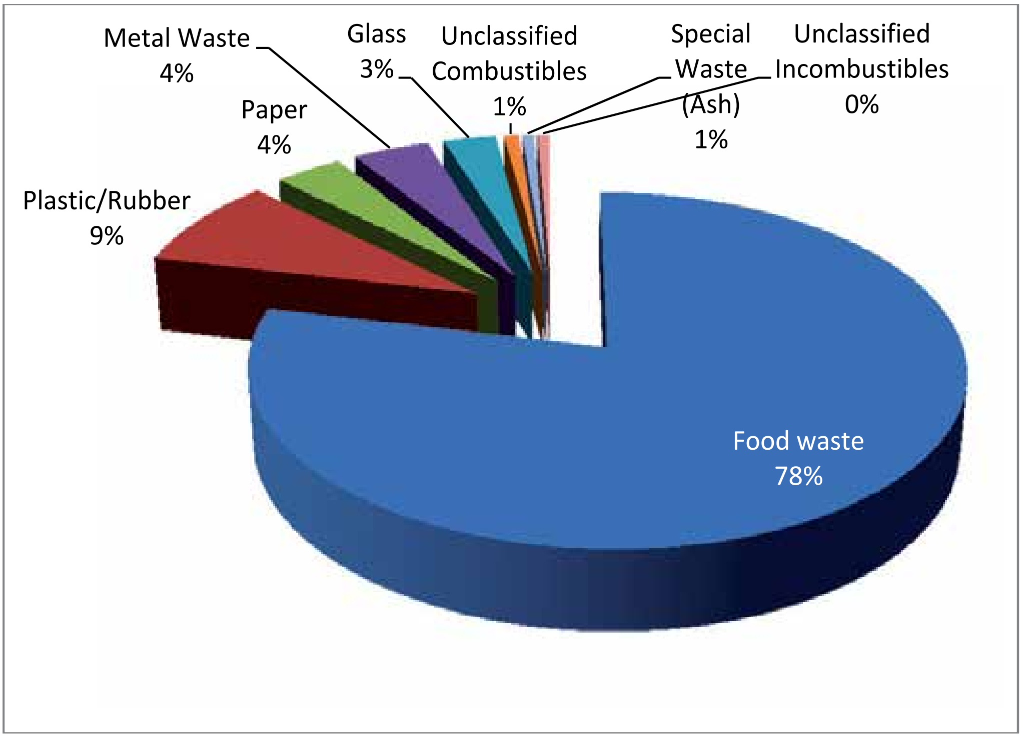

Fig. 3. Average % component of household solid waste generated in Benin metropolis

City	Food waste Percent (%)
Lagos	47
Benin metropolis, Nigeria	78.30
Georgetown, Guyana	72.8
Katmandu, Nepal	70-80
Enugu, Nigeria	56
Gwadalajara, Mexico	52.9
Mumbai, Indian	70
Kano, Nigeria	43
Tianjim, China	58.4
USA	11
Onicha, Nigeria	24-36 (Average 30.67)

Table 3. Food waste in different Countries

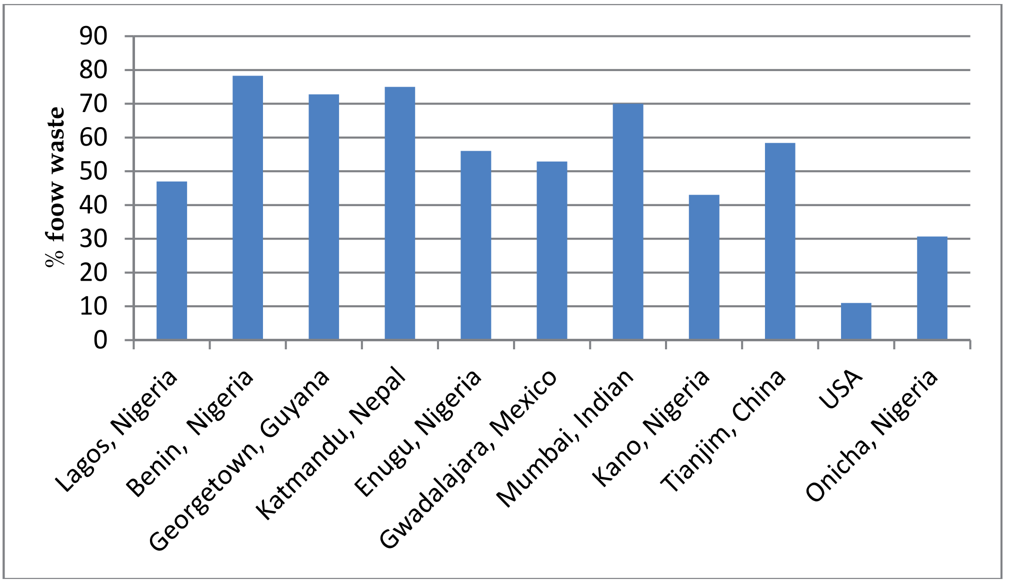

Fig. 4. Food waste in different Countries

In addition a total of 2323.93kg of commercial waste was measured in Benin metropolis and this gave an average of 44.96% of Garbage, 25.43% of plastic waste, 14.27% of paper waste, 3.21% of metal waste, 3.89% of glass, 3.39% and 8.24% of other waste (ceramics, foam, clothes etc) in Benin Metropolis. The percentage of plastics and paper waste are quite high for commercial waste. This can be well understood if we consider the business activities in Benin metropolis which involve sale of products packaged with plastic and paper material and the use material packaged with plastic and paper materials for services. The recent upsurge in the production of table water in Nigeria has brought with it the problem of how to contain the sudden rise in the polythene sachets that are discarded daily after the water has been drunk. Thus, polythene sachets have increased the volume of various waste plastic products requiring attention in Nigeria.

8.4 Solid waste storage and collection

In Lagos and Benin metropolis external waste bins fig. 5 are generally used for storing waste before they were collected for disposal. Small waste bins are also used internally to store the waste before transferring them to the external waste bin for storage for collection by the waste disposal agents. Three private waste disposal agents – Private Sector Participators (PSP) in the solid waste disposal sector were registered in Mushin LGA. Other means of waste disposal are illegal and in some cases punishable. Data shown in table 5 indicates that about 69.09% of household have their waste collected by private solid waste disposal agents for disposal, 5% disposes off their waste by themselves and the 26.36% patronize hand carts in Mushin LGA (see fig. 6). It is imperative to note at this point that hand carts are not registered PSP waste disposal agents and that the areas where hand carts are largely patonised are areas with poor access roads. In addition 51.81% of the household sampled had their solid waste collected for disposal once a week, 22.73% had their waste collected for disposal twice a week and 25.46% had their solid waste collected for disposal twice month. The cart pushers are one of the disposal agents that service the group that had their waste collected for disposal by disposal agents twice a week. They do not have external temporary solid waste storage bin where solid waste can be store before the disposal agents comes to collect them. Hence when there is delay in the cart pushers turning up to collect the waste, members of the household takes the waste to any piece of unused land or throw then into drainage system. Worst still during the raining season they throw their solid waste into the flood water and the waste eventually find their way into the drainage system as shown in fig.7.

Fig. 5. External bins for storing solid waste before collection by disposal agents

Fig. 6. A hand cart Pusher carrying domestic solid waste for disposal

Fig. 7. Solid waste accumulated in the entrance of a manhole of a drainage system

Number of Household Sampled	Collectors and Disposal agents					Collection Frequency				
	Private agent	Self	Cart Pushers	Local Govt Council	LAWMA	Daily	Once a week	Twice a week	Once in 2 wks	Once a month
110	76	5	29	Nil	Nil	Nil	57	25	28	Nil
Percentage	69.1	4.6	26.4				51.9	22.7	25.5	

Table 4. Household responses on collection of solid waste from their homes for disposal in Lagos metropolis

8.5 Solid waste source segregation

In developing countries solid waste source segregation awareness is very poor. In Lagos and Benin metropolises solid waste source segregation is not practiced by residents and commercial operators. A pilot study carried out in Benin metropolis showed that the residents have poor attitude to solid waste source sorting. The results shown in table 5 indicated that in the week one of the study, 43% of the participants achieved 100% source segregation of biodegradable waste. In weeks 2 and 3 the number of households that achieved 100% source segregation of food waste increased to about 52% and 60%, respective. This was as a result of serious sensitization of the households on the benefits that will be realized from source sorting of solid waste. In this case no personal benefit accrued to the generators, may be in the form of discount on solid waste disposal charges. Thereafter, there was a decline in the numbers of households that achieved 100% segregation of the waste and the participant started asking for what were their benefits from the work and declined on further participation. This implies that for sustainability of the process there must be mutual benefit from the waste for the managers of the solid waste and the generator of the waste.

It is believed that if discounts are granted on disposal service charges, more generators will achieve 100% separation of biodegradables and plastic waste at source as the discount will represent the benefit of the generator on the segregation exercise. Considering the results from the pilot study on source segregation, two waste bin source segregation of biodegradable waste and other waste (metal, paper, plastics etc) will be much more effective at the start of the system. This will gradually be increased to three waste bins for biodegradable, plastics and other waste with time.

Week	% of households with 100% separation	
	Food waste	Plastic waste
Week 1	43.48	0.00
Week 2	52.17	0.00
Week 3	60.87	34.78
Week 4	50.12	31.45
Week 5	38.10	28.26
Week 6	46.08	28.5

Table 5. Percentage Cooperation of households in each waste Bin in Benin metropolis solid waste management

8.6 Land fill site

Landfill is the common practice in the Benin and Lagos metropolis. Closed mining sites were converted to solid waste dumpsite without preparation for use as solid waste landfill site. There are three approved dump sites in the Lagos metropolis. These land fill sites have weight bridges at the gate house of the site. When vehicles carrying solid waste get to the gate their weights were taken at entry and exit. The difference between their weights on entry and exit were determined and recorded. Table 6 shows the records of weekly average solid waste delivered at the three approved waste dumpsite in the Lagos metropolis in 2006. The table showed that an average of 12,940.15 metric tons of solid waste was delivered at the three approved dumpsite per week. The 2006 national census puts the population of Mushin Local government area at 633,009 people. This population will therefore result to the generation of 360.8 metric tons of solid waste per day and 2525.7 metric tons per week of domestic solid waste alone from the Mushin Local Government. This value is a far cry compare to the total average weekly solid waste (commercial and domestic) of about 1753 metric tons delivered at the approved dump site from Mushin LGA. This therefore explains why solid waste is eventually dumped at illegal waste dump and thrown around in street corners.

In contrast the Benin metropolis had eight approved dump sites. At the time of this study but only two were operational – Iguomo and Uzebu land fill site. There were neither gate houses nor measuring instruments at the sites. Solid waste was dump indiscriminately at the site. However an experiment was carried out to determine the solid waste delivered at these dumpsites during the period of the study. The result from the experiment showed that a daily average of 33.61 metric tons of solid waste was delivered at the Iguomo dump site and 226.40 metric tons of solid waste was delivered at the Uzebu dump site. Table 5 shows the result of the site-specific studies in Benin metropolis and the result revealed that an average of 0.425Kg per person per day is generated from Benin metropolis. The population census put the population in the Benin metropolis at 1085676 people in 2006. This will give a total solid waste

generated from residential site of about 461.41 metric tons per day. When this value is compare with the total of 260.00 metric tons obtained from the experimental determination of solid waste delivered at the dump site, we have a short fall of 201.41 metric tons of solid waste from residential site. It should be noted that solid waste is delivered to the dump site from all the source of solid waste – domestic, commercial and industrial sites in the metropolis. This therefore explains the reason why solid waste also is seen littered all around in the metropolis.
The current method of solid waste management at the landfill site is simple. The waste disposal trucks and other vehicle that deliver waste to the site drive into the dumpsite through the access road and dump their waste. The workers at the site use shovel to manually push the waste from the road and try to spread them as much as their strength can go. This, of course is a Herculean task. Thereafter, scavengers descend on the waste to pickup recyclable materials for sale and, open air incineration without pollution control is also carried out on the waste for volume reduction (see fig. 8)
Environmental health is very important in location of landfill sites. One of the two functional landfill sites in Benin metropolis is located by a stream of water. This of course can cause eutrophication. Hence there is a serious indication of adverse effect on people in the metropolis as the stream is one of the sources of water to the people nearby.

Date/ Period	Abule-Egba (Metric tones)	Soulous (Metric tones)	Olushosun (Metric tones)	Weekly Total (Metric tones)
Week 1	246	1459	8871.6	10,576.6
Week 2	320	1946	9224	11,490
Week 3	690	2051	10608	13,349
Week 4	1354	2058	13541	16,345
Monthly Total	2610	7514	42244.6	51760.6

(Source: LAWMA Solid waste Record)

Table 6. Average Weekly solid waste delivered at the approved dumpsite by the waste management agents in 2006.

Fig. 8.a Open air incineration of solid waste at dump site.

Fig. 8.b Open air incineration and scavenging activities at the dumpsites

Period	Solid waste(Metric tones)
Week 1	1,749
Week 2	1,595
Week 3	1,848
Week 4	1,815
Monthly Total	7,007

(Source: LAWMA Solid waste Record)

Table 7. Average Weekly metric tons of solid waste delivered at the approved dumpsites From Mushin LGA by the waste management agents.

Number of Household Sampled	Collectors and Disposal agents					Collection Frequency				
	Private agent	Self	Cart Pushers	Local Govt Council	LAWMA	Daily	Once a week	Twice a week	Once in 2 wks	Once a month
110	76	5	29	Nil	Nil	Nil	57	25	28	Nil
Percentage	69.10	4.55	26.36				51.81	22.73	25.46	

Table 8. Household responses on collection of solid waste from their homes for disposal

8.7 The solid waste chain in Nigeria

Solid waste management in Lagos and Benin metropolis revealed a serious deviation from the solid waste chain shown in fig. 2. Solid waste was seen in huge heaps in illegal solid waste dump site, in the open market place, around home and in drainage systems. And this has resulted to serious community environmental health crisis in the metropolis such as water flooding and diseased epidemic. The solid waste chain obtained from the studies is represented in fig.9.

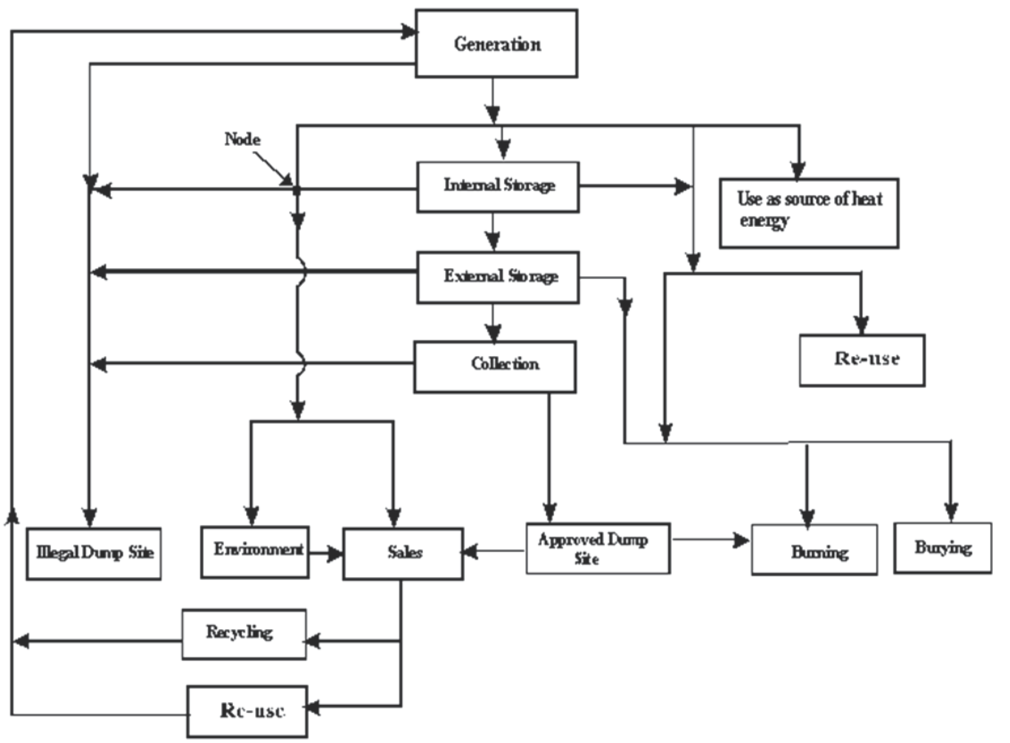

Fig. 9. Solid waste chain in large towns and cities in Nigeria

8.8 Terms in the solid waste chain

8.8.1 Generation

Generation refers to the source of solid waste. In Nigeria municipal solid waste is generally made up of waste generated from domestic and commercial activities.

8.8.2 Internal storage

Internal storage refers to storage of waste within the house or office at source. The study revealed that small waste bin such as perforated bins, small plastic bags, empty paint drums, plastic and metal buckets etc were used for storing waste internally at source. The wastes are usually stored in the internal waste bins until the bins are filled up and then transferred to the external waste bins.

8.8.3 Re-use

In most cases, the generator keeps solid waste items such as bottles and beverage cans for storing liquids (kerosene, cooking oil etc), salt and other food items. In addition other solid waste items such as paint cans, buckets and drums were also kept for storing water in the home and commercial unit. After a while, the items find their ways back to the waste stream when they become broken or old.

8.8.4 External storage

External storage is the temporary storage of the wastes in bins usually located outside the house for domestic source and outside the premises for commercial source. The types of external waste bins in use in Lagos and Benin metropolis include: Metal and plastic drums, Bins constructed with blocks, Pits, Sacks, etc.

8.8.5 Collection

This refers to the collection of the solid waste from the generators for disposal by the waste disposal agents. The preliminary study showed that solid wastes were left in the external storage bin for a period of one to four week before collection by the disposal agent in Lagos and Benin metropolis.

8.8.6 Environment

Environment in this context means the surroundings of houses and business premises in the metropolis. These include road sides, drainage system, market areas, etc. Large amount of solid waste are thrown and made to lie around the environment in the metropolis.

8.8.7 Illegal dumpsites

These are unauthorized solid waste dumpsites where solid wastes were dumped indiscriminately by residents in the metropolis. Many unused lands are converted to solid waste dumps illegally. In fact, the waste control agencies are unable to enforce the environmental laws and hence resident do whatever they like with their waste including illegal dumping in the metropolis.

8.8.8 Approved dumpsite

These are authorized landfill sites where solid wastes are dumped and managed by the waste management agencies. At the time of this study, there were eight approved dumpsites in the metropolis, but only two were functional due to neglect by the authorities in charge. Hence, there was indiscriminate dumping of solid waste at the site by disposal agents.

8.8.9 Sales

The generators sell some of their solid waste such as waste bottles which buyers use for various purposes such as storage of vegetable oil and other cooking items. Some item were also picked up by scavengers at the illegal dumpsites and external waste bins for sale. Bottles used for packaging medicine were also sold.

8.8.10 Recycling

Recycling is the process of adding value to the waste to make it economically useful. Some recyclable waste such as nylon and plastic/rubber bags were sorted and sent for recycling in the metropolis. Recycling is at low ebb. The facility used for recycling plastic bags installed in the metropolis was not very functional. A study of the recycling facilities showed that the power consumption of the facility is about 225hp of electrical power. This is too high for small and medium scale enterprises (SME). And the system was poorly designed.

8.8.11 Burning

This is the process of setting the waste on fire and allowing it to burn to aches. This study revealed that uncontrolled open burning of solid waste was practiced by generator and waste management agents around houses, business areas and at the dumpsites in the metropolis.

8.8.12 Burying

Burying is the process of covering the waste in a hole with sand in the ground. Burying of solid waste is wildly practiced in Nigeria. This study revealed that residents dig hole behind their houses to get sand for filling the foundation of their building to damp proof course (DPC) levels during construction. When they move in to live in the houses they bury their solid waste in such holes dug behind their houses during construction.

8.8.13 Use as source of heat energy

Residents of the metropolis burn solid waste to generate heat energy for cooking purpose. For example, some residents in the metropolis go to the wood sawmill industry for collection of wood sawdust which they burn to get heat for cooking their meals. In addition, during the corn season, the boiled corn seller burns the corn curb to get heat for cooking corn for sale.

9. Feasible suggestion for improved solid waste management

1. There is need to pay more attention to the prevention of blocking of water ways. Not only is this an unpleasant sight, it results in flooding of homes, breeding of pathogens and pest. There must be improved litter control in the large town, cities and metropolis. A very good way to promote this is by providing more public waste bins throughout the metropolis and replacing the existing ones when they become old or when they are damaged. If bins are available, then at least people will have the option of using them. Without available waste bins, the only option that the people will have is to throw waste on the ground which is the current practices.
2. It is obvious that funding is a major constraint in solid waste management; hence special attention should be paid to financial planning by the Waste management authorities in the metropolis. The government should create special charges that will be paid by residents and business operators in the metropolis. And these charges should be dedicated to management of solid waste in general in the metropolis. The collection of these charges should be planned is such a way that the difficulties associated with the collection of levies and charges currently will be eliminated.
3. Many officers in charge of solid waste management, particularly at the local and State Waste Management authorities and other agencies handling the issues of waste, have little or no technical background or training in environmental engineering or management. In fact all the problems that the solid waste Management system is faced with are exacerbated by the lack of trained personnel. This includes workers in all ranks, from the administrators to the refuse men. There is no formal training programme and communication is poor. Training for personnel is important. Adequately trained managers, supervisors and foremen in both collection and final disposal site positions are important for a smooth running operation and operational data collection. New policies should be created for the management of solid waste in the metropolis which will indicate the training requirement for various positions in the

solid waste management system and these new policies should be officially implemented by the responsible body

4. Presently, public awareness on solid waste issues is very poor. Public awareness needs to be improved. This can be achieved using various means such as integration of environmental education with emphasis on solid waste into school curricula beginning with primary/elementary school. Other factor that could be applied includes news releases, letters to the editor, news articles, newsletter articles, speeches, guest on the radio and local TV programmes, messages in churches and mosque, notices in church and mosque bulletins. These are plausible and financially feasible methods that can be used for increasing public awareness on solid waste management.
5. Presently, landfill appears to be a method that will continue to be employed, hence funding should therefore be improved for provision of landfill liners. Effort should also be made to obtain liners from foreign sources as donations even if they are not the best ones. This should also apply to leachate and gas collection system even if they are older technology. It is better to have older technology than no technology at all (Zavodska, 2003).
6. Interview with the workers of the solid waste disposal agents indicated that protective gears were not provided for them. Protective gears should be made available for the solid waste collection workers and workers at the landfill sites. Heavy boots and heavy-duty hand gloves should be provided to all as the biggest risk that they are exposed to is stepping onto object that could penetrate their legs and also sharp objects could scratch their hands when picking them up.

10. Conclusion

This has been primarily concerned with the assessment of the status of municipal solid waste with particular reference to Nigeria. The municipal solid waste load assessment in terms of types and quantity generated was carried out in Lagos and Benin metropolises in Nigeria. This is what is termed as characterization of municipal solid waste. This is the case due to the fact that knowing the expected waste load is the first step in any solid waste management design project. The study showed that 0.425kg of solid waste is generated per person per day (ppd) in Benin metropolis and 0.57kg per person per day is generated in Lagos metropolis. The study also showed that over 20% of recyclable solid waste is generated in from domestic source of solid waste in Nigeria. Assessment of the Solid waste management in Lagos and Benin metropolis revealed a serious deviation from the conventional solid waste chain. The landfill situation is in bad state as they were not prepared for sanitary landfill.Considering the results obtained from the study there is need for urgent attention to be paid to the issues of solid waste in Nigeria as it poses serious environmental threat.

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3

Municipal Solid Waste Management in Developing Countries: Future Challenges and Possible Opportunities

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1. Introduction

Most developing and least developed countries are currently facing serious development challenges that may enhanced if same old traditional development plans are still in persistence. In the wake of the recently emerged global economic crises development challenges are expected to increasing as a result of the adverse impact on the capabilities of developed countries to offer needed assistance to developing countries. Since the 1992 Rio Earth Summit and later the adoption of the Rio Declaration and Agenda 21 and following the declaration and adoption of the millennium development goals (MDG) in 2000 things on the ground have not significantly improved. The United Nations' recently released a report that shows that most developing and least developed countries are far from reaching the MDG targets set for the year 2015 (UN, 2010). The impacts are expected to exceed the continuous widening of the gap between the developed and developing countries to the extent that might badly affect sustainable development. After more than two decades from the adoption of the notion "*sustainable development*", it could be claimed the notion was portrayed in different ways when comparing developed to developing countries. Developed countries treat Sustainable development as an environmental concept placing the emphasis on inter-generational equity focusing on future needs (Carter, 2001), while most developing countries are placing emphasis on intra-generational equity focusing at present needs which are often social and economic ones. Such different portrays played significant role in shaping the capabilities of developing countries to meet the sustainable development challenges they are facing and consequently in widening the gap between developing and developed nations.
In this context it is important to shed lights on major challenges facing sustainable development in several developing countries. These could be summarized as follows:

- **Population growth**: the Population Reference Bureau (PRB) projected the 2050 world population to a range from 9.15 – 9.51 billion with different decrease in fertility rates in many developing and least developed countries (Bremner et al, 2010). The largest percentage increase by 2050 will be in Africa where population is expected to jump to more than 2 billion. Asia with 4.2 billion will likely experience smaller proportional increase than Africa, however this depends on China and India, where both populations accounts for about 60% of total Asia's pollution. Latin America and the Caribbean are expected to experience the smallest proportional growth due to fertility declines in several of its largest countries, such as Brazil and Mexico. The age structure of most countries in Africa, Asia, and Latin America favors young population at working age that, if well managed, could be the driving force behind economic growth prosperity or, if improperly managed, could adversely impact socio-economic growth. The expected decrease in fertility and increase in youth will lead to the "demographic dividend" where youth populations become older and have fewer children of that previous generations leading to a bulge in the working age population. This represent a window of opportunity for developing countries to save money on healthcare and social services and to invest more on technology and capacity buildings to strengthen economy and to cope with future aging of the population.

- **Public health**: The World Health Organization report on World Health and MDG (WHO, 2010) showed that over the last decades average annual mortality rate in children under 5 years old in most developing countries has fallen by a range of percentages from 1.7% (in Eastern Mediterranean Region), 1.8% (in Africa) to 3.8% in south and east Asia region. Despite these encouraging trends, the report indicated several health problems that still in persistence such as maternal mortality and HIV/AIDS. The estimated number of death caused by malaria in 2008 is 863000 with 243 million estimated cases. Health implications due to poor sanitation facilities are considered very serious. The same report estimated that in 2008 over 2.5 billion people were not using proper sanitation facilities resulting in high level of environmental contamination and exposure to risks of microbial infections. Death caused by non-communicable diseases or injuries in developing countries totaled in 2004 to 33 million. The absence of adequate healthcare systems will still adversely affect the public health conditions. Health problems caused by poor hygienic and sanitation conditions require improving and upgrading infrastructure for waste management and introducing the integrated management approaches.
- **Vulnerability to climate change:** the Intergovernmental Panel on Climate Change (IPCC, 2007) defined vulnerability of people as their propensity to be harmed due to their exposure to stresses including climate stress. It is believed that the continuous increase emissions of greenhouse gases (GHG) several decades ago due to human anthropogenic activities resulted in the global climate change, which turned to be the most serious challenge facing development in the 21st century. The accumulation of GHG emissions in the atmosphere, in particular carbon dioxide (CO_2) and Methane (CH_4) is believed to be responsible for the global warming and the associate frequent occurrence of extreme climate events. CO_2 concentration in the atmosphere has risen to 391 ppm by end 2010[1]; an increase of about 6% compared to records of 2000. The 2010 world energy statistics (BP, 2010) show that 44% of total CO_2 emission comes from 17% of the world total population (developed nations) while the rest 83% of the world population (developing and least developed) contributes to the rest half of the total emissions (figure 1). climate events such as floods, storms, droughts, hurricanes, etc., and the rise of sea level resulted from melting of the glacier covers are also observed (IPCC, 2007). As vulnerability to climate change is shaped by factors such as the population dynamics and economic status as well as adaptation measures such as appropriate norms and codes, it is likely that people in developing and least developed countries will be more vulnerable compared to those in developed countries. Adverse

[1] In http://co2now.org

direct impacts on health, land-use, agricultural productivity, water resources availability, etc. may further heightened and indirectly impact population, economy, and social-economic growths butting extra burdens of development processes on developing countries.

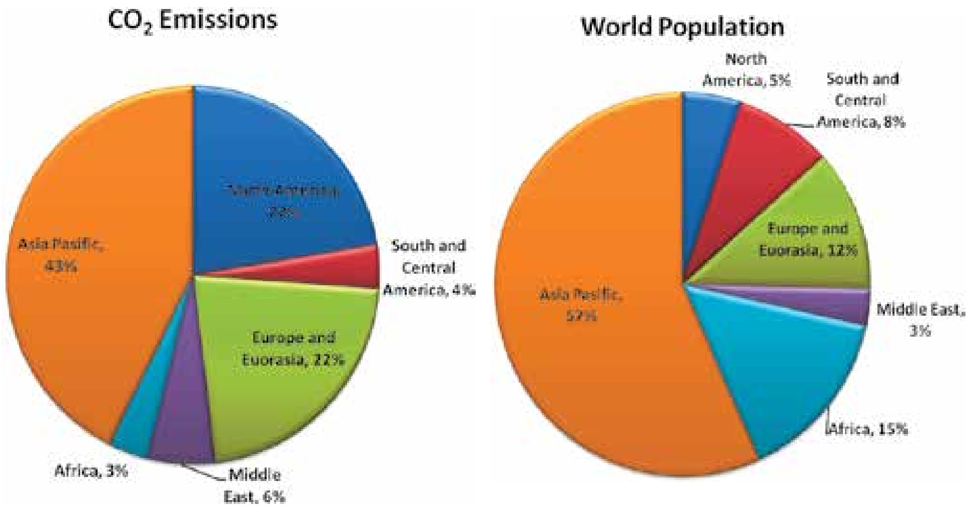

Fig. 1. Comparison of World Population and CO_2 emission

- **Human development and the economical growth**: the 1990 Human Development Report states that "People are the real wealth of the nation." (UNDP, 1990) putting people at the center of the development process, which objective is to create an enabling environment for people to enjoy long, healthy and creative lives. The UNDP recently released the 2010 human development report (UNDP, 2010) which indicated that developed countries have recorded considerable economical growth in the last four decades compared to developing countries. The un-attainable convergence in income between the developed and developing nations resulted in setbacks, particularly in service such as health and education and consequently on human development and on the sustainable development process as a whole.

2. Solid waste in developed and least developed countries in the context of development

The last three decades witnessed the development in urban areas over rural ones in a process called urbanization. Growth of urbanization is much more in developing countries than the developed countries (figure 2) to the extent that it became a trend characterizing several developed and even least developed countries. Growth in urbanization is coupled with the growth of population living in urban areas. In e.g. China, urbanization led to increase in urban population to about 35% percent of its total population with annual growth in urban population of about 4%. Similarly, it is anticipated that by 2025 Asian urban population will reach 50% of the total population; and probably more. This expected increase will cause major shift in the distribution of the countries' populations and will lead to the expansion of urban boundaries (World Bank, 2003).

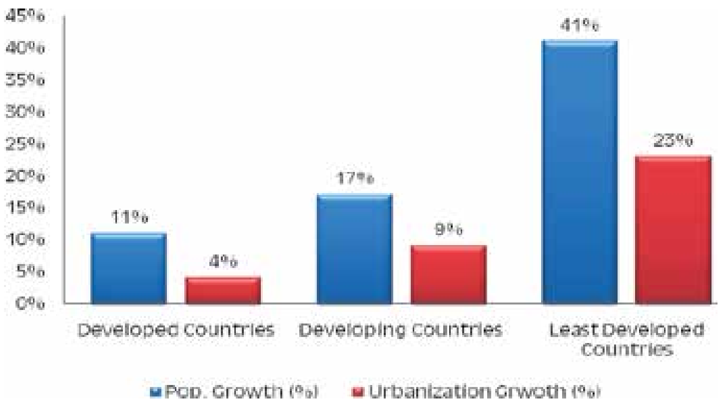

Fig. 2. Population and urbanization grwoth (1990-2010)

The reality is that the growth in urbanization does not always mean improving situations, including sectors developments. In the recently published 2010 human development report (UNDP, 2010), indicators that describe the accessibility to water and sanitations in developing and least developed countries are not encouraging at all. It is found that an average of about 45% of countries' populations are lacking proper sanitation infrastructures, and an average of 20% are lacking proper accessibility to water. In addition, the report showed that the percentage of populations living on degraded land is increasing to an average that exceeds 15%. Hence, the consequence of the unplanned urbanization growth will definitely lead to huge problems on governments especially for meeting the increasing demand for proper and healthy municipal services. The growth will result in increase in the quantity and complexity of the generated wastes and overburdens, including solid wastes, and in particular municipal solid waste (MSW). MSW includes materials discarded for disposal by households, including single and multifamily residences, and industrial waste from canteens/restaurants and hotels and motels and from commercial and industrial entities essentially the same as waste normally generated by households and collected and disposed by normal municipal solid waste collection services. Such MSW is considered a problem that having impacts on the environment and the public health if not properly managed. Comparing conditions related to MSW management in developed and developing countries brings indicators that quantify the problem. Considering the MSW generated in general, its main constituents are to some extent similar throughout the world, but the quantity generated, the density and the proportion of streams vary widely from country to country depending mainly on the level of income and lifestyle, culture and tradition, geographic location and dominant weather conditions. Low income countries with yearly per capita GDP that does not exceed US$ 5000 have the lowest MSW generation rates, which are in the range 0.3 – 0.9 kg/capita/day. The increase in per capita daily generated waste is found linearly proportion to the per capita GDP. In high income countries it reaches a range of 1.4 – 2.0 kg/capita/day. Figure 3 shows the linearly coupled GDP to Waste generation rate diagram with examples from countries of low, medium, and high incomes.

Another element that characterizes differences between the generated MSW in low and high income countries (developed and most developing countries) is the percentage composition of MSW constituents. There, the lifestyle of peoples decisively characterizes the percentage composition where organic waste stream and overburden form more than 50% of the total generated MSW. This is the opposite in high income countries, where lifestyle favors fewer homes cooking, relying mainly on the readymade backed food. This is reflected in the figures that represent the percentage of organic waste stream which does not exceed an average of 30% of the total generated waste and that more packing material characterizes the MSW. Figure 4 shows the differences in parentage compositions of MSW between high income countries (developed and some developing), medium income countries (most developing) and low incomes countries (some developing and least developed countries).

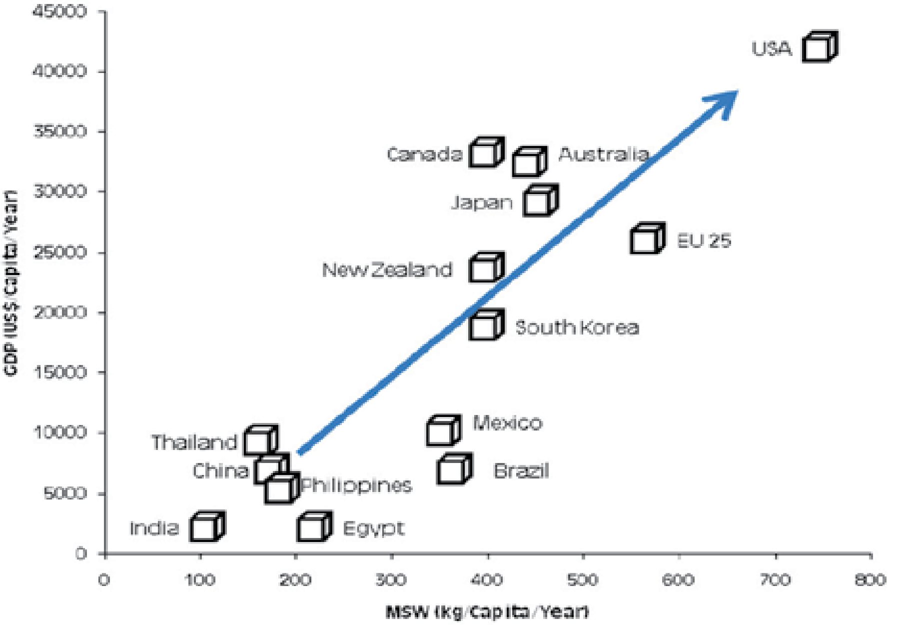

Fig. 3. Countries income and the rate of generated MSW (UNDESA, 2010)

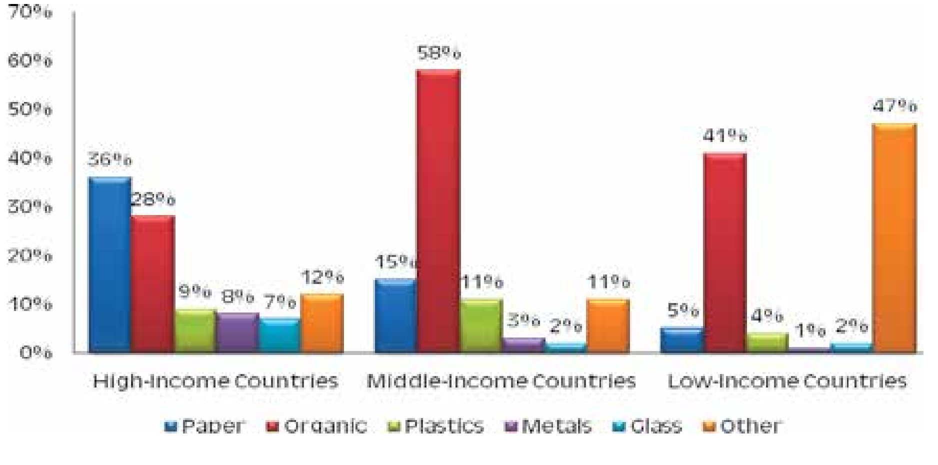

Fig. 4. Charactersitisc of MSW streams depending on income (UNDESA, 2010)

In urban areas of most developing and least developed countries generated MSW is at best collected and dumped in arbitrary dump sites that mostly lack the appropriate norms. Such disposal requires collecting, transport and dumping into the nearest open space area. In other countries MSW is dumped into water bodies and wetland and part of the waste is burned to reduce its volume. Such practices have their adverse environmental impacts ranging from polluting the natural resources and the ecology to the creation of health problems which might turn into long-term public health problems. Studies conducted in the last decade in several developing countries showed that same old non-environmental sound practise are still used. Although lots of significant efforts have been done in the last few decades in many developing countries supported technically and financially by developed countries and international organizations, substantial reforms in the management of MSW are still not attained. This is due to the fact that frameworks recommended where mostly similar to that adopted in developed countries but without seriously addressing the socio-economic differences between the developed and developing countries.

In the Middle East there are countries of high income (e.g. Saudi Arabia, UAE, Kuwait, Qatar, and Israel) and other of middle and low income countries. The per capita rate of generated MSW shown in figure 5 (Kanbour, 1997, Mashaa'n et al, 1997, Al-Yousfi, 2002, METAP, 2004, Israel MEP, 2010) is rather diverse but reflects the country's income level. However, when looking at the percentage composition of MSW constituencies shown in figure 6 (WH, 1995, Al-Yousfi, 2003,METAP, 2004, Israel MEP, 2010), it could be realized that the major MSW stream in Arab countries of the Middle East is organic. This is primarily due to the fact that these countries share a common lifestyle and eating habits.

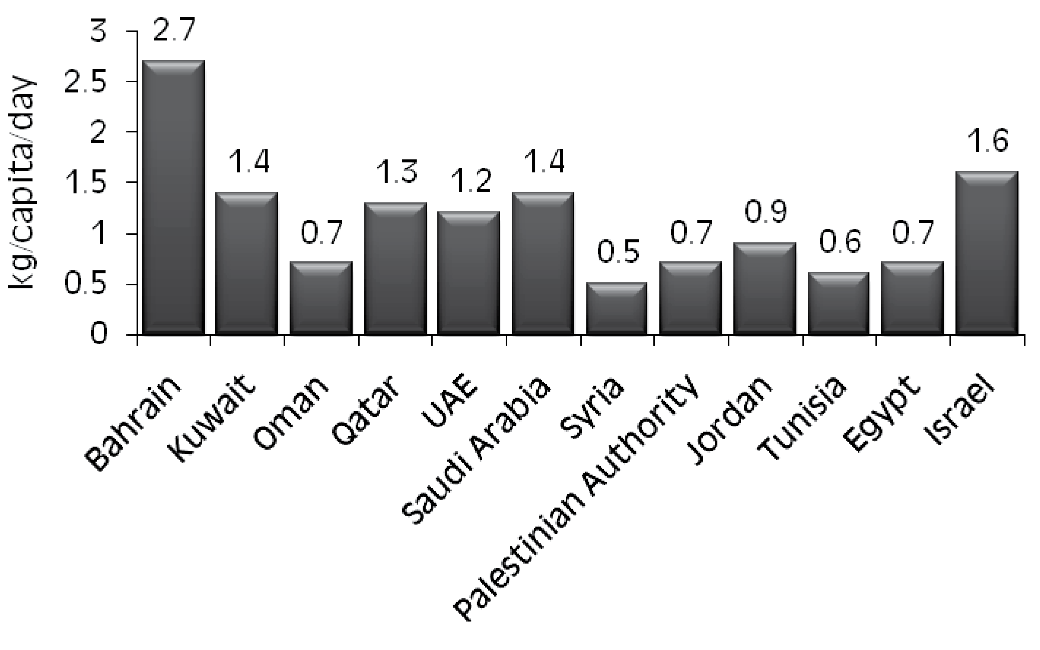

Fig. 5. MSW generation rates in countries in the Middle East

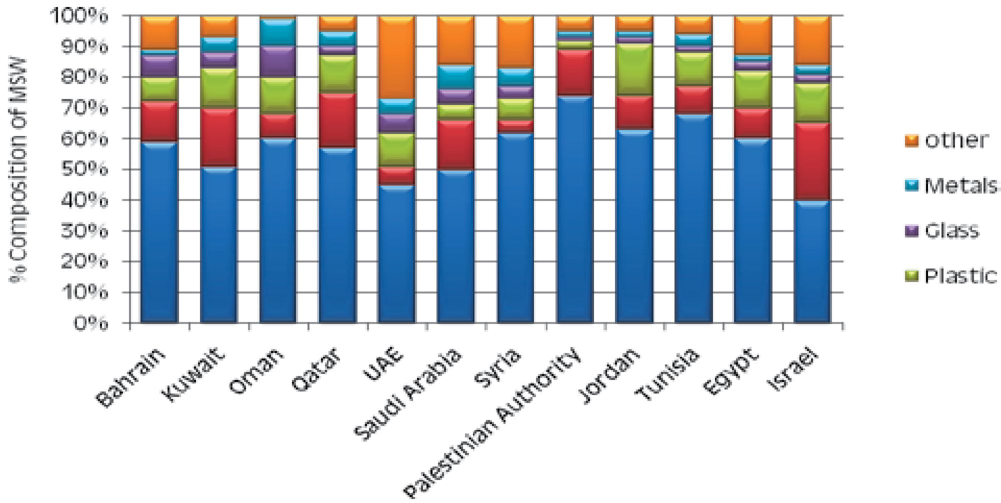

Fig. 6. Percentage compostion of MSW in some Middle East countries

Countries of medium and low income in southeast Asia have simillar per capita rate of generation of MSW but they have different percentage composition of generated MSW streatms. Figure 7 (Glawe, et al, 2005) shows the diversity compostion of MSW where organic overburden dominates in most countries.

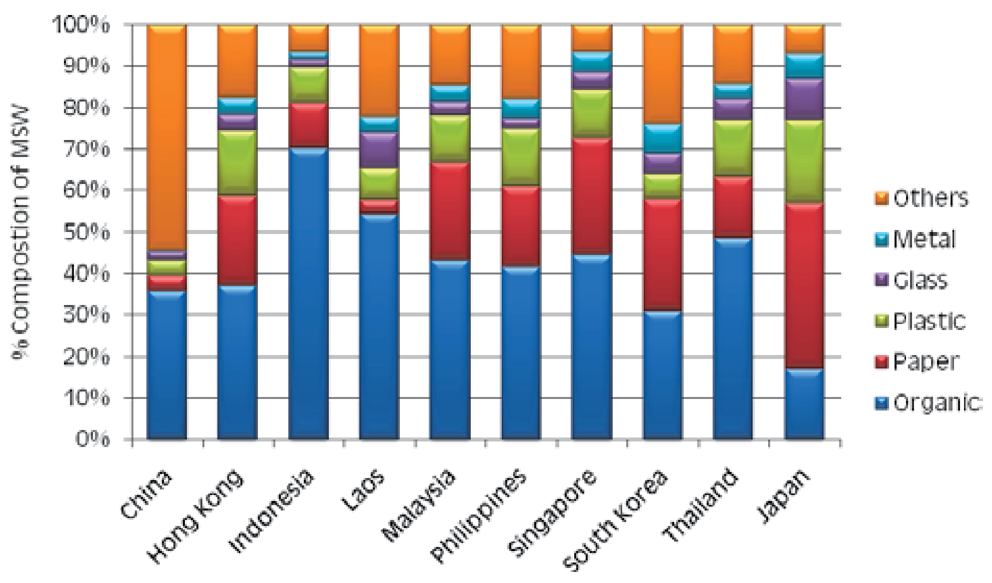

Fig. 7. Percentage compostion of MSW in some Asian countries

It is clear that most of the generated MSW constituencies in most developing countries are decomposable and recyclable. If properly managed, such MSW would provide high opportunities for the development of the socio-economy of the countries. However, the fact

is unfortunately the opposite as the MSW remains a socio-economic that faces many problems.
There are diversities of management options of MSW in the different developing countries. In Egypt, which is an African Middle East Arab country 75% of the MSW is generated in urban areas. Total estimated MSW for 2025 is expected to reach 33 million tons for a growth rate of 3.2% based on 2001 records. Collection services cover less that 30% of urban and rural areas and the rest are disadvantaged. A portion of 8% of the total collected MSW is sent to compost plant but the rest is sent to dump sites scattered in the country open spaces posing high risk to public health and the environment (METAP, 2004). This is very similar to the situation in many developing and least developed countries of the region such as Syria, Jordan, and the Palestinian Authority as well as countries in Southeast Asia, Africa, and Latin America. Israel on the other side has generated in 2006 around 6 million tons of MSW and Industrial waste from urban and rural areas. The solid waste services cover almost all regions in the country. There are more than 15 state of the art landfills located in different regions in addition to recycling plant where 23% of the total generated waste (i.e. 1.4 million tons) are recycled (Israel MEP, 2010). In the Gulf Arab countries and specifically in the UAE some 25% of the generated MSW in Dubai, Abu Dhabi, and Sharjeh is diverted to compost plants. MSW in other emirates of the UAE is collected and sent to landfills (UAE-ME, 2006). Particularly in Dubai more than 60% of the emitted methane is recovered.
It is clear that the main problem facing the proper management of MSW in many developing countries are the lack of adequate administrative and financial resources. There is no clear reliable framework by which the solid waste sector is administered from the collection, transformation to disposing or treatment phases. This situation is usually coupled with limited investment allocated for the MSW sector with complications of collecting or raising proper service fees. The management activities of MSW are considered public services which are directly controlled by governmental institutions. Such management arrangement is considered weak as it lacks the market mechanisms, and in this case economical incentives cannot be used to improve and develop the MSW management services.
Another related common problem is the absent of effective and comprehensive legislative frameworks governing the solid waste sector and the inadequate enforcement mechanisms, which are no less important than the legislations themselves. Such short comings in the management of MSW create gabs and intensify the problems. Standards and norms are also critical for the implementation of the legislative frameworks especially that concern the setting, design, and operation of the landfills and the dealing with possible hazardous and healthcare wastes. In many developing countries where financial resources exist, shortcomings are found in both the human and organizational capacities. In Palestinian Authority donors have spent considerable amount of funds for rehabilitating devastated infrastructure and for providing facilities for the collection, transportation, and disposal of solid waste but they have compromised building the needed institutional and human capacities and raising the public awareness (Khatib and Al-Khateeb, 2009). This created a problem that was only recently rectified as will be elaborated in the successive sections.
The last significant problem related to management of MSW is the availability of the significant amount of accurate background data and information on the status of solid waste, including MSW, such as rate of generation of different solid waste constituencies, assessment of natural resources and land-use, collection and transportation needs, scenarios of treatment, growth scenarios of solid waste which is linked to several driving forces. Data and information are the crucial elements for developing MSW management system including the adequate monitoring of the sector.

To overcome the a.m. main problems, the following prerequisites should be addressed and dealt with:

- Institutional set up,
- Human awareness and capacities,
- Proper standards, laws, guidelines and norms,
- Proper infrastructures,

Enabling management of the generated solid waste in an adequate approach will mitigate any adverse impacts to the environment, natural resources, and the public health, which are obviously the main aims of MSW management. In developed countries integral management of MSW has reached an advanced stage where MSW are reduced in amount at sources, i.e. before collection. This has been achieved with both, the intervention of the available technology and the public awareness. Technology has provided better design for the consumable products with less material in size, weight, and packaging. In addition same technology offered the possibility of at source re-using products' packaging. The benefits of this integrated management approach are many, including:

- Conservation of natural resources,
- Reduced amount of waste to be recycled or transported to landfills or waste recycling facilities. ,
- Decreased air pollution and the production of greenhouse gases,
- Reduced toxicity of waste, and
- Reduced costs of waste collection and disposal.

Developed countries have also succeeded in applying different treatment and re-use methods for the generated MSW, including; recycling, composting, and energy recovery, in addition to the disposing of the waste in proper landfills.

It should be emphasized however, that as long as impacts of MSW are properly mitigated, there are no overall 'best' or 'worst' approach and that the conditions in persistence and the identified driving forces for any country; whether developed, developing or least developed, should provide baselines for the best integrated feasible approach.

3. The integrated sustainable solid waste management approach

The integrated sustainable solid waste management (ISSWM) was first developed in mid 1980s by a Dutch NGO called WASTE[2] and further developed in 1990s by the Collaborative Working Group on Solid Waste Management in Low- and Middle-Income Countries[3] (CWG), then it became as a norm. The ISSWM is a system approach that recognizes three main dimensions including stakeholders, elements, and aspects. These dimensions are shown in figure 8.

The stakeholders are the people or organizations participating in solid waste management. This includes the waste generators who use the services, the service providers, the formal and informal private sector dealing with solid waste management, and other local or international institutions. Elements comprises the technical components of the waste management system starting from the generation of solid waste then the collection, transfer and transportation of waste to dumpsites or to treatment plant. Treatment ranges from

[2] http://www.ecosan.nl
[3] http://www.cwgnet.net

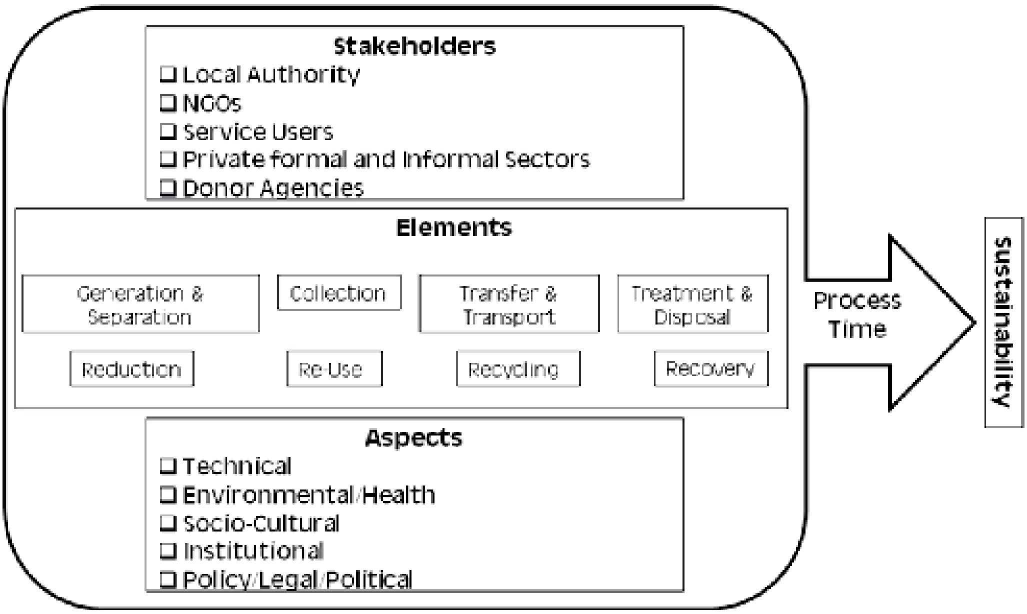

Fig. 8. Integrated Solid Waste Management (Source: http://www.ecosan.nl)

reducing the size of the generated waste to recovery of the waste, in particular the biodegradable component that comprises more than 60% of the total municipal solid waste generated in urban areas of the developing countries. In order that the integrated waste management be sustainable, all required aspects, such as financial, social, institutional, political, legal, and environmental that assesses the feasibility of the management should be addressed in a sustainable way. The different dimensions are interrelated and their linkages institutionally, legally, and economically enable the overall function of the system. It could therefore, be indicated that ISSWM considers MSW management not just a technological system with infrastructure and facilities that facilitate handling and disposal of MSW, but it is a management system that consider and deals with many other elements including the socio-economic settings, the physical environment and growth in public demands and management scenarios.

Several principles of ISSWM were extensively based on contributions of many authors (i.e. Moreno et al., 1999, Coffey, 1996, Schuebeler et al., 1996, van Beukering et al., 1999). These could be summarized based on figure 8 as follows:

- Technological and operational principals should be adapted to the physical setting, local environment, and land use, of the region. Efficient technology should be preferably a local efficient technology that reliably provide spare parts and efficient O&M,
- Environmental and health principles that ensure that any technical setting is clean with the minimal impact on the environment and its elements. This is attained by following the waste management hierarchy preferring options that promote waste reduction and separation at sources,
- Financial principles should ensure "all beneficiaries contribute principle" in which fees are collected for the services and in return relevant government institution contributes

by allocating revenues to MSW. Financial principles should ensure highest productivity of labor relying of capital intensive system and not on labor intensive. Full cost recovery should also be considered,

- Socio-economic principles that permit public in all regions to receive adequate and affordable management system without any adverse health impacts while acknowledging the different economical incomes of beneficiaries,
- Administrative principles necessitate building the capacities of the personal involved in the management of MSW, in addition to encouraging the involvement stakeholders in the planning and implementation of the management activities,
- Policy and legal framework principles, that while support decentralizing of relevant authorities and finance they, at the same time, encourage the involvement of stakeholders including non-governmental organizations and the private sector.

4. Palestinian authority and the newly adopted ISSWM

Palestinian Authority (PA) is considered as a developing entity having many in commons with other developing countries. It is only after the emerging of the PA that development in the Palestinian territories started taking place although it has been heavily retarded after the Israeli re-occupation of the Palestinian areas in 2002 in response to Palestinian second Intifada (Uprising). The solid waste sector is managed by different institutions. In towns and villages, municipalities and village councils are providing the services whereas in refugee camps the United Nations Relief and Works Agency (UNRWA) is taking care of the services. The long-term occupation with its daily harsh measures against Palestinians and their infrastructure have exerted heavy burden on the PA and other responsible institutions for launching a complete development process. In 2009 and with the help of donor countries and institutions, the PA has started preparing its first strategy for integrated solid waste management while facing huge challenges similar to those facing other developing countries and mentioned previously in the chapter. Although the Europeans and World Banks have both supported the sector over the period 1994 until 2010 with a total of US$ 72.274 million (Palestinian Authority, 2010) very little progress has been witnessed. This is due to the fact that the Israeli military incursions into PA areas after 2002 and through until 2006 has left a devastated infrastructure, let alone the Israeli occupation closure policies which prohibit people commuting among Palestinian communities. After 2006 money has been spent as previously mentioned on rehabilitating the destroyed infrastructure and for providing the required facilities but without building the needed institutional and human capacities.

In the effort to describe the solid waste status in the PA, a survey study was conducted in 2002 and 2008 (khatib and Al-Khateeb, 2009) which showed that the daily average per capita generated municipal solid waste is in the range 0.5 – 0.9 kg. This average takes into account communities living in urban, rural, and refugee camps. It was found also that solid waste consists mostly of biodegradable organic waste, a characteristic that agrees well with studies done for other similar developing countries (El-Edghiri, 2002). Later in 2009 same characteristics were reaffirmed by the Palestinian Central Bureau of Statistics (PCBS) which further suggested an annual generated solid waste of one million tons in the PA area.

Due to the importance of the sector on the development process, the PA has in 2009 declared the solid waste management sector as a national priority and therefore, issued the guiding principles for the Palestinian ISSWM which are:

- The principle of sustainable solid waste management that ensures optimal use and protection of the environment,
- Clarity of roles and responsibilities and separation between regulatory, monitoring, and executive duties,
- Facilitated availability of information and the transport exchange among stakeholders involved,
- Transparency of institutional, financial, monitoring, and administration systems,
- Partnership based on integrity and clarity of roles of each stakeholder,
- Recognition of private formal and informal and NGO sectors,
- Transparency in dealing with public complaints,
- Principles of "polluter pays" and "Producer pays"
- Principle of self funding and providing services at reasonable prices,
- Principle of economy scale in planning and developing the services,
- Gradual implementation of initiatives technologies, and new models related to solid waste elements; i.e. reduction, recycling etc.
- Creating incentives to encourage successful practices,
- Compatibility of technology and facilities used in the solid waste management to local conditions and needs,
- Penalty system against parties that do not adhere to appropriate procedures in dealing with solid waste.

These guiding principles where considered in the prepared strategy and where impeded while defining the Palestinian ISSWM strategic objectives, which are:

- An effective legal and organization framework for solid waste services,
- Strong and capable institutions,
- Effective and environmentally-safe management of solid waste services,
- Financially viable and efficient management services and activities,
- Principles and mechanisms suitable for managing medical, hazardous and special wastes,
- More participating and aware community,
- Effective information and monitoring systems.

The implementation of the Palestinian ISSWM has been launched and hopefully the scenario which favors political stability will prevail to ensure the expected outcomes.

5. Conclusion

Developing and least developed countries have no alternative but to plan for a sustainable development processes acknowledging the importance of encountering the problems in persistence and facing the development challenges with an active participation of stakeholders including the public. With the growth in urbanization MSW services is becoming one of the most challenges which if not properly and sustainably dealt with will adversely impact all other development sectors. The best approach for dealing with solid waste sector is by implementing an integrated and sustainable management approach that ensures the good health of the society and the environment and the active participation of the society. An example of implementing the ISSWM approach has recently been initiated by the Palestinian Authority and if political atmosphere permits the adequate implementation of the ISSWM strategy feedback would be of most beneficial to many developing and least developed countries.

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4

International Practices in Solid Waste Management

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1. Introduction

Emergent urbanization and changes in the pattern of life, give rise to generation of increasing quantities of wastes and it's now becoming another threat to our already degraded environment. However, in recent years, many programs were undertaken for the control of urbanization gift in the world because the dumping of industrial wastage without proper treatment, responsible for the lowering of a soil fertility, which increases the amassing of essential and non essential trace metals in the plants. On the other hand domestic waste management in a soil and aquatic resources are also accountable for the reduced field productivity. At this time the world is now facing an extreme situation of waste management from both the side i-e from industrialization and municipal waste management especially in a under developing countries. There is a need to address both problems in such a way that there should be resolution which can give proper management of both kind of waste. For this purpose public awareness about the waste management can play a crucial rule in controlling the waste of both the sides.

One of which waste-to-energy technologies have been developed to produce clean energy through the combustion of municipal solid waste in specially designed power plants equipped with the most modern pollution control equipment to clean emissions. Other waste management includes recycling of waste into fertilizers for use agriculture which is a common practice of waste management. The recycling of hazardous industrial wastes into fertilizers introduces several dozen toxic metals and chemicals into the nation's farm, lawn and garden soils, including such well-known toxic substances as lead and mercury. Many crops and plants extract these toxic metals from the soil, increasing the chance of impacts on human health as crops and plants enter the food supply chain. The report based on the use of recycle fertilizers from waste in agriculture industry represent the highly toxic substances found by testing fertilizers, as well as the strict regulations needed to protect humans and the environment from these toxic hazards.

Between 1990 and 1995, 600 companies from 44 different states sent 270 million pounds of toxic waste to farms and fertilizer companies across the country [(1) Shaffer 1998]. The steel industry provided 30% of this waste. Used for its high levels of zinc, which is an essential nutrient for plant growth, steel industry wastes can include lead, arsenic, cadmium, chromium, nickel and dioxin, among other toxic substances. Although the industrial facilities that generate these toxic wastes report the amount of chemicals they transfer off-site to the U.S. Environmental Protection Agency's (U.S. EPA) Toxics Release Inventory every year, they only report the total amount of a given chemical contained in wastes transferred over the course of a year, making it difficult to determine the chemical make-up of a given waste shipment. With little monitoring of the toxics contained in fertilizers and fertilizer labels that do not list toxic substances, our food supply and our health are at risk.

2. Tested fertilizers contain harmful toxic metals

California Public Interest Research Group (CALPIRG) Charitable Trust and Washington's Safe Food and Fertilizer tested 29 fertilizers from 12 states (2) for 22 toxic metals. This report documents the results of these fertilizer samples, demonstrates that the problem of toxic fertilizers is widespread, and details concerns with proposed regulations for the practice.

3. Toxic fertilizers threaten human health

The toxic substances found in the tested fertilizers have been linked to adverse human health impacts. The metals found in these fertilizers are known or suspected carcinogens, reproductive and developmental, liver, and blood toxicants. For example, beryllium is a suspected carcinogen, chromium and arsenic are known to cause cancer and barium can cause kidney and lung damage. Children are most susceptible to the toxic effects of most metals, especially lead, which has been the subject of intense government efforts to reduce lead exposure to children. Products like fertilizer are of great concern as children spend more time on or near the ground and are often exposed to ground level substances through hand-to-mouth behavior(3-5).

4. Toxic fertilizers pressure agricultural soils, food safety and waterways

As demonstrated in this report, the tested fertilizers contain toxic substances at high levels. These substances can accumulate in agricultural soils, become available for plant uptake, and run off into waterways (5).

5. Agricultural soil quality

Farming, especially single-crop farming, requires consistent and dependable soil conditions. The introduction to farm soils of toxic substances like lead and cadmium can adversely affect growing conditions and result in increased toxic accumulation as these metals are highly persistent in soils. This can negatively affect critical growing requirements, such as soil acidity or the solubility of beneficial metals like zinc in the soils.

6. Plant uptake

Some crops are more likely than others to absorb non-nutrient toxic substances from soils. For example, fruits and grains can absorb lead, and lettuce, corn and wheat can absorb cadmium from soils. This means that our food supply is at risk of contamination by toxic substances that could threaten human health.

7. Water quality

The overall health of the nation's waterways has declined dramatically over the last quarter-century. Forty percent of our rivers, lakes, and estuaries are still too polluted for safe fishing or swimming (7). Agricultural runoff is a common cause of waterway pollution. A 1998 U.S. EPA report found that metals are the second most common pollutants found in lakes, ponds, reservoirs, and estuaries. In fact, agriculture is the industry most responsible for lake pollution (8). The introduction of toxic substances from fertilizers to agricultural environments will only add to their concentrations in waterways that state and federal agencies are working to make safe for fishing and swimming.

Hazardous waste is usually viewed directly as a health hazard to humans. However, pollution in the form of hazardous waste can have a much more pronounced effect on the plants and animals of our environment. According to the Environmental Protection Agency, "When chemicals are disposed improperly, they can have harmful effects on humans, plants, and animals." By learning how hazardous waste affects the environment in a negative fashion, it's possible to recognize the threat and act (9-10)

8. Evolutionary

Hazardous waste also has the power to shape evolutionary changes. When species survival is threatened by pollution, the species must either adapt or become extinct. This is seen frequently in the case of antibiotic-resistant bacteria. This is also seen in the case of the peppered moth. According to Time magazine, "Since the passage of smoke-control laws in the 1950s, England's landscape has begun to emerge from its layers of soot. The cleaner trees or phytoremediator thus provide a lighter and safer resting place for any surviving speckled moths." The soot created by transport or industrial pollution in England or in Pakistan made the bark of tree's black, making it extremely easy for a speckled peppered moth to be seen, and thus eaten. However, since pollution control laws were enacted in the 1950s in England, the bark of trees has become lighter, thus creating evolutionary pressure that once again favors the speckled peppered moth, versus the black peppered moth, which the presence of pollution favored (11-13).

9. pH Change and oxygen depletion

According to National Geographic, "When humans burn fossil fuels, sulfur dioxide and nitrogen oxides are released into the atmosphere. These chemical gases react with water, oxygen, and other substances to form mild solutions of sulfuric and nitric acid." The change of pH caused by acid rain can have drastic effects on the plants and animals of the environment. It is also a one of the basic reason of Al toxicity in plants in acidic soil which was already reported by many researchers.(Azmat et al 2007). Entire forests, rivers and lakes can die due to the acidity of acid rain. Oxygen depletion in water originates from the overuse of fertilizers as well as hazardous waste. When a substrate such as oil or fertilizer is consumed by bacteria in water, it reduces the available oxygen for plants and fish.

Oxygen depletion can also be related with accumulation of heavy metals in the aerial parts of plant due its mobility in soil and plant uptake of heavy toxic metals due to which plants adopt certain survival strategy to overcome the stress like lignin deposition in vascular tissues or in veins or thickening of stomata to control the water evaporation but side effects was dominants i-e exchange of gasses viz CO_2 and O_2 as oxygen will not released due to smaller size of stomata it will produced oxidative stress in plants which is a common property of heavy metal accumulation (Azmat et al 2009). This ROS species will lead to increase the concentration of secondary metabolites and ultimately goes into food chain. When even small amounts of heavy metals or toxic chemicals are present in soil or water, their concentration is greatly amplified during travel up the food chain. This causes potentially lethal or trouble some concentrations of pollutants to be present in plants and animals. According to Young People's Trust for the Environment, "organo-chlorines caused many birds to lay thin shelled eggs, which cracked easily " (11-13).

In under developing countries using sewage sludge in agriculture fields is the risk of soil contamination with heavy metals and their possible transference to humans via food chain. Heavy metals, however, are regarded as inhibitors of enzymatic and microbiological activity of soil. This is because if added to soil (whether on purpose or by accident) they cause quantitative and qualitative changes in the composition of microflora and in enzymatic activity.

The new technologies in controlling waste based on adsorptions but if surface already heavily contaminated by adsorbates is not likely to have much capacity for additional binding. Up till now freshly prepared activated carbon which has a clean surface is use as adsorbing material and charcoal made from roasting wood differs from activated carbon in that its surface is contaminated by other products, but further heating will drive off these compounds to produce a surface with high adsorptive capacity. Although the carbon atoms and linked carbons are most important for adsorption, the mineral structure contributes to shape and to mechanical strength. Spent activated carbon is regenerated by roasting, but the thermal expansion and contraction eventually disintegrate the structure so some carbon is lost or oxidized. New adsorbents like seaweeds powder and solid tea wastage for controlling the heavy metal accumulation in plant are cost effective techniques

10. Use of natural resources and domestic wastage for effective remediation of toxic metals

Natural resources hidden in the sea are becoming more and more important in human life. It is an establishing fact that seaweeds are full of innumerable wealth of bioactive properties [Azmat et al., 2006a]. Seaweeds has been used by plant growers for centuries but the reason for beneficial results has only recently been attributed to the naturally occurring growth regulators and micro nutrients in the seaweed. The green alaga, Bryopsis Corymbasa/ codium iyengrii was found to display an antifungal activity [Azmat et al., 2006b]. Many types of seaweed have been used as food since they are not poisonous and usually have soft tissues. Seaweeds selectively, absorb from the seawater, elements like Na, K, Ca, Mg, Cl, I and Br, which are accumulated in their thalli. Seaweeds are known as alkaline food since their inorganic components play a very important role in preventing blood acidosis [Azmat et al., 2007]. The seaweeds can be used directly on the plant in the form of a spray and minerals in seaweed spray are absorbed through the skin of the leaf into the sap with not only mineral but auxins too. Seaweed sprays stimulate metabolic processes in the leaf and so helped the plant to exploit leaf locked nutrients and thus increase the rate of photosynthesis. Seaweeds could also be helpful in controlling the acidity of soil due to its alkaline nature

Although a lot of work on seaweed has been done on their taxonomy, distribution, morph-ecological studies, phytochemistry and antibacteial activity, but diminutive data is available in literature related to control the toxicity of heavy metal contaminated water. Therefore two research works were undertaken to control the toxicity of heavy metals by processes of adsorption or biosorption from proteins and nutrients of seaweed and tea wastage, for the management of toxic waste by domestic waste.

Biosorption (adsorption through living marine algae or microorganism) used for removing heavy metals and other pollutants is a newly developed environmental protection technique (Azmat *et al*.,2006a & b). Adsorption (surface absorption) on seaweeds or tea solid surface was found to be a promising alternative method to treat industrial effluents, mainly because of its low cost and high metal binding capacity (Kailas *et al.,* 2007; Utomo & Hunter, 2006; Yan-xin, 2001; Cay *et al.,* 2004) within a contaminated soil. These are low cost and easily available adsorbent and having strong adsorptivity towards metals like Cd, Zn, Ni, and Pb (Amarasinghe & Williams, 2007; Singh *et al.,* 1993; Ahluwalia & Goyal, 2005) because of the soft colloid and chemical components like palmitic acid of fatty group, terpenes and di-Bu phthalate present in it (Amarasinghe & Williams, 2007; Shyamala *et al.,* 2005; Mahvi *et al.*, 2005). In under developing countries using sewage sludge in agriculture fields is the risk of soil contamination with heavy metals and their possible transference to humans via food chain (Quaff & Ashhar, 2005). Heavy metals, however, are regarded as inhibitors of enzymatic and microbiological activity of soil. This is because if added to soil (whether on purpose or by accident) they cause quantitative and qualitative changes in the composition of microflora and in enzymatic activity (Wyszkowska, 2002).

Extensive studies on marine resources decontamination revealed that seaweeds can efficiently accumulate heavy metal, due to presence of polysaccharides, proteins which provides wide range of ligands for interaction with heavy metals ions and other macro and micronutrients(Same *et al*,2002; Stirk, & Staden., 2002 ; Schiewer & Wong 2000). Seaweeds extracts are used at the recommended times and rates for increasing growth of plants (Azmat *et a*l.,2007). These extracts supply the amounts of iron, zinc, copper, molybdenum, cobalt, boron, manganese and magnesium that most crops require. They form complex with metals ions by changing the oxidation state of metal consequently detoxification is occurred (el- Sheekh & el – Saied. ,2000 ; Azmat *et al*.,2006b). Seaweed can be used directly on the plant in the form of a spray, which may directly absorb through skin of leaves or absorbed by the root. Seaweeds is a potential candidate algae for biosorption of a number of heavy metals, but little is known about the phytotoxicity of mercury (Hg) in different plant species, like distribution and phytotoxicity in the whole plant and at cellular level and its control by seaweeds. Bio availability of Hg in soil, uptake of Hg at phytotoxic level, growth retardation, affects on palisade and spongy parenchyma cells in leaves (Ahmed, 2003, Ladygein, 2004), collated deposition in the vascular bundles and change in vacuoles with electron dense material along the walls of xylem and phloem vessel (Ladygein & Semenova**,** 2003; Boulia *et al*.,2006)**.** Shaw and Rout (1998**)** observed significant inhibition of root elongation, which was more prominent with Hg than Cd leading to increase in the cell size grown in aquatic medium with Hg. Mor *et al.,* (2002) reported growth inhibitory effect of Mercuric chloride in cucumber leading to the disorientation of root and shoot, while hypocotyle elongation, growth and cell wall loosing in young *Phaseolus vulgaris* was observed due to inhibition of cell wall division in apical meristem region. Hg, which is common in irrigation water and soil sediments, causes irreversible damages in tissue structure of plants leading to reduction in the productivity of crops. Introduction of seaweeds in Hg contaminated soil results in the improvement of growth parameters which showed that marine plant spray or powdered form were effective in immobilization of toxic metal in the food chain via cop plants.

As already know that heavy toxic metals concentration in the environment is basically related with the industrial operations which are leather processing, refractory steel, and chemical manufacturing industries (Andaleeb *et al.*, 2008). Due to broad industrial use, metals are painstaking serious environmental pollutants, increasing day by day. Excess of Cr in the environment causes hazardous effects on all living beings including plants (Arun *et al.*, 2005). A gradual decrease were reported for various morphological parameters like root fresh and dry weights, shoot fresh and dry weights, and plant height with increase in Cr levels (Andaleeb *et al*., 2008). Contamination of soil and water by chromium (Cr) is of recent concern. Cr also causes deleterious effects on plant physiological processes such as photosynthesis, water relations and mineral nutrition (Azmat & Khanum, 2005).This showed that soil contamination with heavy metal is now a day, a worldwide problem leading to agricultural losses and hazardous health problems as metals enter to food chain (Azmat & Khanum 2005 ; Azmat *et al.,* 2007). Also metals toxicity above certain threshold level important for the animal and human being but toxic at certain level for both. Samantaray *et al.,*(1998) reported that high concentrations of chromium exhibited severe chlorosis, necrosis and a host of other growth abnormalities and anatomical disorders including the regulation of the mineral metabolism, enzyme activity and other metabolic processes. Cr^{3+} taken up by plants because of its mobile nature in soil. Since trivalent and hexavalent Cr may interconvert in the soil and soil immobilize both trivalent and hexavalent chromium. It is difficult to asses separately the effects of the two types of Cr on plants. Consequently, it might be appropriate to use the term Cr toxicity in plants, (Arun *et al.,*2005) instead of toxicity of trivalent or hexavalent Cr. The effects of chromium on plant growth, crop yield, uptake and distribution in vegetative and reproductive parts are not yet fully understood. Although a number of studies were made to investigate the chemistry of chromium in soil and its uptake by plants and found that it a stimulant the plant growth (Arun *et al.,*2005) . Barcelo *et al*. (1985) described the inhibition of P and K translocation within the plant parts when bean plants were exposed to Cr in nutrient solutions. Some microbacteria were also used as a growth promoting agents in presence of heavy metals like Wani *et al.* (2008) reported the role of mesorhizobium strain RC3 in soil amended with Cr, he found that inoculation of RC3 strain results in plant growth-promoting substances which reduces the Cr by 14, 34, and 29 % in root, shoot and grains respectively with the increased in N contents by 40 to 46 % in root and shoot. Conventional methods to alleviate the toxicity of chromium include its chemical reduction followed by precipitation, ion exchange and adsorption on activated coal, alum or ash. Most of these methods require high energy or large quantities of chemicals.

The information available on the role of solid tea waste in soil contaminated by Cr on plants growth and their important nutritive value is of significant. Because it is only value reports which give the knowledge about the removal / control the mobility or remediation of toxic metal within the soil through complex formation (Azmat *et al*., 2010). Moreover, the report by Azmat *et al* (2010) discuss the effect of solid tea wastage on chromium activity on the growth promoting potentials of plants like potassium, phosphorus, protease and proline activity as a bio- indicator of environmental stress and their function in soil remediation (Azmat & Hira 2010). This research demonstrates the effect of solid tea wastage on chromium contaminated soil as a new technology to remediate the metal through adsorption on the surface of solid tea wastage, which immobilized the metal in the soil. The results of remediation have been checked on some important parameters of *Vigna radiata* which is very effective in the world of new technologies based on management of solid toxic wastage. This is the report which discusses the management of industrial waste by municipal waste management for beneficial purpose like immobilization of toxic metal within the soil and safe plants from environmental hazards.

11. Morphological and physiological parameters in presence of additive surface

Initially increase in the root length at lower concentration may be due to increased relative proportion of pith and cortical tissue layers that later on reduced at further increase in concentration of metals. The reduction in root growth could be due to the direct contact of seedlings roots with metal in the soil causing a collapse and subsequent inability of the roots to absorb water from the medium (Barcelo *et al.*, 1986) or may related with the inhibition of root cell division/ root elongation or to the extension of cell cycle in the roots. Whereas the seedlings with tea waste showed approximate normal root length especially in the plants grown with thoroughly mixed tea wastage showed normal growth rate (Fig. 1) reflects the remediation of Cr or immobility or complex formation of metal with applied tea surface.

Fig. 1. Effect of tea wastage on bean plants in Cr contaminated soil

Observed adverse effects of Cr on plant height and shoot growth (Fig. 2) is successfully controlled by solid tea surface (Fig. 2). The reduction in plant height might be mainly due to the reduced root growth and consequent lesser nutrients and water transport to the above parts of the plant.

Fig. 2. Effect of tea wastage on bean plants in Cr contaminated soil

The research investigation revealed that metals like Cr, Cd and Hg causes significant decrease in fresh and dry weight, of root and shoot, protein, carbohydrate, chlorophyll and carotenoids in plants under study (*Vigna radiate)* while application of seaweeds in contaminated nutrient solution causes significant healthy growth (Fig.3 & 4)

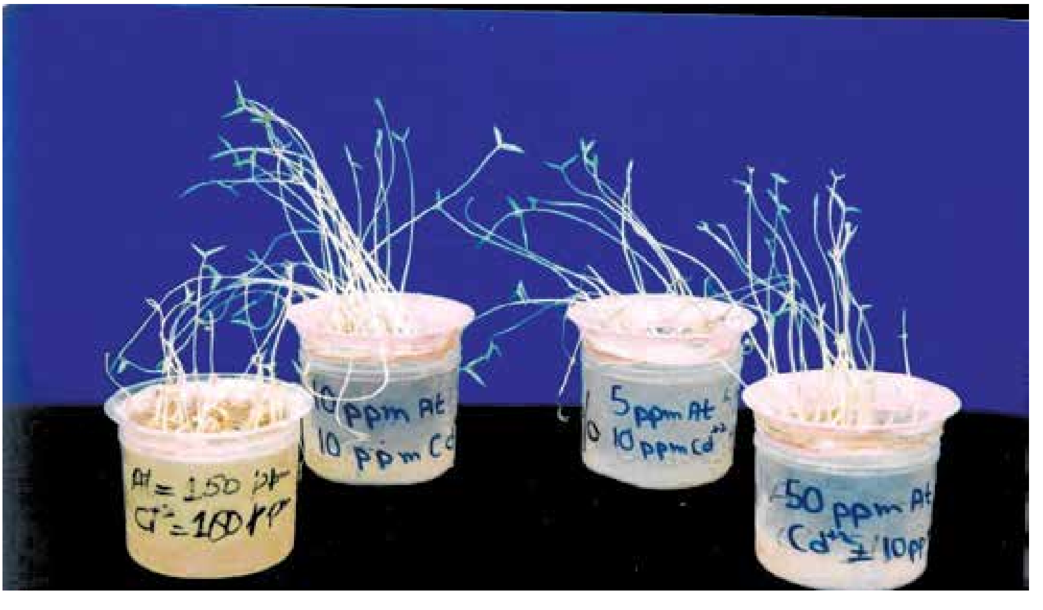

Fig. 3. Effect of Cd metal on Bean Plant

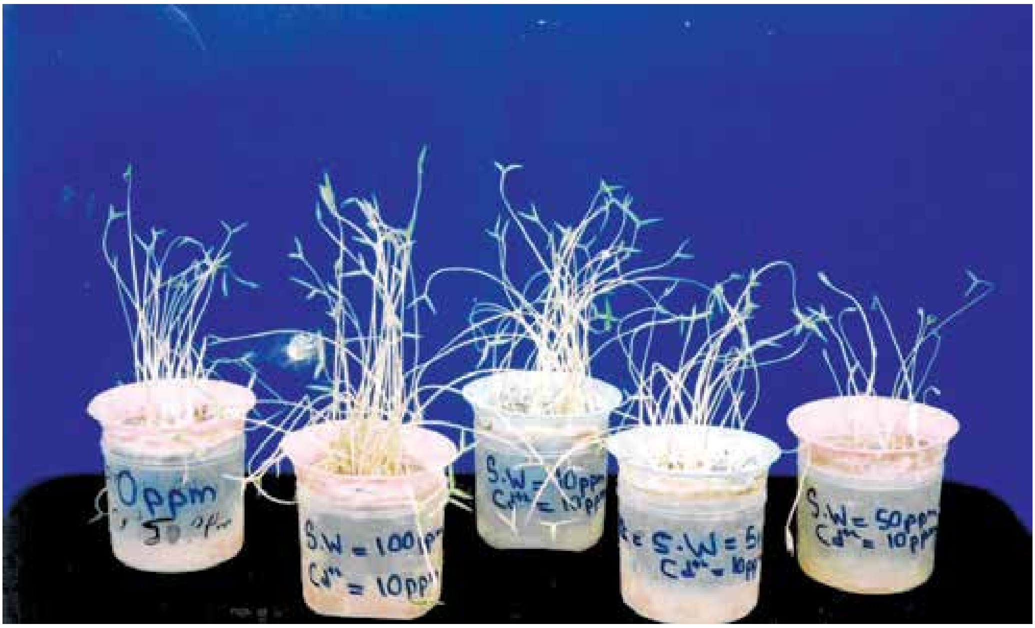

Fig. 4. Effect of seaweeds in presence of Cd on bean plants

The accumulation of chromium by germinating seedlings appears to be significantly affected by Cr concentration and occurred in a linear manner (Azmat *et al.*, 2005). Cr-treated plants showed growth depression and decrease in fresh and dry weight. It was found that dry matter production was severely affected by Cr(III) concentrations greater than 100 ppm. The reduction in plant height might be mainly due to the reduced root growth and consequent lesser nutrients and water transport to the above parts of the plant. In addition to this, Cr transport to the aerial part of the plant can have a direct impact on cellular metabolism of shoots contributing to the reduction in plant height. The effect of Cr on water relations was highly concentration dependent, and primary and first trifoliate leaves were affected differently (Azmat *et al.*, 2007). The seedlings which were grown in spreaded tea wastage and soil mixed with the tea wastage showed remarkable effect on the growth as well as physiological parameters. There were significant improvements in growth of the plants, indicated the adsorption of toxic metal on surface applied which were effectively controlled the mobility of metal within soil on applied domestic waste. Figure 1 &2 showed healthy growth of plants in tea wastage although it was severely affected due to the decline in protein contents ($p<0.05$) at elevated concentration of Cr^{3+} with elevated concentration of carbohydrate ($p<0.001$) and amino acid ($p<0.05$) that may be attributed with use of protein under stress and non utilization of carbohydrate whereas plants grown with tea wastage reflects the effect of tea wastage as a manure and biosorbent surface which showed inhibitory effects on the mobility of toxic metal Cr^{3+} and proves that tea wastage could be utilized to increase the soil fertility even in draft condition of toxic metals. Healthy growth in tea amended soil may also be related with the high K and P contents. Results of use of tea waste and marine green algae on the plants showed the decrease in the concentration of K contents both in root and shoot at increasing concentration of Cr and Cd compared with the control plants ($p<.005$) which showed a dramatic affect on the plants ability to survive and functioning during metal stress periods. Initial potassium deficiency shows up as yellowing of older leaf blades, lower leaf blades, which is then followed by dieback of the leaf tip and scorching of the leaf margins as the deficiency problem becomes worse (Biddappa & Bopaiah, 1989).

Once these conditions occur, wear injury for the turf plants will increase significantly. Factors which can lead to potassium deficiency include: leaching in sandy soils or soils irrigated with the contaminated water. Many plant physiologists consider potassium second only to nitrogen in importance for plant growth. Potassium is second to nitrogen in plant tissue levels with ranges of 1 to 3% by weight. Potassium is the only essential plant nutrient that is not a constituent of any plant part. Potassium is a key nutrient in the plants tolerance to stresses (Arun *et al.,*2005). Increase in potassium contents in tea waste amended plants showed more tolerance surviving capability under various stresses, such as metals, cold/hot temperatures, drought, and wear and pest problems (Dahiya *et al.,* 2005). And both role of potassium as biophysical and biochemical were visible in this investigation. Potassium acts as catalysts for many of the enzymatic processes in the plant (Azmat *et al*., 2010) that are necessary for plant growth to take place. Another key role of potassium is the regulation of water use in the plant (osmoregulation). This osmoregulation process affects water transport in the xylem, maintains high daily cell turgor pressure which affects carry tolerance, cell elongation for growth and most importantly it regulates the opening and closing of the stomata which affect transpirational cooling and carbon dioxide uptake for photosynthesis (Azmat *et al*.,2010). Results of tea waste amended soil showed marked effect on the plant physical and biological processes where K contents reaches to the normal value. and showed more rapid growth in soil contaminated with the Cr. Metabolic alterations by Cr^{3+} exposure and their control by solid tea wastage was already described by Azmat *et al* (2010) showed direct effect on enzymes or other metabolites or by its ability to generate reactive oxygen species which may cause oxidative stress. Increase in concentrations of the phosphorus (Azmat *et al.,* 2010) in different parts of the seedling was observed at low concentration of Cr (50ppm) which gradually lowered with an increase in concentration of Cr as compared to the control plants ($p<0.005$). It is known that P and Cr are competitive for surface sites. Hence, it is possible that Cr effectively competed with this element to gain rapid entry into the plant system. The reduction in K and P at elevated concentration could be due to the reduced root growth and impaired penetration of the roots into the soil due to Cr toxicity (Biacs *et al*., 1995). The magnitude of the content of P in tissues increased at lower concentration and its uptake decreased with increasing levels of Cr in the soil (Tsvetkova & Georgiev, 2003). It is concluded that added P alleviates the deleterious effect of Cr in the soil and improves the growth and the dry matter of shoot in been plants. Poor translocation of Cr to the shoots could be due to sequestration of most of the Cr in the vacuoles of the root cells to render it non-toxic which may be a natural toxicity response of the plant. It must be noted that Cr is a toxic and nonessential element to plants, and hence, the plants may not possess any specific mechanism of transport of Cr. The reduction in nitrogen compounds, K and P could be due to the reduced root growth and impaired penetration of the roots into the soil due to Cr toxicity (Azmat & Khanum 2005) which is successfully controlled by solid tea surface.

Results of tea waste management in the soil contaminated by industrial toxic metal clearly reflects that there should be proper research in this area is require where instead of making fertilizer from industrial waste, new technology of remediation of management of domestic waste to detoxify the industrial waste should be work out

12. Proline as an internal toxic affect manger

Proline , act as a bioindicator , an amino acid accumulated in stress to control the oxidative stress or stress manger internally in the plant. Proline increases the stress tolerance of plants through such mechanisms as osmoregulation, protection of enzymes against denaturation, and stabilization of protein synthesis (Kaushalya *et al*. 2005).Results reported in literature showed that proline contents increases with the increase in concentration of metal because for survival of plants under stress. Accumulation of proline starts under mild water stress and the magnitude of accumulation is proportional to the severity of stress (Ganesh *et al.,* 2009 ; Kaushalya *et al.,* 2005). The investigation, about the toxic metal management through tea waste or marine algae showed , higher proline content was observed at 100 ppm (0.669) of Cr and the decrease may be related with the reduced growth of plant (Fig. 1 & 2). Thus proline accumulation under such condition may also be operative as usual in osmotic adjustment while accumulation of proline in tissues can be taken as a dependent marker for stress. The higher proline content was reduced (0.039) by application of solid tea surface and plant observed normal growth under metal stress because proline has multiple functions such as osmoticum, scavenger of free radicals, protective role of cytoplasmic enzymes, source of nitrogen and carbon for post-stress growth, stabilizer of membranes, machinery for protein synthesis and a sink for energy to regulate redox potential. Proline acts as a cytoplasmic osmoticum as it accumulates to a higher degree under stress conditions, which may play an adaptive role for any stress tolerance (Vartika *et al.* 2004).

13. Enzymes activity as an environmental biomarker

Enzymes serve as the labor force to perform every single function required. Enzymes are the key to life. No enzymes, no life. Altered Enzymes activity under metal stress was successfully managed at lower concentration of metal with domestic tea wastage because an increase and decrease in the enzymes activity could reflect the current situation of environments as biomarkers (Scoccianti *et al*., 2008).

Decrease in protease activity of *Vigna radiata* under Cr stress may be related with deficiency in phosphorus contents which may be attributed with reduce nitrogenous compounds (protein) and potassium contents of seedling due to which biophysical visual symptoms on leaves appeared like leaf growth traits that might serve as suitable bio-indicators of heavy metal pollution. Primary and trifoliate leaves of bean plants due to which Cr showed a marked decrease in leaf area; trifoliate leaves were more affected by Cr than the primary leaves (Barcelo *et al*., 1985; Dube *et al.,* 2003). Protease activity in tea waste amended plants increases which may increase the hydrolyzing capability of seedling to hydrolyze oxidative proteins for survival of plants under stress (Palma *et al*. 2002).

Observation showed that AST enzymes activity initially increases at 50ppm of Cr (589.56) as compared with the control plants ($p<0.05$) and gradually decreases with the increase in concentration of Cr. This showed that elevated concentration of Cr (III) alters the enzyme activity. Mixed tea plants showed significant effect of tea wastage as an adsorbent surface. Results of tea wastage treated plant showed that at 50 ppm, the activity of AST enzyme activity (Fig. 5) reaches approximately to the normal value (556.97) when compared with control (556.97).This reflects that Cr remediation may occur by adsorbing on the solid tea surface. But increase in the concentration of Cr showed negative effects on the AST enzymes activity. Spreaded tea waste plants showed no remarkable effects on the AST enzyme activity where continuous decrease in enzyme activity were observed which showed that for proper adsorption of heavy metal it is essential to mixed the solid tea wastage with soil thoroughly (Azmat *et al*., 2010).

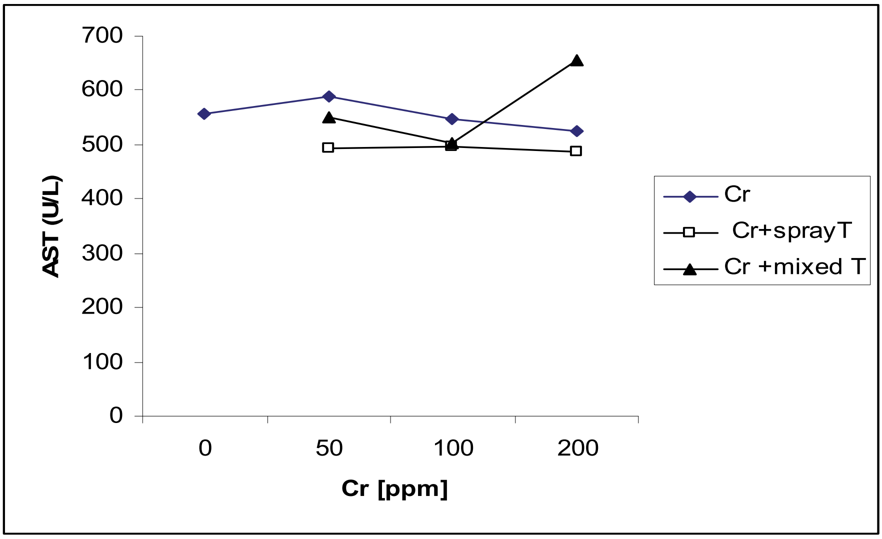

Fig. 5. Effect of Cr and solid tea wastage as a biosorbent manure on enzymes AST activity of seedlings of *Vigna radiata*

The ALT showed (Fig. 6) great variation in its activity. Initially it decreases ($p<0.001$) and then slightly increases with the elevated concentration of the Cr^{3+} i.e. 100 ppm and 200 ppm.

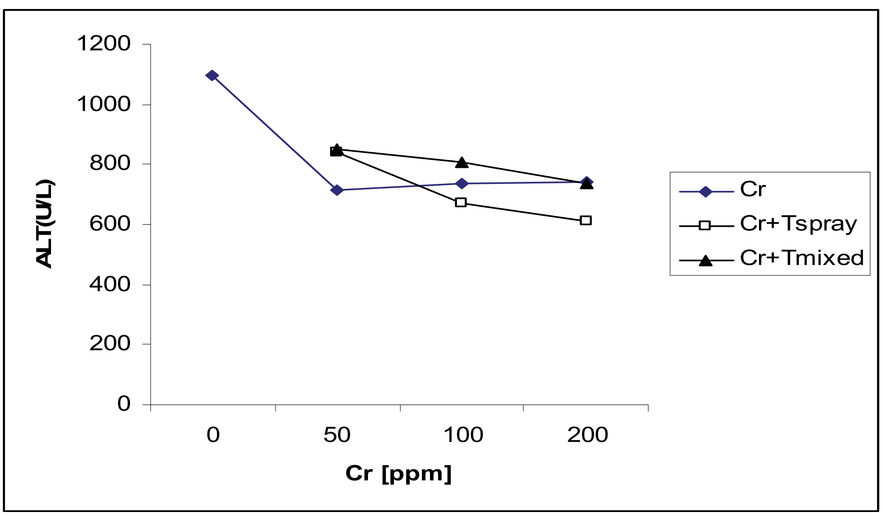

Fig. 6. Effect of Cr and solid tea wastage as a biosorbent manure on enzymes ALT activity of seedlings of *Vigna radiata*

Plants with mixed tea waste again showed slight improvement in the enzymes activity but not to that of control one. This showed that the toxicity of metal to the biomarker at higher concentration of metal.

ALP ($p<0.05$) showed completely different response to the heavy metal Cr which showed an increase in enzyme activity to overcome the heavy metal stress. This may be attributed with decrease or increase in the enzyme activity due to the denatured of site of interaction of enzymes under metal stress (Fig. 7).

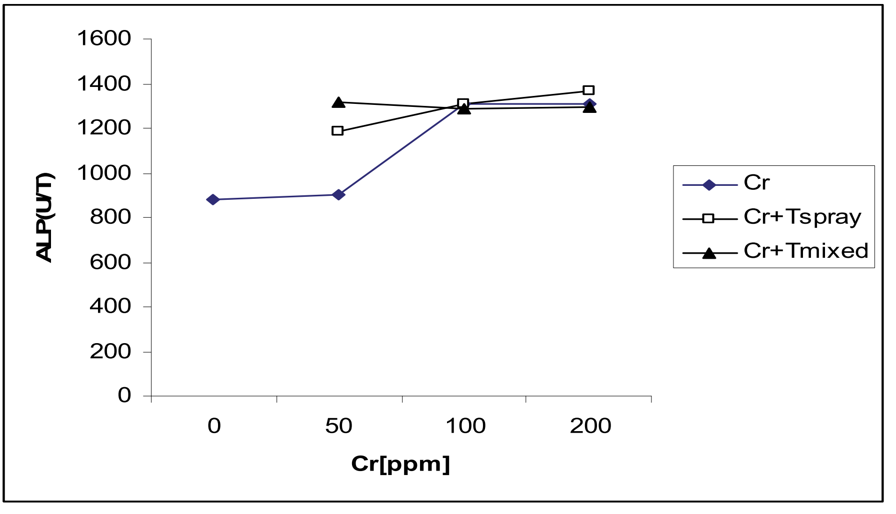

Fig. 7. Effect of Cr and solid tea wastage as a biosorbent manure on enzymes ALP activity of seedlings of *Vigna radiata*

The results of the analyses, reported hereby prove that activity of individual enzymes is quite a sufficient indicator of soil fertility. Enzymatic activity of soil is a reliable measure of current biological status. Naturally occurring amounts of heavy metals do not disturb the biochemical balance of soil. Slightly higher quantities of heavy metals might stimulate the activity of soil enzymes, whereas in larger they will have an inhibitory effect (Wyszkowska, 2002). Heavy metals, however, are regarded as inhibitors of enzymatic activity. It was observed that the potential soil fertility index decreased at higher concentration of chromium contamination and showed appreciably negative affect on the germination processes and altered the activity of enzymes correlated with chromium. It was interesting to note that the activity of two enzymes decreased up to 100 ppm level of Cr while ALP as found to be increased at all applied doses and experiments also revealed that chlorophyll a and b decrease with the concentration of Cr (Azmat *et al.*, 2005) but improved with the application of tea waste amended soil. These results showed that chlorophyll as a vital pigment for life of plants which is not tolerant to chromium and its toxicity was correlated with the degree of soil contamination. Soil amended with the solid tea wastage showed increase in concentration of chlorophyll contents. Carbohydrate contents of *Vigna radiate* showed increase in concentration which may be related with the metal toxicity and alteration in key enzymes activity of protein and carbohydrate metabolism (Azmat et al 2010). The effect of Cr(III) on the activity of AST, ALT and alkaline phosphatase depended on the rate of soil contamination with chromium. Wyszkowska (2002) reported that in the objects polluted with 100 mg Cr kg^{-1} of soil, plants became necrotic at the stage of seedlings, and in the soil treated with 150 mg Cr kg^{-1} of soil, the emergence of plants was inhibited.

14. A probable mechanism of remediation of metal within the soil

The probable mechanism of adsorption of metals like Cr ,Hg, and Cd based on complex formation with fatty acids, algainate,polysaccharides found in the algae and solid tea surface, The incorporation of these two cost effective adsorbate play a crucial role in checking the mobility of metals. It is proposed that metal in a complex state doesn't moves in the free state to accumulate in the plants through false signal to the plant growth system. The mechanism of remediation of Cr^{3+} based on adsorption of Cr^{3+} on tea solid wastage within the soil where it was found that in the pot which contained thoroughly mixed tea waste with the garden soil shows soil stony structure and the plants of this pot was quite erect and more healthy as compared to plants with $Cr^{3+}$and with no Cr^{3+}. The available biochemical experimental data offered here that plants with mixed tea showed more tolerant morphological as well as physiological parameters. The remediation mechanism for the adsorption of heavy metal Cr^{3+} using tea waste has been presented here showed that soft colloid and chemical components like palmitic acid of fatty acids group, terpenes and di-Bu phthalate play a key role for complex forming with the metals reduced the mobility of metal in the contaminated soil and reduced the accumulation of Cr^{3+} in plant tissues in the early stage of development of seedlings whereas the plants grown in a contaminated soil with seaweeds show swollen state of soil when watered and soil wet long time which indicate that seaweeds retained water in it and increases the water holding capacity which ultimately benefit to the soil under stress and supply water into the plants, results to overcome the stress which results in the better growth and clean food from every unnecessary material (Fig.8-10)

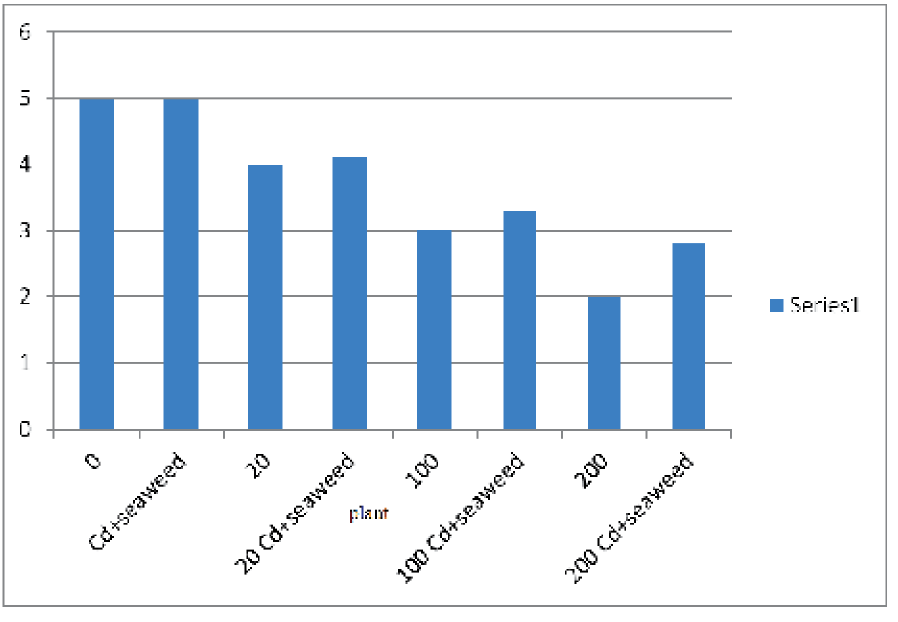

Fig. 8. Effects of seaweeds in root length of *Vigna radiata* in Cd contamination

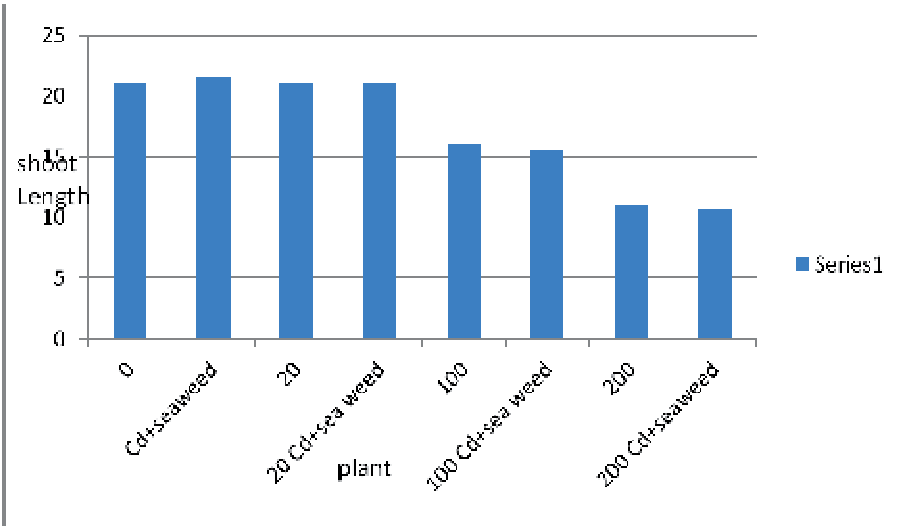

Fig. 9. Effects of seaweeds in shoot length of *Vigna radiata* in Cd contamination

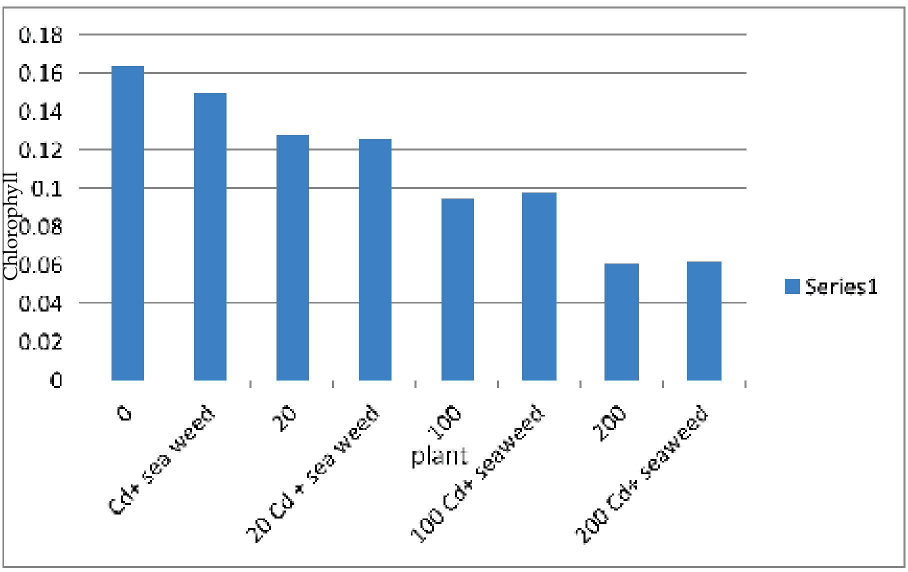

Fig. 10. Effects of seaweeds in chlorophyll content of *Vigna radiata* in Cd contamination

These topics require further researches in the field of biosorption and new technologies of remediation of one wastage with others toxic waste.

15. Mechanism of complexation

The biosorption of metals (**Ahalya et al 2005**) take place through both adsorption and formation of coordination bonds between metals and amino and carboxyl groups of cell wall polysacchonides of seaweeds. The metal removal from sewage sludge may also take place by complex formation on the cell surface after the interaction between the metal and the active groups of proteins and amino acids found in green algae. Complexation was found to be only mechanism responsible for calcium, magnesium, cadmium, zinc, copper and mercury accumulation by marine algae.

Investigation showed that application of dry seaweed powder to the sludge provides multiple levels of potential benefits. These potential benefits have been identified during seaweed spray including nutritional level, physiological process, morphology, mineral and metal ion (Schiewer and Wong; 2000) uptake by Plants . The physico-chemical interaction occurs between the toxic metal and the surface polysaccharides of the biomass (algae}, ion – exchange, complexation and adsorption takes place and the phenomena is not metabolism dependent (Fig.1-4). The surface of the seaweeds is constituted of polysaccharides and proteins that provide a wide range of ligands for heavy metal ions. These processes are rapid and reversible. Seaweed contains all known trace element and these elements can be made available to plant by chelating i-e by combining the mineral ion with organic molecules. Starches, sugars and carbohydrates in seaweed and seaweed products possess such chelating properties (Ahalya et al 2005). As a result, these constituents are in natural combination with the iron, cobalt, copper, manganese Zinc and other trace elements found naturally in seaweed. That is why these trace elements in seaweed product do not settle out in alkaline soils, but remain available to plant, at the time of need. Fig. (4) showed that when seaweeds mixed with the sludge, biosorption of toxic metals takes place, which stimulate the growth rate and physiological processes (Azmat *et al* 2007 & Azmat *et al* 2006).

16. Conclusion

Today's industrial world has contaminated our soil, sediments and aquatic resources with hazardous material. Metal water is often resulting of industrial activities, such as mining, refining, and electroplating, Hg, Pb, As, Cd and Cr are often prevalent at highly contaminated sites. Therefore it is our responsibility to check and develop the low cost techniques to remove the toxic metals by methylation, complexation or changes in valance state from the environments for humanity. Domestic waste is generated as consequences of household activities such as the cleaning, cooking, repairing empty containers, packaging, huge use of plastic carry bags. Many times these waste gets mixed with biomedical waste from hospitals and clinics. There is no system of segregation of organic, inorganic and recyclable wastes at the household level. Improper handling and management of domestic waste from households are causing adverse effect on the public at large scale and this deteriorates the environment. Segregation of this different type of waste is essential for safety of the environment because the improper management and lack of disposal technique of the domestic waste pollutes to the environment. It affects the aquatic resources. It also changes the physical, chemical and biological properties of the water bodies. Uncollected waste is scattered everywhere and reaches to the water bodies through run-off as well as it percolate to underground water. The toxics contain in the waste, contaminates water. It also makes soil infertile and decrease the agricultural productivity. Few researches on laboratory scale cannot give the proper use of such a big hazard. It should be duty of all citizen to disposed the waste in separate begs to keep the environment safe for their lives from spread domestic wastage because dispersed uncollected waste and improper disposal techniques drains also get clogged which lead to mosquitoes by which various diseases like malaria, chicken-guinea, viral fever, dengue etc. arise and affect the health of people adversely. The lack of literacy programmes on waste management and disposal techniques which keeps the most of the people ignorant about waste management. This lack of awareness among the people increases the problems. With the growing population the huge waste is being generated day by day. There is wide use of plastics, advanced technology and other materialistic things. This resulted in different characteristics of waste which became complicated problem for management of domestic waste and disposal techniques. This is such a burning problem concerned with environment that needs to be carefully studied and researched, as on every street waste is lying uncollected scattered around local bins and dumped around locality consequently there is occurrence of bad smell as well as hazard to the human health and to the passerby.

Research based on removal of toxic metals by marine algae and tea wastage require further investigations on domestic wastage to keep clean the environment with public environmental education.

17. Acknowledgment

This chapter is prepared by the help of information given in WASTE LANDS: THE THREAT OF TOXIC FERTILIZER Report by **Matthew Shaffer**, Toxics Policy Advocate CALPIRG Charitable Trust The State PIRGs and The Effects of Hazardous Waste on Plants & Animals | eHow.com http://www.ehow.com/list_7174924_effects-hazardous-waste-plants-animals.html#ixzz1McDLWThO based on following **references**

- Time Magazine: Evolution by Pollution
- Young People's Trust for the Environment:Endangered Wildlife
- National Geographic: Acid Rain
- Agency for Toxic Substances and Disease: ToxFAQs™ for Polycyclic Aromatic Hydrocarbons (PAHs) Registry:
- National Geographic: Toxic Waste

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Factory Farming: Toxic Waste and Fertilizer in the United States, 1990-1995," Environmental Working Group, 1998. 2) In addition to California, Georgia, Idaho, Indiana, Michigan, Minnesota, Montana, North Carolina, Pennsylvania, Texas, Virginia, and Washington states, the tested fertilizers (See Appendix B) are available in many other states. This is especially true for home and garden fertilizers like Scotts.3) 40 CFR 266.20, 40 CFR 268.40 (i) 4) Zinc fertilizers are subject to less stringent Phase III Land Disposal Restrictions, which do not include beryllium and vanadium. Zinc fertilizers made from electric arc furnace dust (K061) are not subject to standards. 40 CFR Part 268, [FRL-6153-2], RIN 2050-AE05, EPA, 1998. 5) "Visualizing Zero: Eliminating Persistent Pollution in Washington State." Washington Toxics Coalition, 2000.6) Wilson, D., "Fear in the Fields," The Seattle Times, July 3, 1997, citing Agency for Toxic Substances Disease Registry, EPA. 7) www.pirg.org /enviro/index.htm 8) National Water Quality Inventory: 1998 Report to Congress (EPA841-R-00-001) 9) 40 CFR 266.20 and 40 CFR 268.40 (i) 10) The exception is K061 (the waste code for electric arc furnace dust produced by steel mills) which are not sunject to regulation. 11) Non-zinc fertilizers are subject to Universal Treatment Standards, 40 CFR 268.48 12) http://www.epa.gov/epaoswer/hazwaste/recycle/fertiliz/index.htm13) Environmental Protection Agency, EPA530-F-99-043, December 1999.

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5

Key Areas in Waste Management: A South African Perspective

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1. Introduction

"In the era of industrialization, mining and heavy industry became a major factor in the national economy" (Schreck, 1998). Since industry has become an essential part of modern society, waste production is an inevitable outcome of the developmental activities. In the past industry was geared solely towards economic aspects and totally neglected ecological issues. These industries release huge quantities of wastes into the environment in the form of solid, liquid and gases. A substantial amount of these wastes is potentially hazardous to the environment and are extremely dangerous to the living organisms including human beings.

South Africa's re-integration into the global economy and the Southern African political arena necessitates an improved pollution and waste management system. The country's economic and industrial policy has also turned towards export promotion as a pillar of South Africa's development. Therefore, the country has a growing obligation to meet international commitments and to be a globally responsible country. The government therefore promotes an integrated approach to pollution and waste management as a key factor in achieving sustainable development.

The integrated pollution and waste management policy is driven by a vision of environmentally sustainable economic development. This vision promotes a clean, healthy environment, and a strong, stable economy. By preventing, minimizing, controlling and mitigating pollution and waste, the environment is protected from degradation by enhancing sustainable development.

Having outlined all these, there is still a concern with both the detrimental health effects and environmental impacts of sub-optimal management of waste and increasing levels of pollution in South Africa.

The constitution of South Africa (Act 108 of 1996) established the Bill of Rights that ensures that everyone has the right to an environment that is not harmful to their health and well being. Legislative and other measures should be used to ensure that the environment is conserved and protected for future generations.

According to (Karani & Jewasikiewitz, 2007), in the past, the waste management sector was dominated by private sector with selective operations in what makes business sense through recycling of saleable products. Materials mostly recycled included paper and hard board, plastics, glass, tinplate and aluminum. The rest of the waste materials estimated at 10.2 million tons of both general and hazardous end up in landfills.

South Africa's Emissions per capita in 1999 were estimated at 7.8 metric tons of carbon dioxide (CO_2) equivalent and volumes of waste generated in 1992 and 1997 both general and hazardous accumulated to about 500 million tons (Department of Water Affairs [DWA], 1998).
Given this state of development the country has diverse waste stream, the management of which varies in approach, efficiency and complexity depending on the responsibility of local authority. Waste generation rates for the different market segments are shown in Table 1. The table shows that mining was the largest contributor of waste to this increase followed by industrial, power, land use, domestic and trade and sewage. In 1997, the trend in the table shows that mining was still leading in waste generation while a decline was realized in industrial, domestic and trade and sewage. This trend could be as a result of international standards that impact directly on waste generation.

Waste stream	1992 (CSIR study)	1997
Mining	378	468.2
Industrial	23	16.3
Power generation	20	20.6
Agriculture and Forestry	20	20
Domestic and trade	15	8.2
Sewage sludge	12	0.3
Total	**468**	**533.6**

[a]The table provides information extracted from a study on waste generation rates in millions tons per year in South Africa. The study was conducted by the Council for Scientific and industrial research.

Table 1. Waste generation rates in South Africa in 1992 and 1997[a]

There are ample evidence that improper disposal of these wastes may cause contamination of air (via volatilization and fugitive dust emissions); surface water (from surface runoff or overland flow and groundwater seepage); ground water (through leaching/infiltration); soils (due to erosion, including fugitive dust generation/deposition and tracking); sediments (from surface runoff/overland flow seepage and leaching) and biota (due to biological uptake and bioaccumulation). According to (Misra & Pandey, 2005), contamination of ground water by landfill leachate posing a risk to downstream surface waters and wells is considered to constitute the major environmental concern associated with the landfilling of the waste. In order to safeguard our environment, it is important to regulate such hazardous waste in environmentally feasible and sound manner.
According to the (Department of Water Affairs [DWA], 1998), waste disposal in South Africa is mostly in landfills, but it is estimated that only 10% of landfills are managed in accordance with the minimum requirements.
Most of the cities in South Africa have well-managed landfills as well as recycling programs. Recycling activities are mostly private sector initiatives run by packaging manufacturers through buy-back facilities.

2. South African waste management perspective

Waste management in South Africa has in the past been uncoordinated and poorly funded. According to (Nahman & Godfrey, 2010) key issues include inadequate waste collection services for a large portion of the population, illegal dumping, unlicensed waste management activities (including unpermitted disposal facilities), a lack of airspace at permitted landfills, insufficient waste minimization and recycling initiatives, a lack of waste information, lack of regulation and enforcement of legislation, and, indeed, limited waste-related legislation in the first place.

In response, the National Waste Management Strategy (NWMS) (Department of Environmental Affairs and Tourism [DEAT], 1999) emphasizes the need for integrated waste management, which implies coordination of functions within the waste management hierarchy. In particular, the diversion of waste from landfill through waste minimization and recycling is a national policy objective under the White Paper on Integrated Pollution and Waste Management (Department of Environmental Affairs and Tourism [DEAT], 2000), the NWMS and the Waste Act, which recognize the importance of moving waste management up the waste hierarchy (i.e. greater emphasis on waste avoidance, minimization and recycling to reduce impacts further downstream) (Nahman & Godfrey, 2010).

In addition, to deal with the issue of insufficient funding, the NWMS invokes the Polluter Pays Principle (PPP). In the context of solid waste management, the PPP implies that all waste generators, including households and companies, are responsible for paying the costs associated with the waste they generate. These include not only the direct costs associated with the safe collection, treatment and disposal of waste; but also the external costs (externalities) of waste generation and disposal, such as health and environmental damages (Department of Environmental Affairs and Tourism [DEAT], 1999).

3. Waste generation

- Commercial and Domestic General Waste

Municipal waste generated in recent years is increasing and mainly due to the increasing urbanization.

General waste – is waste that does not pose an immediate threat to man or the environment, that is, household and garden waste, builders' rubble and some dry industrial and business waste. It may, however, with decomposition and rain infiltration, produce leachate, which is unacceptable.

The mixed nature of general waste, the high proportion of recyclable material going to landfill, and the presence of small quantities of hazardous wastes are key challenges that need to be addressed.

- Mining and Industrial Hazardous Waste

The main sources of mining and industrial wastes are gold, platinum, coal, etc. and power industries, ore extraction, pulp and paper, petrochemical industries, etc.

According to (Adler, 2007), following the discovery of immense gold resources in South Africa in 1886, the mining industry played a central role in the country's economic, political, and social environment. Because minerals in South Africa are highly diversified, plentiful, and profitable, government has allowed the industry to be privileged, enabling it to maximize profits. But South Africa recently incorporated objectives of sustainability and social justice into its constitution. Not based on notions of sustainability, the early gold-economy was simply an extractive industry with little consideration given to possibly adverse long-term effects.

Hazardous waste – is waste containing or contaminated by poison, corrosive agents, flammable or explosive substances, chemical or any other substance which may pose detrimental or chronic impacts on human health and the environment.

Mining waste – is waste from any minerals, tailings, waste rock or slimes produced by, or resulting from, activities at a mine.

`The composition of mining waste varies according to the nature of the mining operation and many other factors, but where the same mineral is extracted from a similar style of metalliferous or industrial mineral deposit or coal, the waste usually has similar characteristics. There are many potential sources of industrial minerals from mining waste. Waste from one mine may be a byproduct or co-product in a mining operation elsewhere' (Scott et al., 2005).

Mining activities, from exploration to extraction and processing, have recently come under increasing public scrutiny in South Africa as competition for environmental resources has intensified and the post-Apartheid government's attitude has shifted towards improved environmental quality and health (Department of Minerals and Energy [DME], 1997).

'First, the nature of environmental and health risks from mining makes them difficult to quantify and even more difficult to evaluate in monetary terms. For example, in coal and other mining operations, surrounding downwind areas, which are not owned by mining firms, are often subject to dust particles emanating from the mines. In addition, acid run-offs can pose hazards to mine workers, to fish and wildlife, and to consumers when they persist in water and food` (Wiebelt, 1999). Most of these risks are not immediately apparent to either producers or consumers and the nature of these risks varies widely among types of mineral being extracted, on whether mining is onshore or offshore, and on the methods and technologies of extraction used. The major form of environmental externalities in South African mineral extraction is solid waste generation (Table 2).

The solid waste generated comprises of mostly potentially hazardous tailings and slags (Department of Environmental Affairs [DEA], 1992a). These make up the bulk of the mining's solid waste stream, which in turn represents nearly 90 percent of the total South African waste stream. Only 0.007 percent of mining waste takes the form of air emissions, and only 0.4 percent is discharged with waste water.

Although the quantity of waste discharged in waste water is small in comparison with the solid waste stream, the waste water stream is an important vehicle for hazardous mining waste. Table 2 shows that a small number of total waste streams in gold, platinum group metals, and antimony mining, and most of the waste in zinc refining have to be rated as hazardous with acid cyanide-containing goldmine effluents representing the largest hazardous waste stream in mining. However, it has to be kept in mind that environmental externalities in mining not only depend on the rates at which extraction takes place but also on the cumulative amounts of mineral ores already extracted.

It is estimated that backlog in mining waste includes some 12 billion tons of overburden and depleted processed ores, and about 30 thousand tons of semi-purified concentrates containing high concentration zinc, copper, cadmium or cobalt (Department of Environmental Affairs [DEA], 1992a). Thus, high environmental damages are incurred as a result of past and current mining activity.

Highly hazardous waste: contains significant concentrations of highly toxic constituents persistent in the environment and bio-accumulative;

Moderately hazardous waste: is highly explosive, flammable, corrosive or reactive, or is non-hazardous waste which are easily accessible, mobile or infective, or contains significant concentrations of constituents that are potentially highly toxic but only moderately mobile, persistent or bio-accumulative, or that are moderately toxic but are highly mobile, or persistent in the environment, or bio-accumulative;

Sector	Air Emissions	Waste Water	Solid/Liquid Waste	Total	Hazardous Waste[d]	Potentially Hazardous waste	Non-Hazardous Waste
Agriculture[b]	-	-	-	-	-	-	-
Coal mining	-	-	45,600	45,600	-	34,200	11,400
Gold mining	-	1,538	190,188	191.726	1,013	531	190.181
Other mining, of which	27	18	139,268	139,313	46	41	139,226
-Platinum group metals	27	27	45,137	45,182	18	28	45,136
-Phosphate	-	-	10.920	10,920	-	-	10,920
-Base metal	-	-	59,600	59,600	-	-	59,600
-Zinc	-	-	41	41	28	14	0
-Antimony	-	-	420	420	-	-	420
-Diamonds	-	-	23.000	23.000	-	-	23,000
-Asbestos	-	-	150	150	-	-	150
Total mining	**27**	**1.556**	**375,056**	**376,639**	**1.059**	**34,773**	**340.807**
Metallurgical and metals industries[c]	13	16	4,872	4,902	335	4,567	-
Non-metallurgical manufacturing industries	323	602	14,448	15,373	452	4,772	10,149
Services[c]	1,609	7	20.275	21,891	47	1,654	20,190
Total economy	**1,972**	**2,182**	**414.651**	**418,805**	**1,893**	**45.766**	**371,147**

[a]Excluding carbon dioxide emissions and sediments from waste water. - [b]Agriculture is not included in the survey [c]includes power generation. - [d]includes highly, moderately and low hazardous waste.

Table 2. Mining and industrial waste in South Africa, 1990/91 (thousand tons per annum)[a]

Low hazardous waste: is moderately explosive, flammable, corrosive or reactive, or contains significant concentrations of constituents that are potentially highly harmful to human health or the environment.
Potentially hazardous waste: often occurs in large quantities, and contains potentially harmful constituents in concentrations that in most instances would represent only a limited threat to human health or the environment.

4. South African environmental legislative framework

Hazardous wastes, in particular, require more stringent regulatory and technical controls due to their toxicity, persistence, mobility, flammability, etc. There is increasing public concern about the numerous problems and potentially dangerous situations associated with hazardous waste management in general and disposal practices in particular.
South Africa has introduced a range of legislative measures aimed at improving the quality of the environment. The effective regulation of hazardous wastes requires sufficient compliance and enforcement capacity on the part of Department of Environmental Affairs.
Waste in South Africa is currently governed by means of a number of pieces of legislation, including:

- The South African constitution Act 108 of 1996
- Hazardous Substance Act 5 of 1973
- Environmental Conservation Act 73 of 1989
- National Water Act 36 of 1998
- National Environmental Management Act 107 of 1998
- Minerals and Petroleum Resources Development Act 28 of 2002
- Air Quality Act 39 of 2004
- National Environmental Management: Waste Act 59 of 2008

The Environmental Management Policy for South Africa sets a number of objectives for integrated pollution control and waste management system.

The objectives include:

- Promoting cleaner production and establishing mechanisms to ensure continuous improvements in best practices in all areas of environmental management.
- Preventing or reducing and managing pollution of any part of the environment due to all forms of human activity, and in particular from radioactive, toxic and other hazardous substances.
- Setting targets to minimize waste generation and pollution at source and promoting a hierarchy of waste management practices, namely reduction of waste at source, reuse and recycling with safe disposal as the last resort.
- Regulating and monitoring waste production, enforce waste control measures, and coordinating administration of integrated pollution and waste management through a single government department.
- Setting up information systems on chemical hazards and toxic releases and ensuring the introduction of a system to track the transport of hazardous materials.

The South African waste management principles aim:

- To secure the conservation of nature and resources, waste generation must be minimized and avoided where possible (prevention principle).
- To secure a reduction in the impacts from waste on human health and environment, especially to reduce the hazardous substances in the waste through precautionary principle.
- To make sure that those who generate waste or contaminate the environment should pay the full costs of their actions through the principle of pollute pays and producer responsibility.

In relation to the mining waste, the strategic focus in terms of waste hierarchy is on ensuring the treatment and safe disposal of mining waste. However, opportunities for reuse of mining waste need to be fully exploited.

The overall goal with regard to regulating waste invariably is to minimize health and environmental impacts with the concurrent optimization of economic and social impacts on society.

5. Best practice technologies and possible approaches

Integrated Waste Management (IWM) maintains that waste management can be planned in advance because the nature, composition and quantities of waste generated can be predicted. Advanced planning, means that an orderly process of waste management can ensue. This includes:

- **Waste Prevention**: the prevention or avoidance of the production of certain wastes, sometimes by regulation. Waste prevention initiatives address the industrial sector, by promoting the use of cleaner technology as well as schools and private households in broader awareness campaigns. As prevention has the highest priority in waste management principles, South Africa should make efforts in order to aim at reducing the quantity of waste generated.
- **Waste Minimization:** the economic reduction of the volume of waste during production, by means of different processes, or uses, or 'clean' technology implementation; Waste minimization is the application of a systematic approach to reducing waste at source.
- **Resource Recovery**: recycling of wastes of one process as raw materials, or the recovery of energy through incineration or biodegradation. Recovery contributes to utilizing the resources embedded in waste and contributes to saving raw material.
- **Waste Treatment**: contributes towards the reduction in hazardous character of the waste, or its volume, to ease environmental or human health risks and impacts;
- **Waste Disposal**: is the preferred and mostly used option. This has traditionally been by the disposal of waste to landfill sites. Land filling is ranked the lowest in the hierarchy of waste due to the lack of utilization of the resources in the waste, yet, it remains to be the most common waste treatment method in South Africa, (See Fig. I).

Waste management hierarchical practices that remain a key principle of our waste management are in Table 3 below:

Waste Hierarchy	
Cleaner Production	Prevention
	Minimization
Recycling	Re-use
	Recovery
	Composting
Treatment	Physical
	Chemical
	Destruction
Disposal	Landfill

Table 3. Hierarchy of waste

"In terms of implementing the waste hierarchy for industrial and mining waste, waste avoidance and reduction is of particular importance due to the significant environmental impact of this waste, and the potential harmful consequences for human health. Where hazardous waste cannot be avoided, emphasis needs to be placed on regulation, not only in defining standards for treatment and disposal, but also in ensuring reuse and recycling takes place in a safe and responsible manner". (Department of Environmental Affairs [DEA], 2009).

6. Priority options: Waste minimization, recycling and recovery

In line with international norms, the National, Provincial and Local Authorities, as well as society and industry at large, are encouraged, in cases by regulation, to seek to implement measures and means by which waste generation and disposal rates can be economically reduced, including the adoption of cleaner technologies, separation and reclamation/recycling of wastes (see Fig. 1).

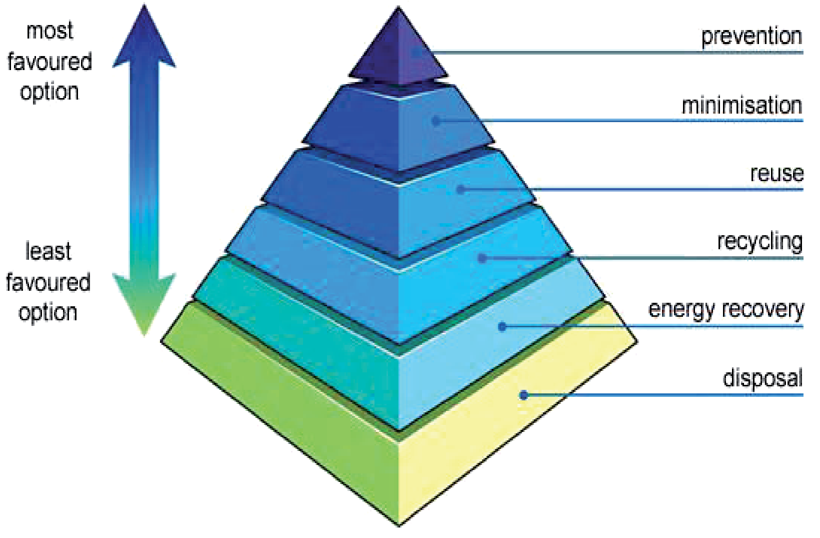

Fig. 1. The Waste Hierarchy

Waste minimizations involves a number of processes, mechanisms and stakeholders in the production, marketing, packaging, selling and consumption of goods that produce waste at all stages of the consumption cycle. By implication, it will require a conscious, comprehensive and intentional decision and effort by all stakeholders to ensure that waste and the secondary effects of poor waste management can be reduced through waste minimization to increase landfill site lifecycles and the environment. This may involve additional mechanisms and processes that include the following:

- Improving product and packaging designs to reduce resource consumption;
- Changing marketing and sales approaches to influence consumer perceptions and behaviour;
- "Extended Producer Responsibilities" (EPR) of producers of products, which may require producers to accept their used products back for recycling.
- Changing procurement policies and practices in large organizations that should encourage environmentally-aware production and manufacturing;
- Encouraging waste separation, streaming and diversion practices;
- Creating infrastructure to enable waste to be diverted from landfill sites;
- Developing infrastructure for processing waste for reuse/recycling;
- Developing markets for recycled materials and products;

7. Hazardous waste management

According to (Misra & Pandey, 2005), the management of hazardous wastes that has already been generated is one of the burning problems which require immediate attention. The principal objective of any hazardous waste management plan is to ensure safe, efficient and economical collection, transportation, treatment and disposal of wastes.

Steps towards effective management of hazardous wastes, and these are:

- Waste characteristics, including waste types, degree of hazards, chemical and physical stability, waste compatibilities, and the ability to segregate ignitable, reactive or incompatible wastes. To select suitable treatment and disposal techniques.
- Fate and transport characteristics of chemical constituents of wastes and their projected degradation products.
- The critical media of concern (such as air, surface water, ground water, soils/sediments, terrestrial and aquatic biota).
- Evaluation of potential release and exposure pathways of waste constituents and the potential for human and ecosystem exposures.
- Assessment of the environmental and health impacts of the wastes, if such waste reaches critical human and ecological receptors.
- Characterization of disposed sites, including site geology, topography, hydrogeology and meteorological conditions.
- Determination of extent of service area for proposed waste facility i.e. handling waste from local industry only or from regional and/or national generators.
- Suitability of proposed location for waste facility based on environmental, social and economic concerns including proximity to populations, ecological systems, water resources, etc.
- Best available technology (BAT) for handling the particular wastes. In addition, there should be contingency plans and emergency procedures in the design of waste management plans.
- Provision for effective long-term monitoring and surveillance programs including post-closure maintenance of facilities.

The capacity of a disposal facility is an exhaustible resource; however, the transportation of hazardous waste residue to disposal sites is a continuous process. In fact, the quantity of wastes arriving to a treatment/disposal facility may even increase over a period of time because of the industrial growth, unless waste minimization measures are implemented and enforced.

Rehabilitation of abandoned sites and re-entry therein and reuse also have to be done.

8. Treatment methods available

The purpose of treating waste is to convert it into non-hazardous substances or to stabilize or encapsulate the waste so that it will not migrate and present a hazard when released into the environment. Stabilization or encapsulation techniques are particularly necessary for inorganic wastes such as those containing toxic heavy metals.

Treatment methods can be generally classified as chemical, physical, thermal and/or biological.

Chemical methods - examples of chemical methods include neutralization, oxidation, reduction, precipitation and hydrolysis.

Physical methods - examples of physical methods include encapsulation, filtration, centrifuging and separation.

Thermal methods involve the application of heat to convert waste into less hazardous form. It also reduces the volume and allows opportunities for the recovery of energy from waste.

Biological methods involve the use of micro-organisms under optimised conditions to mineralise hazardous organic substances.

9. Landfill-disposal of hazardous waste

Disposal of the wastes is the final process and a key issue in overall hazardous waste management programme. The disposal facilities act as a permanent repository for the waste residues generated from the treatment facility. Even the most advanced treatment methods result in residues that are no longer amenable to cost-effective treatment.

The economics of waste disposal will determine, ultimately, the amounts and types of wastes that will be moved to distant disposal sites. The choice of disposal should be based on evaluation of economics and potential pollution risks.

The majority of domestic residential and commercial, business and industrial waste from urban areas is disposed to landfill sites. These landfill sites are generally operated by the local authority in whose area the site is located, or by private service providers. Although some of the industrial waste is handled by local authority services, and private service providers handle much of this stream. Most of the waste generated by industry (especially metallurgical) and agriculture are disposed of disposed of on the industrial or agricultural premises, with little information available on quantities, qualities or management thereof.

There are several environmental impacts from landfills. One impact is contribution to the greenhouse effect through the emission of methane gas. Leachate may also damage groundwater if there is no liner system. Other impacts include odours and general inconvenience for neighbours to landfill sites.

Waste management is emerging as a key sector for sustainable development in South Africa with opportunities for enhancing investments in carbon credits that target reduction of methane from landfills and moveable assets in relation to environmentally sound equipment required for effective waste management. It is true that the focus is towards two key areas for investments include capturing methane emissions from landfills for trading in carbon markets and financing both physical and moveable assets to enhance sustainable development. However, the challenges for cost-effectiveness, efficiency and sustainability in the sector prevail in relation to lack of sound knowledge to design and implement integrated programmes that incorporate environment, development and sustainability. Henceforth, financial resources according to (Karani & Jewasikiewitz, 2007), are imperative to waste management and sustainable development as the sector requires capital investments for necessary infrastructure.

10. Environmental and social impacts

According to (Adler, 2007) since negative externalities associated with mining were not internationalized under apartheid, the mining industry failed to adequately prepare for closure and to dispose of mine water and waste in a manner that is consistent with current international best practice.

Following the transition to democracy, government faces conflict caused by the legacy of weak regulation that has exaggerated problems associated with limited natural resources. In particular, cumulative harm to off-mine populations resulting from modified water tables, contaminated ground water sources, acidic mine drainage, and ground instability must be addressed before they lead to even more devastating socioeconomic, political, and environmental damage.

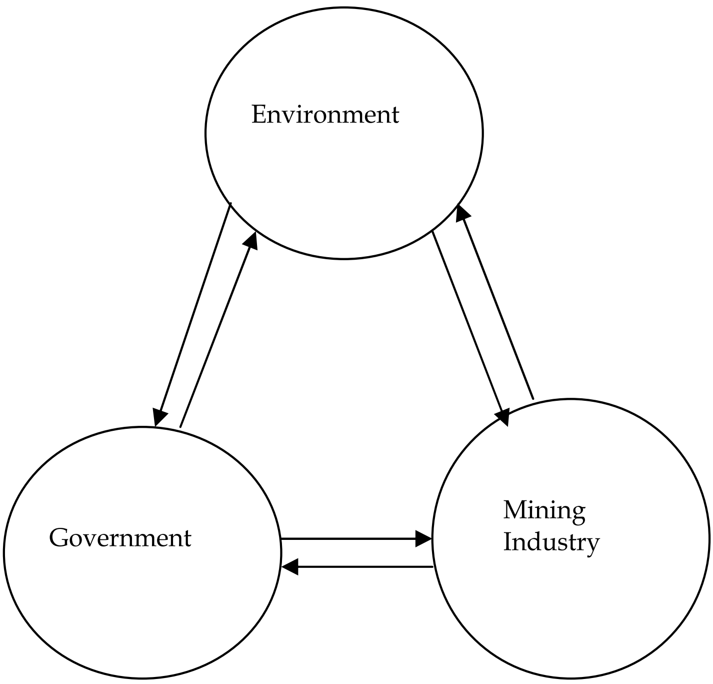

Fig. 2. Adapted Trialogue Model.

The trialogue model captures interactions among (1) government, (2) mining industry, and (3) environment. The environment includes society, economy, and the natural environment. Each sector places pressure on the others, as represented by the double arrows.

The outcome of these effects can be described in terms of governance Trialogue Model (Figure 2). It shows how regulation (or lack thereof) can result in conflict among industry, government, and environment (which includes society-at-large).

In the case of South Africa, new policies have been drafted by government to address these issues, but in most cases the regulation of mining-related activities is fragmented throughout multiple pieces of legislation, to be enforced by various agencies at the national, provincial, and municipal levels.

The impacts of non-sustainable waste management are difficult to quantify, however, potential consequences may be identified and include the following:

- Long term effects of pollutants entering the surface or groundwater resources, air and soil affecting the fitness for use, and availability of the resource for use. More specifically:
- Pollution of watercourses and groundwater by leaching of pollutants from waste inappropriately disposed of, or where waste management service provision is inadequate, particularly evident for dense urban informal settlements.
- Pollution of watercourses and groundwater by leaching of pollutants from waste residue deposits, particularly mine and power station waste dumps.
- Air pollution by dust releases from particularly mine residue deposits, but also general and hazardous waste sites (methane gas production) and HCRW incinerators.

- Nuisance from odours of waste degradation in landfill sites, waste disfiguring the environment especially plastic bags, and littering where waste service provision is limited.
- Reduced biological diversity in the areas of waste management operations, as a result of land disturbance or effects of emissions and discharges from the waste facilities.
- Increased waste management costs to provide safe and effective long-term disposal sites for increasing waste loads, including treatment of wastes to render them less environmentally available, and effective closure and rehabilitation of historically inadequate waste sites.
- Increased pressures through the negative societal impacts of inadequate service provision fostering illegal waste dumping, littering and abuse of open spaces.
- Increased health and environmental risks associated with inadequate waste collection and disposal services, and informal salvaging on landfill sites.
- Poverty encourages salvaging on waste sites for recyclables, refuge materials, fuel and food.
- Environmental risks as many waste sites which do not meet the Minimum Requirements stipulated by DWAF, requiring upgrading to the specifications, or closure and rehabilitation.

Although hazardous waste is produced by practically all areas of society, some of the worst waste produced, with a legacy of the poorest controls, comes from the mines and industries. Some of these contaminants are discharged into the aquatic environment.

The consequences and impacts of waste management inherently link to other indicators of environmental health and sustainability, particularly:

- Water resource, the focus being on water quality deterioration and pollution;
- Biodiversity;
- Social environment, the focus being on human health;
- Air quality, the focus being on visual and odour nuisance; and
- Land, the focus being on provision of suitable locations for landfills and waste services.

11. Economic impacts

According to (Wiebelt, 1999), while in many developed countries mining has been relegated to the status of an ugly old industry of little importance to the national wealth, the highly mineralized nature of many parts of South Africa has led to the creation of a mining industry which is quite important to the country's economy.

If the value of processed mineral products such as refined base metals, ferroalloys, iron and steel, and refinery products produced from coal were included, about 60 percent of South African export revenue would have come from mineral-based products.

The Department of Environmental Affairs proposed `eco-taxes', whereby polluters are charged equal to their hazardous waste treatment costs allow the realization of any technologically possible environmental objective at minimum social costs. The analysis is based on a study by the Department of Environment Affairs on Hazardous Waste in South Africa which among others estimates hazardous and non-hazardous waste streams for different sectors (Department of Environmental Affairs [DEA], 1992a) and assesses the economic impact of alternative policies towards hazardous waste management (Department of Environmental Affairs [DEA], 1992b).

The economic impacts of hazardous waste may be clustered along the following three categories:

- The environmental tax on hazardous mining waste will lead to an adjustment of factor demand and final demand and, therefore, to an environmentally more sound use of natural resources.
- Closely connected with the environmental impacts are the economic impacts of the environmental tax. Higher costs for waste management lead to changes in macroeconomic aggregates which have to be included in the analysis. Income and substitution effects will change the international competitiveness of individual sectors as well as the sectoral structure of the economy.
- The taxation of hazardous mining waste will yield higher tax revenues, higher tax revenues.
- Economic instruments such as environmental taxes and subsidies should provide incentives for waste generators and service providers to reduce waste generation and to seek alternatives to final disposal to landfill such as re-use, recycling and recovery. There are opportunities that are associated with the implementation of economic instruments and they include:
    - Potential to reduce the need for landfill airspace and prolong the lifespan of landfill sites;
    - Their potential to stabilise prices of recyclables and thus stimulate and stabilise viable and sustainable markets for recyclables;
    - The socio-economic benefits associated with recycling such as local economic development and the creation of job opportunities in the recycling market;
    - Improved environmental awareness; and
    - The potential to encourage private investment.

12. Conclusion

South Africa has developed waste regulations; and awareness has been created for the management of hazardous wastes; however, effective practice for safe management still needs to be enforced.

To effectively manage waste, public-private partnership should be encouraged to jointly address waste management problems.

The partnership mechanisms would address the following:

- Significantly reducing load of hazardous waste to landfills.
- Finding alternative uses for industrial waste generated in significant quantities with a high potential for environmental pollution.
- Addressing the problem of reluctance from industries to disclose their hazardous waste streams and volumes.

In trying to deal with waste management challenges in South Africa, it is important to rigorously

- Consider both recycling and waste minimization
- Consider extended producer responsibility as a means to emphasize waste minimization
- Explore opportunities for energy recovery
- Ban some waste streams from landfill sites.

An obligation should be made to monitor landfills during their operation and up to 30 years after their closure. The monitoring must include measurement of landfill runoff, emissions of landfill gas, the level of water table and ground water quality under and near the landfill.

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6

Exploring and Assessing Innovative Approaches to Utilizing Waste as a Resource: Toward Co-Benefits

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1. Introduction

Waste is an inevitable byproduct of human activity. In the last two centuries, waste management has passed through a series of transitions in terms of treatment and disposal technologies, as well as in administrative systems and in people's attitudes. Waste managers, engineers, planners, and researchers have contributed to these transitions by responding to issues such as public health, disposal capacity, more-rigorous environmental standards, and public and political pressures (Louis, 2004; Tarr, 1985). More recently, studies on waste management have emphasized the 3Rs (Reduce, Reuse, and Recycle). As a result of the efforts of practitioners and researchers, considerable achievements in waste handling have been realized in a number of countries. The questions that we currently face are "What forthcoming issues related to waste management do we need to respond to?" and "What will be the appropriate methods for addressing these issues in research?" In this chapter, we will explore some of the pressing issues, and we will propose a research framework for assessing the options available for responding to them.

2. Shifting toward utilizing waste as a resource

Nowadays, the macro-level pressures that affect waste management are more diverse and more complicated than ever, while the pressures that previously drove transitions in waste management remain. For example, the impact of waste on public health continues to receive great attention. Wastes that contain hazardous materials, such as waste electrical and electronic equipment, scrap automobiles, and medical waste, require special treatment and disposal (Achillas et al., 2010; K. C. Chen et al., 2010; Jang et al., 2006). The shortage of disposal capacity, especially landfilling capacity, continues to be a major driver for volume reduction and for diversion of waste from landfill (Bai & Sutanto, 2002; Geng et al., 2010; Jin et al., 2006). Despite progress in technologies and the strengthening of environmental standards, the problems of NIMBY ('not in my backyard') attitudes and political pressure remain unsolved. For example, as a result of insufficient public participation during the planning stage, difficulties were encountered in siting of new landfills in Ontario, Canada, and exporting of waste to Michigan in the USA for landfilling caused political tensions between the two countries (Hostovsky, 2006). Similarly, public opposition in Beijing resulted in the cancelation of the construction of a new incineration plant in 2007 (SEPA, 2007).

In addition, pressures arising from environmental and economic problems have rapidly come to prominence in recent years, and these have become new drivers for further waste reduction, reuse, and recycling. Depletion in resources has encouraged recycling of scarce materials (e.g. rare metals) from wastes or, more generally, urban mining, i.e. recycling of resources from urban stock (Klinglmair & Fellner, 2010; Ongondo et al.). As the development of the recycling market has created new business opportunities, economic drivers have come into play. For example, in the USA, eco-industrial development, including the encouragement of industrial symbiosis and the development of eco-industrial parks, was originally considered to be an economic development strategy (Deppe et al., 2000). The eco-town program in Japan had the dual objectives of solving waste-management problems and stimulating industrial development (van Berkel et al., 2009). In China, a circular economy based on reduction, reuse, and recycling appeared to be a practical strategy for sustainable development of the economy and society (Yuan et al., 2006). Currently, attempts to mitigate climate change affect decisions on a wide range of environmental and economic activities, including waste management. Cleary (2009) reviewed 20 studies on life cycle assessment (LCA) in waste management recently published in English-language peer-reviewed journals and found that 19 of these studies assessed global warming potentials, i.e., emissions of anthropogenic greenhouse gases (GHG). In practical terms, carbon credits provide an incentive for waste disposals in a manner that reduces GHG emissions in comparison with conventional practices. For example, as of February 24, 2011, of 2845 projects registered under the United Nations Framework Convention on Climate Change's Clean Development Mechanism (UNFCC CDM), 516 (18%) involved waste handling and disposal; this is the second largest category, following that of the energy industry (http://cdm.unfccc.int/Projects/projsearch.html).

Under these circumstances of multidimensional pressures on waste management, merely diverting wastes from landfills and increasing recycling rates might no longer be a sufficient response. The combination of pressures demands an improvement in the ecoefficiency of recycling and in utilizing wastes as resources to fulfill multiple purposes or, in other words, seeking co-benefits from waste management. Admittedly, local conditions in various countries and regions differ from one another and, as a result, they might have different priorities in terms of their objectives. Despite these differences, however, there is a common goal of improving efficiency in processing and utilization of recyclable wastes after source separation to achieve greater environmental and economic benefits.

3. Assessment of waste treatment and disposal

To improve the efficiency of recycling, it is important that various options be considered and compared during the planning stage. Among various evaluation methods, LCA is a methodology that is widely used in assessing impacts of waste management. LCA can be used to assess and compare the potential impacts of various treatments and disposal methods on the waste hierarchy (Banar et al., 2009; Finnveden et al., 2005; Liamsanguan & Gheewala, 2008). LCA can also be used to evaluate the applications of one method or of one type of facility on different scales (Habara et al., 2002; Lundie & Peters, 2005;

Wanichpongpan & Gheewala, 2007). It can also be used to assess various treatment methods for a particular type of waste (Al-Salem et al., 2009; Cadena et al., 2009; Lundie & Peters, 2005). In most of these studies, the LCA methodology is used to assess the possible consequences of certain decisions (e.g., applying different treatment methods or establishing facilities in different locations or at different scales) by setting up multiple scenarios that represent the various options. Such an approach is often referred to as change-oriented or consequential LCA, and it describes how environmentally relevant physical flows might change in response to possible decisions (Ekvall & Weidema, 2004; Finnveden et al., 2009).

The consequential LCA method also fits the purpose of our research: to identify the co-benefits of efficient utilization of waste. For this purpose, we need to be able to consider several aspects when simulating possible consequences. The first aspect involves the efficiencies and impacts of recycling technologies. A number of studies have been performed in this area. Unlike landfill and incineration, for which most countries have already issued technical standards, recycling involves a combination of many different technologies with no clear standards. For example, waste plastics can be treated through various mechanical recycling, chemical recycling, or energy recovery processes (Al-Salem et al., 2009); sewage sludge can be treated by agricultural landspreading, incineration, wet oxidation, pyrolysis, incineration in cement kilns, or anaerobic digestion (Houillon & Jolliet, 2005); and food waste can be treated by composting, anaerobic digestion, or wet or dry feeding (Kim & Kim, 2010; Levis et al., 2010). These technologies co-exist for a combination of economic and environmental reasons. No single technology appears to dominate in practice, and research efforts have been made to evaluate these technologies from various perspectives.

The other aspect that needs to be considered is that of policies related to recycling, waste reduction, and source separation. LCA studies on waste management typically assess the impacts of managing waste on a unit-weight basis (per kg or per ton) (Ekvall et al., 2007). Results from such studies can be readily compared with one another to identify efficient technologies, but they do not reflect different waste-management policies. In addition to treatment technologies, waste management also involves various regulatory and economic instruments, for example, "pay-as-you-throw" policies, designed to encourage waste separation and recycling. It is therefore necessary to account for the total amount of waste generated in a municipality or a region as a functional unit and to consider the potentials for reduction and for recycling if appropriate policies were to be implemented. In the next section, we introduce a simulation system that combines alternative environmental technologies and policies; we also present two examples of applications of this system.

4. Research framework and examples of application

Fujita and his co-workers have developed a simulation system for assessing urban environmental technology (Fujita et al., 2007; Nagasawa et al., 2007; Wong et al., 2008). The model consists of three main parts: a database, a technology inventory, and a set of environmental policy options. The application of this simulation system is not limited to waste management. Some examples related to recycling are illustrated in Figure 1. For better evaluation of the environmental impacts and costs of waste collection and transportation, the database is built on the basis of a geographic information system (GIS) when digitally based maps and spatial distribution data are available, for example, for road networks and the distribution of populations and waste generation. The technology inventory contains input and output data on waste recycling and disposal technologies, as well as emission factors, and it embodies environmental impacts of utilities. Finally, the model contains policy options (including various waste-reduction and recycling policies) that could be implemented in the city being studied.

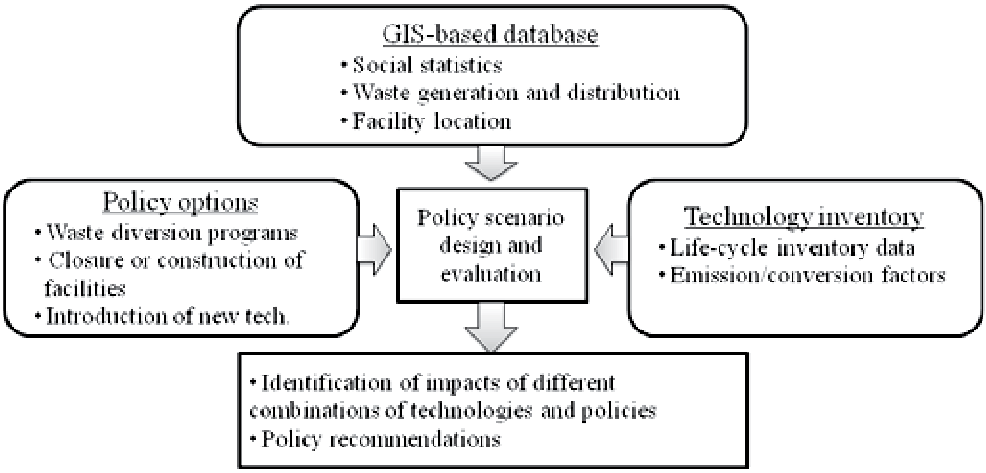


Fig. 1. Urban environmental technology simulation system for recycling

Scenario design and evaluation are based on the LCA approach in terms of selecting the system boundary and functional unit for assessment. Provided that data are available, the impact categories can contain key aspects that are of concern to the city under study, including environmental impacts and economic costs and benefits. For various combinations of policies and technologies, one can set up multiple scenarios to simulate possible consequences. This simulation system can help decision makers to assess potential consequences of alternative options. Below, we present two examples of our recent studies. The first was a case study on options for expanding recycling programs by utilizing existing facilities in Kawasaki, Japan, and the second was a case study on transferring advanced recycling technologies to Shenyang, China.

Case Study 1: Economic costs and environmental impacts of expanding recycling programs by utilizing existing industrial facilities in Kawasaki

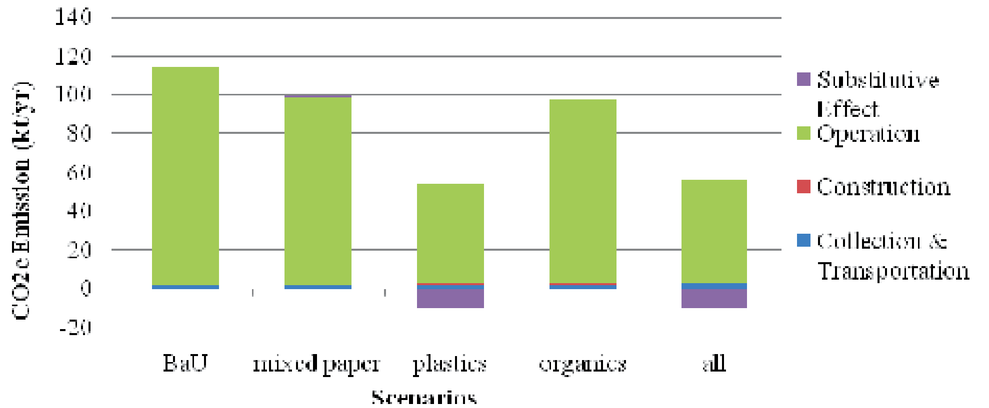
Kawasaki city is the ninth most populous city in Japan, with a total population of 1.4 million, and is located between Tokyo and Yokohama. This area is one of the industrial cores of Japan that supported the country's industrial development and economic growth during the last century. Kawasaki is also a front-runner in environmental protection and recycling businesses. The Kawasaki Eco-Town was among the first group of eco-towns designated in 1997 and, as a result, a number of recycling facilities are located there. Five facilities were subsidized by government, including facilities for recycling waste plastic as a reductant in blast furnaces for iron production, for recycling of hard-to-recycle paper (e.g., train tickets or confidential documents in sealed boxes) to produce toilet paper, for recycling of polyethylene terephthalate (PET) to monomers for re-synthesis of PET, for recycling of waste plastics to produce syngas for ammonia production, and for recycling of waste plastics to produce concrete formworks (GEC, 2005). Nonsubsidized recycling facilities in Kawasaki include a facility for recycling household electronic waste and a cement plant that uses wastes such as dehydration cake from wastewater treatment, ash from sludge incineration, waste plastics, and wood chips as feedstocks and fuels for cement production.

With these recycling and industrial facilities, a number of industrial and urban symbiotic networks have been established (for details, see Geng et al., 2010). These industrial facilities still have sufficient spare capacity to receive additional wastes from municipal sources. The city government has examined the possibility of expanding recycling programs and utilizing greater quantities of recyclable wastes from municipal sources in these facilities.
With regard to the existing facilities, we considered recycling of mixed waste paper and non-PET packaging waste plastics from households, and of organic wastes from commercial sources such as cafeterias, restaurants, and convenience stores. These wastes are currently incinerated but, if properly separated, could be recycled at the existing facilities. The data available in Kawasaki allowed us to develop a spatial database for the distribution of the population in 1-km meshes, as well as for the positions of commercial facilities and of waste recycling, treatment, and disposal facilities. By assuming that the per capita generation of waste plastics and paper is the same for all people, we projected the distribution of the relevant wastes and we calculated the transportation distance based on the road network. We set up four scenarios, in addition to the business-as-usual (BaU) scenario, to evaluate the recycling of each type of waste separately and of all three types together (Table 1).

Scenario	Additional recycling program	Technology
BaU	n/a	n/a
Mixed paper	Mixed paper from municipal sources	Producing toilet paper
Plastics	Mixed packaging plastics from municipal sources	Utilization as reductants in blast furnaces for iron production
Organics	Organic waste from commercial sources	Recycling through anaerobic digestion to production biogas for electricity generation; residue used for cement production
All	All three of the above	All three of the above

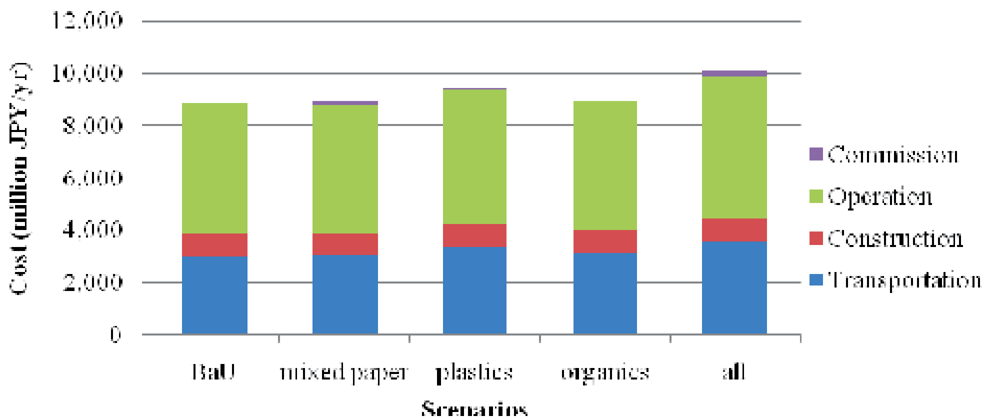
Table 1. Scenarios for expanding recycling programs in Kawasaki, Japan

The system boundary for the assessment encompasses (1) waste collection, transportation, and pretreatment of waste, (2) the processing process, (3) the embodied environmental burden of the replaced products, and (4) the impacts of the treatment and disposal of waste that are avoided as a result of recycling. The functional unit is the total amount of waste in Kawasaki in 2015. Recycling rates of mixed paper, mixed plastics, and organic wastes were set at 62%, 69%, and 30% respectively in accordance with the objectives for 2015 set by the Japanese government. A total of 21 kilotons of waste paper, 18 kilotons of waste plastics, and 15 kilotons of organic wastes are expected to be separated in 2015 (for details, see Geng et al., 2010). The remaining garbage is sent to the four incinerators in Kawasaki that are equipped for electricity generation and heat recovery. To reduce transportation costs and to improve the separation of waste paper and plastics, transfer centers would be required for storing, compressing, and baling the materials. We assumed that two such centers would be built to handle waste paper and waste plastics, respectively. Source-separated waste paper or/and plastics would be collected by trucks operating on the road network and would be delivered to the nearest transfer center, from where they would be transported to existing recycling facilities. In the "organics" scenario, we assumed that a biogas plant capable of generating power from methane generated by fermentation of organic waste would be built. We also assumed that these new facilities would be capable of operating for 25 years without major renovation; the initial impacts of construction were therefore averaged over 25 years. We focused on three categories of impact that are of great concern in Japan: GHG emissions (i.e. global warming potential), landfill reduction, and financial costs. In a similar manner to our assessment of environmental impacts, we also considered the costs of collection, transportation, and pretreatment, as well as the costs of construction and process operation, and the small commission costs for industrial facilities to receive wastes. Initial investments for new facilities were also averaged over 25 years.



[Source: by the authors based on previous results (Geng et al., 2010).]

Fig. 2. GHG emissions for various recycling expansion scenarios in Kawasaki



[Source: by the authors based on previous results (Geng et al., 2010)].

Fig. 3. Financial costs of recycling for various expansion scenarios in Kawasaki

The results show that emissions of up to 70 kilotons of CO_2-equivalent could be eliminated annually (Figure 2). Operations, particularly the incineration process, contribute most to the

CO_2 emissions. Waste collection and transportation and the construction of new facilities account for only a small proportion of the total emissions (below 5% in all scenarios). Reduction by material substitution varies markedly among the scenarios, depending on emissions from processing and the embodied emissions in the materials replaced. Recycling plastics results in a considerable reduction in CO_2 emissions because it replaces coke, a carbon-rich feedstock for blast furnaces. By recycling mixed paper, plastics, and organic waste, inputs to landfill can be reduced by 3.2, 2.7, and 2.3 kilotons, respectively. Unsurprisingly, the total costs of waste management would increase on launching additional recycling programs (Figure 3). With such investment in new recycling programs and facilities, the city could gain environmental and economic co-benefits. Besides reductions in GHG emissions and in wastes to be landfilled, there are several benefits that are beyond the assessment boundaries of this study, such as reductions in the consumption of coal and coke, creation of new jobs, and stimulation of industrial development.

Case Study 2: Environmental impacts of transferring recycling technologies from Japan to China

A lack of efficient waste-treatment technologies is a frequent problem in developing countries. One shortcut for resolving this problem is the transfer of the necessary technologies from developed countries. Although introducing advanced technologies can, to some extent, provide environmental benefits, it is still necessary to understand which technologies are suitable for a particular locality. In this case study, we investigated the potential environmental impacts of transferring waste-plastic recycling technologies from Japan to Shenyang City, China.

Shenyang City is the capital city of Liaoning Province in northeastern China, with a total population of 7.8 million in 2009. The Shenyang Sanitation Research Institute reported that about 3 million tons of municipal solid waste (MSW) were generated in Shenyang in 2008, of which 2.13 million tons were generated within the urban area (Sun et al., 2008). Landfill is currently the major method of disposal of MSW. In the urban area, 1.85 million tons of MSW were landfilled in 2008 (Shenyang Statistical Yearbook, 2009). The remainder (0.28 million tons) was recycled, sold to the secondhand market, or dumped illegally. Taken together, plastics and rubber form the second largest category of MSW after food waste, accounting for 15% of the waste to be landfilled.

Because recyclable wastes have mainly been collected by the informal sector in China, data on the total amounts and characteristics of recyclable wastes are difficult to obtain (Chen et al., 2010). For better management of renewable resources, the Shenyang Supply and Marketing Cooperative Association (SMCA) undertook a comprehensive survey and an onsite investigation on renewable resources in Shenyang in 2009. The investigation (Wang et al., 2009) showed that the total amount of waste plastics produced in Shenyang in 2008 was 631 kilotons, of which 621 kilotons were traded in two marketplaces, and 10 kilotons were delivered to processors directly from redemption centers and junk-buyers. About one-third of these waste plastics consisted of PET, 5% was polystyrene foam, and the remainder consisted of polyethylene (PE), polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC), and acrylonitrile–butadiene–styrene resin (ABS), among others. These waste plastics were collected from various sources, including domestic and foreign sources (for details, see Chen et al., 2011). Collected waste plastics were usually manually separated, washed, and shredded or granulated to form pellets or granules. Most such treatments were undertaken in small informal workshops without any controls over emissions. Approximately one-third of the plastic pellets were utilized in Shenyang. All the PET and more than half the mixed plastics were transported to other provinces. Hebei province, over 800 km away, was the major destination, receiving around four-fifths of the exported waste plastics.

We focused on the exported non-PET plastics, because these could be utilized in more efficient ways locally if advanced treatment technologies were available. Japanese eco-towns have developed a number of plastic-recycling technologies (Van Berkel et al., 2009). Waste plastics can be converted through mechanical recycling, chemical recycling, or energy-recovery processes into products that replace virgin materials, such as plastic resins, lumber, fossil fuels, or feedstock for industrial facilities (Al-Salem et al., 2009; JCPRA, 2007). In the existing recycling process [the business-as-usual (BaU) scenario], waste plastics are shredded into pellets and granules. We assumed that PP and PE are recycled to produce plastic resins (50% PP and 50% PE), and the remainder are assumed to replace wooden products. This recycling process is actually down-cycling, and therefore a 10% material loss and a 20% of loss in quality were assumed (Astrup et al., 2009). We examined four alternative technologies. The first was recycling of plastics to produce plastic boards (known as NF boards) as replacements for wooden boards for concrete formwork. The second was the production of refuse plastic fuel (RPF) from shredded plastics as a source of energy to replace coal or other fossil fuels. The third was gasification of plastics to produce syngas to replace natural gas for the production of ammonia. The final technology involved the use of waste plastics material as a reductant in blast furnaces for the production of iron. Input and output data for these technologies were based on facilities in Japan and were adjusted according to the composition of waste in Shenyang (see detailed LCI data in Chen et al., 2011). Only appropriate compositions were treated by these various technologies. For each recycled product, products with an equivalent function were determined.

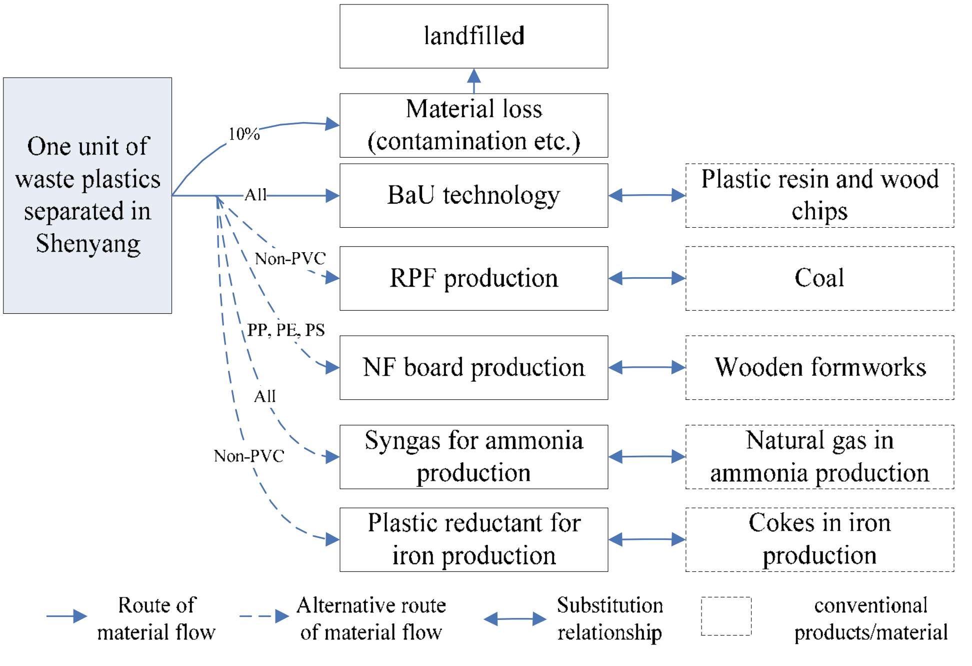


Fig. 4. Recycling technologies and substituted products with equivalent function

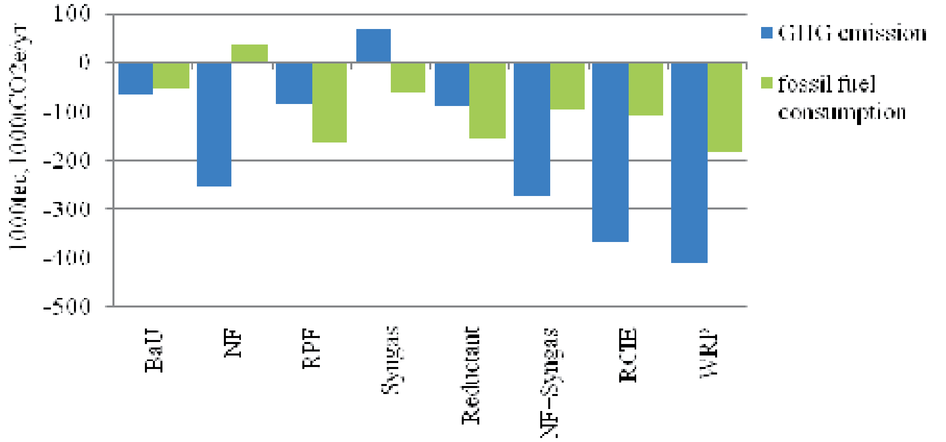
As in the previous case, the system boundary for the assessment included waste transportation, processing, the embodied impacts in the replaced products, and the avoided impacts of waste disposal. Waste plastics were considered to be delivered by freight trucks, as presently. Because Shenyang is located in a heavily industrialized region, it is surrounded by many large industrial facilities. An ammonia plant with an annual output of 322 kilotons is located in Panjin, near the Liaohe oilfield (about 160 km southeast of Shenyang), and two iron plants with a total annual output over two million tons are located in Anshan (110 km from Shenyang) and Benxi (80 km), respectively. We assumed that recycled reductants and syngas could be used in these facilities. RPF and plastic formworks (NF boards) were assumed to be consumed locally. In addition, this case involves "open-loop" recycling technologies that produce products that replace various types of product on the open market (e.g., plastic formworks replacing wooden formworks). Because the compositions of the recycled product and the substituted product differ from one another, their disposal was also included. Because we focused on waste plastics in this case, the functional unit was waste plastics exported to other provinces and landfilled without utilization. The total amount of the former is about 210 kilotons, and that of the latter was estimated to be about 185 kilotons. The impacts assessed in this case included global warming potential (measured in terms of emissions of tons of CO_2-equivalent, or tCO_2e) and fossil-fuel savings (by ton coal-equivalent, or tce). These categories of impacts are currently of great concern in China because, at the 2009 United Nations Climate Change Conference, held in Copenhagen, Denmark, China pledged to reduce the intensity of carbon dioxide emissions per unit of gross domestic product (GDP) in 2020 by 40–45% compared with the level in 2005, and to increase the proportion of nonfossil fuels in its primary energy consumption to approximately 15% by 2020 (Xinhuan News Agency, 2009).

In addition to considering the introduction of technologies, we also considered the impacts of cascading use of waste plastics by mechanical and chemical recycling technologies, lowering the embodied impacts of electricity through national efforts to promote renewable energy, and launching new waste-plastic recycling programs to collect material currently being landfilled but which could be utilized by Japanese technologies (Table 2). The recycling rate of the new programs was assumed to be 50%.

The results for eight scenarios are summarized in Figure 5. The NF board-production scenario showed the largest potential among the individual technologies in terms of reducing GHG emissions (254 kilo-tCO_2e/yr). Much of this reduction is attributed to avoiding landfilling of wooden formworks, because wood undergoes partially biodegradation to form methane. The production of RPF provides the greatest saving in fossil-fuel consumption among the individual technologies, as it directly replaces coal, and the production process consumes little energy. Synergies between various technologies could provide additional environmental benefits. As shown in the 'NF + RPF' scenario, cascading utilization of waste plastics could bring greater reductions in GHG emissions than could any individual technology. The final two scenarios tested the impacts of changes in the carbon intensity of electricity and the operation of new waste-recycling programs. These two factors are closely related to the potential environmental gains achievable by applying the technologies. In comparison with the 'NF + RPF' scenario, a 15% reduction in the carbon intensity of electricity could lead to an additional 93 kilo-tCO_2e/yr (34%) reduction in GHG emissions and a 12 kilo-tce/yr (12%) saving in fossil fuels. These results indicate the existence of a synergy between clean energy and recycling/energy recovery technologies. As an industrial activity, the recycling process itself consumes electricity. If the carbon intensity of the electricity were to drop, recycling would achieve a greater reduction in GHG emissions whereas collecting landfill gases for electricity production would achieve a smaller reduction. The waste-recycling program could potentially permit the use of 92 kilotons of plastics that are currently landfilled. This new recycling program would result in an additional 41 kilo-tCO_2e/yr (11%) reduction in GHG emissions and a 72 kilo-tce/yr (68%) saving in fossil fuels.

Scenario	Introduction of alternative technology	Carbon intensity of electricity	New recycling program
BaU	–	–	–
NF board (NF)	Recycling waste plastics to produce NF boards	–	–
RPF	Recycling waste plastics to produce fuel to replace fossil fuels	–	–
Syngas for ammonia production (Syngas)	Gasifying waste plastics to produce syngas for ammonia production	–	–
Reductant	Recycling waste plastics to produce reductant for iron production	–	–
NF + RPF	Recycling waste plastic to produce NF boards and producing fuel to replace fossil fuels from used NF boards	–	–
Reduced carbon intensity of electricity (RCI)	As above	The carbon intensity of electricity is decreased by 15%	–
Waste recycling program (WRP)	As above	The same as in 'reduced carbon intensity' scenario	Rolling out new a recycling program to divert waste plastics from landfill

Table 2. Scenarios for transferring waste-plastic recycling technologies to Shenyang, China



[Source: by the authors based on previous results (Chen et al., 2011)]

Fig. 5. GHG emissions and fossil fuel consumption for the various scenarios (BaU: business as usual, NF: NF board production, RPF: refuse plastic fuel, RCIE: reduced carbon intensity of electricity, WRP: launch of waste recycling program)

5. Discussions

The two cases discussed in this chapter confirmed the view that recycling would lead to greater environmental benefits than incineration and landfilling. If proper technologies are chosen and there are dependable supplies of separated wastes and a reliable demand for recycled products, more recycling would provide additional environmental benefits. However, by recycling one unit of waste, different recycling technologies would realize different benefits. Their efficiencies vary because of differences in the properties of the treated wastes (such as its composition and level of contamination), in the properties of the replaced products (embodied environmental impacts), in the efficiency of processing, in emission factors, and in the current practices taken as baselines for measuring reductions. As shown in the two cases discussed above, recycling of mixed paper, plastics, and organic wastes can result in different levels of reduction in GHG emissions. Furthermore, recycling plastics by different technologies also leads to different levels of reductions in GHG emissions and in fossil-fuel savings. The compositions of waste plastics in Shenyang and Japan provide different results. Although PET bottles are usually separated because of the high market value of PET pellets and fibers, the compositions of the remaining mixed plastics remain quite different. Because of these differences, the same recycling technology would have different product yield ratios and energy efficiencies in Shenyang than in Japan. Emission factors in the two cases are also different. For example, the embodied GHG emissions for electricity in Shenyang are 1.13 kg-CO_2e/kWh, whereas those in Japan are only 0.55 kg-CO_2e/kWh, because the former are generated mostly in coal-fired power plants and the latter come from multiple sources, including carbon-free sources such as nuclear power and solar energy.

Composition		PE	PP	PS	PET	PVC	Other	Moisture
in Shenyang*		13%	13%	3%	31%	13%	25%	3%
in Japan**		30%	21%	18%	14%	5%	5%	7%
LHV**	kJ/kg	46046	43953	40186	23023	24070	–	–2512
CO_2**	kg-CO_2/kg	3.143	3.143	3.385	2.292	1.408	–	0

Source: * (Chen et al., 2011); ** (JCPRA, 2007).

Table 3. Waste plastic compositions in Shenyang and Japan

Moreover, when counting the avoided impacts of waste disposal, one has to determine how the waste might otherwise be disposed of under typical conditions (usually the common practice at the time). These typical conditions become the baseline against which emission reduction and other environmental benefits are counted. The baselines in different countries can be different, and therefore the benefits of the same technology can also be different. As shown in the two examples discussed above, whereas in Japan the baseline is incineration, in China it is landfilling. Consequently, recycling waste composed of anthropogenic carbon (e.g., plastics) in Japan would result in greater reduction in GHG emissions that in China, because incinerating plastics releases anthropogenic GHG emissions, whereas landfilling plastics, as in China, releases no emissions. On the other hand, recycling biodegradable wastes such as organics and wooden wastes in Japan results in a smaller reduction in GHG emissions, because emissions from incinerating organics would be considered carbon neutral, whereas landfilling organics would produce landfill gases that contains methane, which according to the IPCC Guideline has 21 times the global-warming potential of CO_2. Consequently, the environmental benefits of recycling technologies depend on local conditions, and it is necessary to take these conditions carefully into consideration in case studies.

Methodologically, there is considerable scope for improving the simulation system applied in the case studies above. First, it fails to address several important factors that affect future scenarios. For example, the models analyze scenarios based on the present conditions or on planned conditions in the future, but they cannot project changes in the amount of waste generation and composition. The factors that influence waste-generation rates include household size, residency type, age groups, employment, tipping fees, GDP, education, culture, geography, and climate (Shan, 2010). As many countries are (or will) experience aging and shrinking of their populations or will undergo rapid economic development, some of these factors will change dramatically. Waste generation and composition are also expected to change. Some of these factors can be internalized in the models to make the functional unit more dynamic and closer to reality. Secondly, during our involvement in real projects entailing transfer of recycling technologies from Japan to China, we realized that, as a compromise between high efficiency and the high costs of equipments and facilities, in some cases only partial set of facilities, usually the core parts, are transferred, and these are supplemented by local equipment or by manual work. In such cases, data on input/output and costs derived from surveys on existing facilities need to be amended accordingly. More importantly, by analyzing the costs and environmental impacts of various combinations of transferred and local technologies, we could provide useful information for decision makers to evaluate various options.

6. Conclusions

During the last century, researchers have concentrated on finding engineering solutions to problems of waste management, particularly those of treatment and disposal. However, engineering solutions are not capable of solving all problems of waste. In many cases, they only eliminate the symptoms and do not touch the core problems of how to reduce waste generation in the first place and how to recycle waste efficiently to the industrial-production system. As a multitude of pressures on waste management and recycling arise, attitudes toward waste and waste management need to shift toward managing and utilizing waste as a resource, thereby seeking greater co-benefits for the environment, the economy, and society. This chapter describes a simulation system that can be used to assess urban environmental technologies; this system could become a flexible tool that would permit decision makers to evaluate the impacts of various technological and policy options. Because this approach examines future scenarios contingent on various local conditions, it is difficult to establish common databases and solutions, except for the input and output data for standard technologies. It is therefore important and necessary to revise the methodology and to establish guidelines on how such simulation systems could be applied for different purposes.

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7

Big Game Waste Production: Sanitary and Ecological Implications

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1. Introduction

Big game hunting has been anciently practiced by the humanity as an essential event to survival (Nitecki et al. 1988), and although nowadays man continues to hunt meat, big hunting has expanded to sport. Species included as big game vary with geographical areas and the general range includes mainly medium to large size hoofed ungulates and predators. Such diverse group of species plays essential roles in the ecological dynamics of natural or semi-natural systems (e.g. Putman 1986). Also big game species can often cause conflict with human interests, for example abundant ungulate species damaging to agriculture and conservation habitats or transmitting disease to livestock (e.g. Ammer 1996, Ferroglio et al. 2011).

Big game waste consists of solids generated mainly after hunting activity. This waste comprises the whole body of the animals or parts of them, such as the viscera and heads (when are not used as trophies). Much of this ungulate biomass becomes solid waste after hunting events following the removal of the internal and external offal (head, feet, all intestines as well as all internal organs). These materials play an important role in ecosystems, maintaining complex faunal communities extending from invertebrates to large carnivores (DeVault et al. 2003). Therefore a premise is that big game waste must be properly managed to protect the continuation of natural ecological processes (for example just leaving it *in situ* may be an option), which poses great conservation value. These remains may also be managed to reduce their effect on animal/human health and the environment under certain circumstances, and to a lesser extent, for aesthetics purposes or to recover resources from them. Although there is increasing interest on the ecological role of these materials, the hunting generation and impacts of big game waste has received little attention compared with management of other solid waste. In some contexts this waste may potentially be hazardous for the animal community (including species of high conservation interest), the livestock or the general public. This situation specially arises as a consequence of human interventions in habitats and the natural regulation of wildlife populations (Gortazar et al. 2006). Changes in population density and/or wild host behavior through solid waste may help new pathogens; new hosts or new hazards emerge, favouring disease spread and maintenance. Here we compile existing information on big game waste generation, ecological value, problems and management options under current regulations, remarking the sanitary and environment conservation dilemmas when managing this waste.

2. Big game waste production

Big game is a significant economic resource through the production of recreational hunting and game meat. Among big game, generation of waste from wild ungulates is relevant because ungulate species are abundant and widespread. For instance, there are some 20 species within Europe (Cervidae, Bovidae, Ovidae and Suidae) adding up to 15 million and representing a standing biomass of more than 0.75 billion kg (Apolonio et al. 2010a). Ungulate post-hunting generation may reach considerable figures but this abundance may strongly vary at short scales. More than 5.2 million animals are harvested each year in Europe (Apolonio et al. 2010a), which resembles a potential of more than 0.1 billion kg of solid waste. The distribution of big game population densities and standing biomass are all strongly influenced by natural factors (Ogutu & Owen-Smith 2003, Acevedo et al. 2011), as well as human's (Gortazar et al. 2006). Ungulate populations may build biomasses exceeding 1500 kg per km^2 in Europe (most due to Cervidae), with high spatial variability even at small scales. This indicates the strong influence of human management on ungulates distribution and abundance, and subsequent standing biomass and post-hunting waste generation. As indicative, North America may harbour approximately 80 million of deer (the main wild ungulate group, including several species, Crête & Manseau 1996) and combining all species biomass may range from 28 to 900 kg per km^2. Africa has ungulate communities of unique diversity with high spatial variability (McNaughton & Georgiadis 1986), and the biomass density in different National Parks may vary across from 100 to 20,000 kg per km^2 (e.g. Coe et al. 1976, Fritz & Duncan 1994). Large ungulate biomasses are also common in tropical ecosystems, although supporting lower quantity than other habitats because most of the primary production occurs in the canopy, well out of the reach of terrestrial herbivores (Bodmer 1989). For instance, wild ungulate biomass ranged from 1900 kg per km^2 to 3290 kg per km^2 in study sites from India (Khan et al. 1996).

A high proportion of ungulate biomass is annually converted in solid waste as a consequence of hunting activities. Nonetheless this proportion varies as a function of the prevalent big game extraction planning. In a context of increasing waste production, mainly due to the population growth of big game (Apolonio et al. 2010b), marked temporal variations between years may occur because hunting exploitation usually is reactive to game population changes, there is a strong effect of stochastic factors on populations (i.e., climatic conditions as hard winters in the Northern areas or droughts in other zones) and hunting actions conveys a degree of randomness (Milner et al. 2006). We estimate that 20-25% of the total population of deer (fallow *Dama dama*, roe *Capreolus capreolus* and red deer *Cervus elaphus*), the most abundant and widespread ungulate species in Europe, are annually shot in average. In the case of the wild boar (*Sus scrofa*), above 30% of the total population may be hunted on a yearly basis in this continent. These hunter kills (and subsequent waste) are usually aggregated in both time and space, as hunting takes place in a tightly circumscribed area over a narrow period of time. When the hunting systems yield the kill of multiple individuals just in a journey, becomes what is called gut piles. This regime of solid waste production has an impact on its posterior use (see below). The availability of ungulate offal piles can be high in some regions. For example, the 10-year mean (1992–2001) of 676,739 white-tailed deer (*Odocoileus virginianus*) annually harvested by rifle hunters in Wisconsin would have produced an average density of about 5 offal piles per km^2 for the area of the entire state (Dhuey 2004), and the harvest of elk (*Cervus elaphus nelsoni*) in other area (Bailey

1999) results in an approximately 70 kg gut pile left at the kill location, which represents 2.5 gut piles per km^2, a 5-fold increase since the 50ies.

Much of this ungulate biomass becomes solid waste after hunting events following the removal of the internal and external offal (head, feet, all intestines as well as all internal organs). In practice, there may be slight differences in the final presentation of the dressed carcass due to local cultural practices, type of trophy and final carcass use. Trophy hunters hunt majorly for the trophy (generally the antlers or horns) and in this case the whole body usually remains as waste (although meat may be consumed by the local "natives" or commercialized by the hunting event organizer or commercial). It can be generated also as a consequence of sanitary confiscations after inspection, ranging from the whole body to specific parts and due to several causes: traumatisms (such as dog bites, bullet-caused massive damages to meat), infectious hazards, unpleasant aspect (i.e. caquectic carcasses) or putrefaction (associated to environment high temperatures, excessive time lapse from the shooting to evisceration, digestive content contamination of the meat, etc.).

The quantity of remains is also variable among taxonomic groups because each presents particular foraging digestive morphology and size. Ruminants, the more numerically important ungulate group among big game, have a large digestive system which conveys a high production of hunting waste: for most of the ruminants the offal weight ranges between 40 and 50% of the total live body weight (Van Zyl & Ferreira 2004). The destination given to big game remains, while following legislative imperatives, may vary in part due to differences in species present, their relative abundance, cultural particularities, conflicts experienced between wild ungulates populations and other land-use interests (i.e. sanitary risks), and whether management is primarily directed towards control, conservation or exploitation (by hunting). Box 1 resumes the solid big game waste production in Ciudad Real, a province of Castilla-La Mancha Region (Spain), a typical big game production area for recreational purposes, attending to temporal, spatial, hunting management and social aspects. Detailed updated figures for national hunting bags for big game (which is mainly due to ungulates) and subsequent production of solid waste can be seen in Apolonio et al. (2010b), but see also Milner et al. (2006).

Box 1. The example of solid big game waste production in South Central Spain

Big game capture volume in Spain (which includes red deer, roe deer, Iberian wild goat *Capra pyrenaica*, fallow deer, Pyrenean chamois *Rupicapra pyrenaica*, Barbary sheep *Ammotragus lervia*, mouflon *Ovis aries*, wild boar and the Iberian wolf *Canis lupus signatus*) has increased during last decades. A conservative estimation of the total captures is over 300,000 per year (Forestal Annuary 2007), the higher figures belonging to wild boar (over 160,000) and red deer (over 100,000). This represents approximately a total of 950,000 kg and an estimated value of over 29,000,000 Euros. Hunting activity in Castilla-La Mancha Region has a great importance, by generating business. Ciudad Real province (19,813 km^2) is a rich big hunting area, which is predominantly red deer and wild boar. To ensure the sustainable use of game species, each estate has its technical plan of hunting, and a compulsory inspection of animal carcasses and remains is done by authorised veterinarians after hunting events. The Mediterranean woodlands and scrublands predominates in the north, west and south borders of the province, and are constituted by largely independently managed private or public hunting estates. The densities of big game populations are highly variable owing to game management practices, but densities often are above the natural carrying capacity (Acevedo et al. 2008), which associates with high disease prevalences (e.g. Vicente et al. 2006, Gortazar 2008, see Box 2). This is also an area of conservation value for species such as the Iberian lynx (*Lynx pardinus*), wolves, the Iberian imperial eagle (*Aquila adalberti*) and the cinereous vulture (*Aegypius monachus*), a specialized scavenger. Vulture species distribution overlaps with the Mediterranean habitats where big game is the prevalent activity. We show the figures of hunting extraction and subsequent generation of big game solid waste. Data presented here come from official statistics (veterinarian inspection) and own elaboration (period 1998-2007). Over 95% of hunting events (and average of per regular hunting season, from October to February) correspond hunting systems with multiple captures (predominantly hunting drives), and correspond to an average number of reserves (public or privates) of 331 per year (range 315-345 per season). During the study period, up to a maximum of 26,014 red deer and 10,126 wild boar per hunting season where shot. This represents a production of 2.3 (approximately 60 kg) and 0.7 (approximately 10 kg) individual gut piles per km^2 and year, for red deer and wild boar, respectively (3 per km^2 and 70 kg both together). The generation of deer gut remains may reach up to 15 (approximately 400 kg) per km^2 and hunting season in some high density estates. Overall, about 1000 mouflons, fallow deer, Barbary sheep and roe deer are also shot per yearly regular season. Big game waste is produced very aggregately in time and space in this area. About 30% of hunting events are concentrated just in two fortnights (in the middle-beginning and the middle-end of the season) in a given season. The average number of hunting event organized by estate is 2.1 per year (ranging from 1 to 17), which is mainly a function of the size of the estate. Figures 1a and 1b show the capture effort (average number of shot animals per hunting and year) for red deer and wild boar at the different Municipalities, respectively. Although red deer is hunted in 57% of the province area, and wild boar in 73%, the production of game waste per hunting event is much aggregated, firstly by Municipalities, resembling the natural conditions for big hunting, but also the intensity of big game management and subsequent densities. Also the data reveals a highly aggregation at the Estate level, since practices such a fencing makes management and densities very variable even at local scale for close Estates. The mean number of red deer shot per hunting event and year per Estate is 14.23 ±14.83 (ranging between 0-65, over 48% of estate shot an average of over 11 red deer per hunting event and year) and for wild boar 18.36 ± 21.99 (ranging between 0-111, 50% of estates shot an average of over 11 wild boar per hunting event and year). These figures are also indicative of the large volume of big game waste generated per hunting event. It is compulsory compiling all the remains at the inspection point (usually close to the hunter meeting site), which determines large gut piles (see Figure 2), which should thereafter be managed according to normative. The mapping of the production of big game solid waste may help optimizing the logistic of treatment programs (such the collecting of the remains) or the design of a net of feeding points for vultures.

3. Ecological value of big game waste

Big game carcasses greatly contribute to the total available carrion that is consumed by scavengers and decomposers in many ecosystems and areas (e.g. Magoun 1976, Hewson 1984, Wallace & Temple 1987, Selva et al. 2003, 2005, Wilmers et al. 2003a, b). Since extensive cattle farming is in serious decline mainly in many areas of developed countries (e.g. Bernues et al. 2005), wild ungulates may be able to or have already occupied this vacuum. They generate naturally a significant amount of carrion (e. g. Blázquez et al. 2009, Blázquez & Sánchez-Zapata 2010) which originates from the kill remains of large predators

(a)

(b)

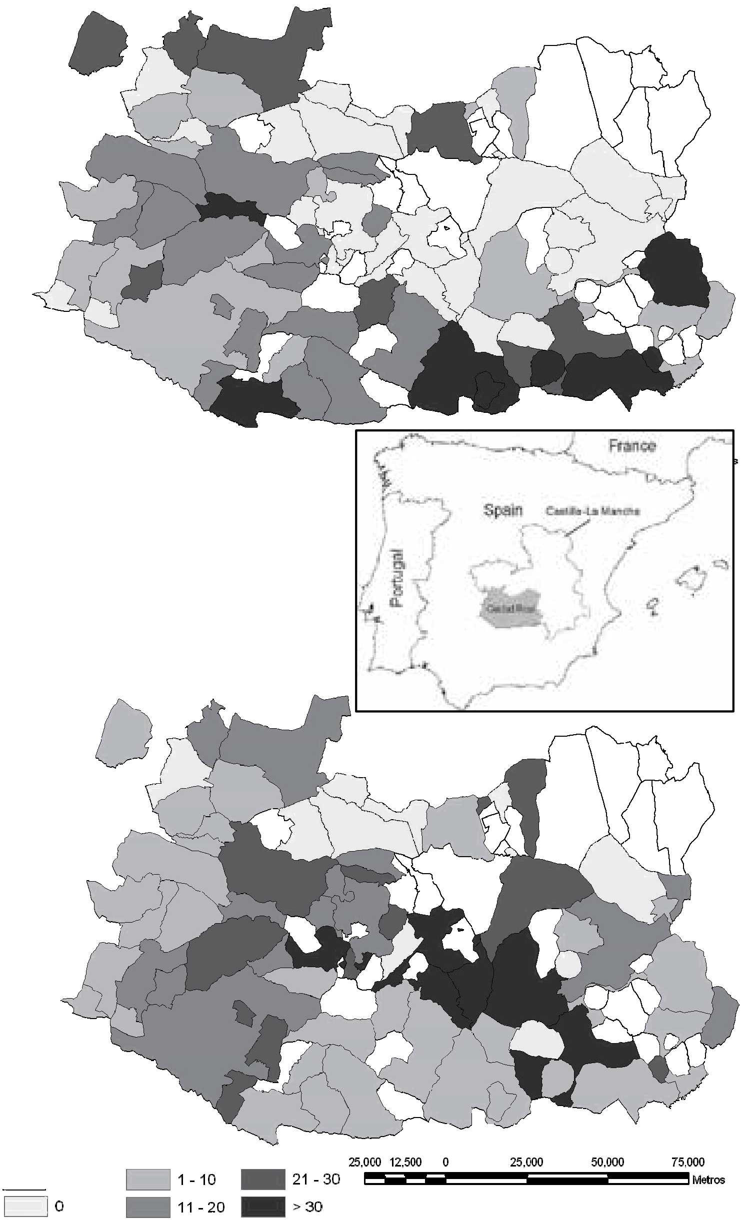

Fig. 1. The capture effort (average yearly value of shot animals, per hunting event and hunting estate), which equates to the individual big game offal generated, for red deer (a) and wild boar (b) at municipality level, respectively (red deer hunted in the 57 % of the province area, wild boar in 73 %) in the province of Ciudad Real (Castilla-La Mancha Region, South Central Spain, location is depicted in the inset). No data for municipalities in white

(although predator strategies either rapidly consume most of them or hide prey remains make them not to be available for scavengers, DeVault et al. 2003) and natural deaths (malnourish or diseased animals). In Bialowieza primeral forest (Poland), for example, a wolf pack kills an ungulate every two days and annually wolves kill on average 72 red deer, 16 roe deer, and 31 wild boar over a 100 km^2 area (Jedrzejewsky et al. 2002). These processes maintain complex faunal communities extending from decomposers and invertebrates to large carnivores (DeVault et al. 2003), and improve soil nutrient quality (Towne et al. 2000). Human hunters probably provide a larger food resource (hunting waste) to scavengers in many areas. This supply occurs in landscapes and periods of time with limited food availability for scavengers, reason for which is very valuable. In many cases the amount of carrion in the form of big game waste is much more abundant than that of natural origin because large predators are not longer present in many areas and/or human promotes large abundances of ungulates for hunting purposes. Solid waste originated from wild animals represents an important part of the diets of avian scavengers in areas devoted to big hunting (Blazquez & Sanchez-Zapata 2009). For example, in South Central Spain (Vicente et al. 2006), the country where inhabits the majority of European vultures, hunting remains are key to the maintenance of this endangered and rich avian scavenging community.

There exists a certain degree of competence between vertebrate scavengers and arthropods and decomposing microbial. Microorganisms are generally the first in colonizing the carrion or waste, using enzymes and toxins to degrade the tissues, in some cases monopolizing the use of this resource (Janzen 1977, Braack 1987), especially in hot weather areas. Nonetheless microbial hardly ever colonize all the biomass, although they have the potential to transform the carrion or waste into a unpleasant even toxic mass that is not further used by vertebrate scavengers. At the same time, substances derived from the decomposing process will signal vertebrate scavengers (DeVault & Rhodes 2002a). The range of scavenging species primarily may vary as a consequence of the availability of biomass found in particular regions. The scavenging community includes obligate (vultures) and facultative scavengers (avian or mammal), each of the species either uses different parts of the carcass, or locates different types of carcass or has a distinct geographical range. Whatever the origin, ungulate carrion represents the principal source of food for obligate scavengers. In spite that vultures tend to concomitantly exploit the resources, there exists certain degree of specialization among them. Although the available food supply is utilized very efficiently by the obligate avian scavengers, the status of many vulture populations is of acute conservation concern as several show marked and rapid decline (e. g. Donazar et al. 2002). Also most carnivorous and omnivorous vertebrates can be considered to be facultative scavengers (DeVault et al. 2003), although the tendency to consume ungulate carrion varies widely from frequent (e.g. Gasaway et al. 1991, Green et al. 1997) to limited consumption (Delibes 1980, O´Sullivan et al. 1992, DeVault & Krochmal 2002b). In general, where abundant specialized scavengers are present, facultative scavengers may proportionally account for a smaller proportion of the scavenging activity than they would do in the absence of vultures. Nonetheless, since human activities have an influence on endangered and unmanaged wildlife, as the loss of certain habitats or food resources, different species has been lead to exploit ungulate carrion as alternative resource (e.g. Iberian lynx feeding on ungulate carrion, Perez et al. 2001). Facultative scavengers may locally specialize on the exploitation of the hunting solid waste exploitation due to the large amount produce that in not fully consumed by vultures (see below). Different studies have revealed active guilds of vertebrate scavengers in wild ungulate carcases all over the world and some of them have quantified this use and the factors involved in the consumption (e. g. Selva et al. 2003, Blázquez & Sanchez-Zapata 2009). For example, the effect of habitat on the quantitative consumption of the carcasses also may differ between habitats and prevalent scavenging communities. Very few studies have attended the interactions (direct or indirect) occurring between different scavengers, for example between nocturnal (most of which are mammals) and diurnal species, so as the competence and dominance relationships occurring among them, and how they specialize in exploiting the resource.

The unpredictable availability of natural carrion has probably inhibited the strict evolving towards strict scavenging specialization in vertebrates (Houston 1979). The carrion provided by natural enemies (predators and diseases) arrives consistently over the course of year (Selva 2004), but the generation and mode of disposal of big game waste differ from the natural regime of carrion pulses along time and space. Because of the high temporal and spatial overlap of carrion at hunter kills, especially in the form of large gut piles, scavengers from the local area surrounding the gut piles may become super-saturated with resource. How beneficial result to scavengers the temporal resource patterns of hunter kills depends on a trade off between an ability to assimilate and/or cache large amounts of resource quickly and/or tracking that resource over time (Wilmers et al. 2003b). Such super-saturation reduces competition and allows far ranging species to gather in high numbers, not always with beneficial results. Even facultative scavenging individuals in the proximity get used to exploit this resource at predictable sites. Scavenger feeding stations, which are designed to favour vulture supply of resources, provide carrion regularly in time and space, and therefore are predictive, with consequences that may not meet always the original conservational objectives. From 2002, a number of dispositions to the EU regulations (discussed below) enabled conservation managers the creation of vulture feeding stations aimed at satisfying the food requirements of vultures, but these conservation measures may seriously modify habitat quality and have indirect detrimental effects on avian scavenger populations and communities (e.g. Donazar et al. 2010).

4. Hazards potentially present in big game waste

Big game carcass and waste may bear infectious, toxinfectious or toxicological hazards primarily for scavengers. Often, once it has been confirmed a health problem in a given population, community or environment, studies focus on the role of scavenging on wildlife carrion/solid waste to favour the spread and perpetuation of such problem, in many cases usually confirming the initial suspects. For example, we can mention the case of scavenging on possums (*Trichosurus vulpecula*) by ferrets (*Mustela furo*) and the bovine tuberculosis problem in New Zealand (Ragg et al. 2000, Lugton et al. 1997). Wildlife disease surveillance and monitoring is a necessary first step to identify risks and develop adequate management schemes of big game waste. The use and management of such waste must be based on scientific knowledge in order policy makers develop equilibrate regulations and decisions, balancing sanity and conservation priorities, while avoiding alarmism on the risks for disease transmission coming from big game solid waste disposal.

Sanitary risks posed by big game are dependant upon the prevalence, incidence, and magnitude of disease agent carriage in the animal, the degree of interaction between the animals and the environment, and animal behaviour and ecology (Morris et al. 1994). Usually the most abundant big game species in a particular region are of the greatest concern as the risk of exposure by these animals remains may be the highest. Under certain circumstances big game waste disposal may contribute to the establishment and subsequent maintenance of pathogens and disease in scavengers, the rest of the animal community and the environment. In order this to occur, pathogens must be present and viable in accessible-to-scavenger waste. The scavenging species in turn must be susceptible to infection and be able to, somehow, transmit the pathogen to favour its persistence. A particular scavenger, although not being the most affected species in terms of prevalence or disease severity, may play a key factor in maintaining the problem because of its epidemiological role as reservoir of disease.

To briefly describe infection dynamics, an infected animal population can be classed as either a maintenance or spillover host, depending on the dynamics of the infection. In a maintenance (true reservoir) host, infection can persist by intraspecies transmission alone, and may also be the source of infection for other species. In a spillover host, infection will not persist indefinitely unless there is re-infection from another species or the environment. The presence of a disease and reservoir may involve the maintenance of disease may pose management implications in relation to big game waste. Fenton and Pedersen (2005) proposed a conceptual framework based on the pathogen's between- and within-species transmission rates to describe possible configurations of a multihost-pathogen community that may lead to disease emergence. Spill over and apparent multihost situations are those where, without between-species transmission (for example inter-specific scavenging), the disease would not persist in the target host. In true multihost situations the pathogen can independently persist in either host population in the absence of the other. One example of multihost situation is bovine tuberculosis (bTB), caused by *Mycobacterium bovis*.

Bovine tuberculosis is mainly a disease of domestic cattle and goats, but can affect many other domestic and wild species, as well as humans. Also some species of conservation interest have resulted affected, such as the Iberian lynx in their last two strongholds in southern Spain (e.g. Perez et al. 2001). Consumption of infected prey or infected carcass or game waste is also a suspected as the way of transmission (Vicente et al. 2006). The existence of wildlife bTB reservoirs is the main limiting factor for controlling this disease in livestock. Major problems with wildlife bTB occur in areas with a high density of susceptible host species (de Lisle et al. 2001), such as the possum in New Zealand, the buffalo (*Syncerus caffer*) in South Africa, and the badger (*Meles meles*) in the UK and Ireland, white-tailed deer in North America, and transmission may get magnified when scavengers of infected gut piles become infected (Bruning-Fann et al. 2001, Gortazar et al. 2001, 2008, Renwick et al. 2007). In contrast, some pathogens do exclusively infect a single host species. These pathogens are frequently specialized; highly coevolved parasites with limited effect on the primary host's population (Crawley 1992), or the possible secondary hosts are just unknown. Then, big game consumption may become a risk for the transmission of these pathogens when it involves cannibalism (e. g. wild boar, although carrion consumption by ruminant ungulates has been extraordinary detected). These pathogens are generally, in the absence of environmental changes, considered less relevant from the wildlife management and conservation and domestic animal perspective. Emerging infectious diseases include those where the pathogen will become self-sustaining in the new host once the initial (environment, host- or pathogen-related) barrier to infection has been crossed, for example, by big game waste ingestion. Wild animals are the most likely source of new emerging infectious diseases that put at risk the health of human beings and livestock.

Human impacts on natural processes favour that some species contributes to maintenance of diseases, for which game waste may play a role. In Europe, as in many other parts of the world, the changes occurring across the last 40 years have had a pronounced effect on the environment, creating a dynamic situation where pathogens or new hosts emerge o re-emerge. In particular, there have important been changes in big game population density and/or host behaviour (management favouring aggregation, Acevedo et al. 2007), which affect disease prevalence and, in some cases, may allow disease agents to boost their virulence and widen their host range (Ferroglio et al. 2010). Big game becomes often reservoir of disease as a consequence of overabundance. According to Caughley (1981), overabundance ("overpopulation") of a given wildlife species occurs, among other premises, when it causes dysfunctions in the ecosystem. This occurs also in form of disease spread and maintenance in the population that otherwise would not occur. In fact, the most obvious cases of relationships between overabundance and diseases occur among wild ungulates. The European wild boar is a good example. This species is increasing its range, reaching levels previously unrecorded (Geisser & Reyer 2004). This has contributed to the spread of many diseases, including classical swine fever, Aujeszky´s disease, Porcine Circovirus type 2, and bTB (see Box 2 and Figure 2), among others. It has also been shown that the increased density and spatial aggregation of wild boar in fenced hunting estates increases the risk of getting in contact with multiple disease agents (Ruiz-Fons et al. 2006). These situations are good examples of how overabundance affects animal health through the consumption of gut piles from fall ungulates in overabundance situations. In many cases, big game gut piles are left in the own hunting place or at meeting points, and remain available not only for obligate scavenger species but also can be used by the facultative scavenger, such as terrestrial carnivorous and omnivorous mammals. Under such circumstances, big game waste consumption by facultative scavengers (among which many are mammals) favours the feed back on the transmission chain and the maintenance of diseases (Bruning-Fann et al. 2001, Renwick et al. 2007, Jenelle et al. 2009, see Box 2 for the scavenging activity of wild boar). Therefore care should be taken with ungulate waste, especially in overabundance situations, since susceptible facultative scavengers may access to waste, which includes endangered species that scavenge to some extent (Perez et al. 2001). This situation secondary increases the risk of disease transmission from wildlife to the domestic flock and humans, which can also undermine conservation efforts if wildlife is seen as the source of a disease affecting livestock or human health (Brook & McLachlan 2006). Obligate scavengers effectively remove infectious tissue, thus decreasing the load of pathogens from the environment. It is therefore desirable that legislation be applied in a way that would allow for the selective access of vultures to the abandoned carcasses and gut piles that appear during the hunting season (see below). Although sanitary authorities should consider the removal of infected hunted animals and viscera to limit potential pathogen contamination where facultative scavengers can access, the conservation of obligate scavengers and other birds requires of selective disposal that guarantees their food supply (see bellow). In view of the potential risks of big game waste for the food chain, diseases that benefit from wildlife overabundance are of special concern, affecting public health, livestock health, and the conservation of endangered species.

A large number of infectious agents have been found in big game species. For example, foodborne pathogens may be present in the gut and faeces of wild ungulates without causing outward signs of illness or disease, making it difficult, if not impossible, to determine by visual inspection if an animal is carrying a specific pathogen. Following we briefly review some of the most relevant ungulate diseases that may be transmitted via big game waste. Along with the nematodes of the genus *Trichinella*, the cestode and *Echinococcus* *granulosus* are some of the parasitic helminths of greatest zoonotic concern. *E. granulosus* has a domestic dog – sheep (and other livestock) cycle, but also a sylvatic wolf – wild ruminant (and other herbivores) one. These cycles may get linked through dogs consuming carcass remains of hunted game and wolves consuming livestock as carrion or as prey (e.g. Sobrino et al. 2006). Although is illegal, feeding wildlife or domestic pigs with wild boar offal still occurs, which increases risk for human trichinellosis outbreaks. Furthermore, where absent, programs are needed that emphasize the necessity of ensuring testing for *Trichinella* spp. infection in all wild boar intended for human consumption and promoting education of humans regarding thorough cooking of meat to guarantee food safety. In general, any infectious disease that is prevalent in a given area that affect also other species than big game, either zoonotic or not, have the potential to be present in the carcass, which justifies the need of rigorous inspection by well-qualified people. Obviously, any animal showing disease symptoms or lesions suggestive of such diseases (for example during an outbreak of disease) should be out from the food chain, and in some cases when the destination are avian scavengers and there are suspects that they may be affected or may act as vectors of disease (e. g. Bullock 1956 concerning vultures as disseminators of anthrax).

Aujeszky´s disease (AD), a viral disease of swine and wild boar that can affect most mammals except man and primates, has been proposed as a risk for carnivore. AD causes fatal infection in non suid species including carnivores such as the Florida panther (*Felis concolor corii*) or the European brown bear (*Ursus arctos*) (Glass et al. 1994, Zanin et al. 1997). In Europe, endangered carnivores such as the Iberian lynx, the brown bear, or the Iberian wolf may include wild boars among their prey species (Clevenger et al. 1992), and thus may eventually be at risk due to AD. Cannibalism after wild boar AD outbreaks has been related to disease transmission (Gortazar et al. 2002). Rabies is the most classical wildlife related zoonosis. Data suggest that oral transmission of rabies virus among scavenger species may be a common occurrence (Schaefer 1983). In Europe, the red fox (*Vulpes vulpes*) is the main reservoir of this viral disease. Rabid foxes can transmit the virus to wild and domestic mammals and humans, or infect pets or livestock that can in turn infect humans. Hepatitis E virus also circulates actively among red deer and wild boar (Boadella et al. 2010).

Apart from bTB (commented above) other bacterial disease such as brucellosis are of concern when disposing big game waste. Brucellosis, is endemic in elk and bison using winter feed grounds of western USA presumably because of increased animal density, duration of attendance, and subsequent contact with aborted foetuses (foetuses, placentas, and fluids). Similar hazard could occur from game remains. The bacteria *Brucella* exists in the reproductive tissues, yet elk, bison and deer are hunted every year, and hunters leave gut piles all over the landscape. It have been seen that several eight species of scavengers consumed foetuses, and therefore protection of scavengers on and adjacent to feed grounds would likely reduce intraspecific transmission risk of brucellosis (Maichak et al. 2009). *Escherichia coli* O157 has become an important cause of illness attributed to food born contamination. Ruminant animals are among the most common reservoir species for this pathogen. In studies of free-ranging deer, the faecal prevalence of *E.coli* O157:H7 was estimated to range from zero to less than 3% (e. g. Fisher et al. 2001, Dunn et al. 2004, Branham et al. 2005). Among wild ungulates, apparently low prevalence of *Salmonella* faecal shedding occurs (Renter et al. 2001), although *Salmonella* were detected in 8% of rumen samples from white tailed deer (Renter et al. 2006). Finally, it should be noted that hunters and contaminated equipment may also be vehicles by which pathogens are transferred from contaminated locations to the growing field.

Chronic wasting disease (CWD) is a transmissible spongiform encephalopathy (TSE) of North American cervids. As suggested for CWD in high-risk regions from North America (Jenelle et al. 2009), recent laws on TB endemic areas have considered the removal of hunted animals (including viscera) to limit potential TB deposition near a kill site that mammals, and particularly wild boar, can access. Carcasses infected with CWD are an important source of infectious prions to susceptible cervids and may expose vertebrate scavengers (Miller et al. 2004, 2006). Agents that cause TSEs can remain viable in the environment for many years (Seidel et al. 2007). During decomposition, ungulate carcasses release nutrients into surrounding soils, stimulate subsequent biomass production that attracts herbivores, and serve as a potential source of infectious material (Towne 2000, Miller et al. 2004). It has not been demonstrated transmission of CWD to other species, but it has anecdotally been reported that deer may consume animal tissues, including flesh and bone of dead conspecifics (Cook et al. 2004). Avian scavengers also are consumers of ungulate carrion (Wilmers et al. 2003a, Cook et al. 2004), but birds are not susceptible to mammalian TSEs (Wopfner et al. 1999). It may be possible, however, for avian and mammalian scavengers to consume TSE-infected materials and spread prions, or other infectious agents, through deposition of feces in the environment (Houston & Cooper 1975, European Commission 2002) or by transport of infectious carrion during food-caching or young-provisioning. Interestingly Jenelle (2009) found that the daily rate of deer contacts with gut piles was greater than with whole carcasses, suggesting a higher daily risk of potential exposure for susceptible deer. However, the total risk of exposure will also depend on the persistence and abundance of gut piles *versus* carcasses in the environment.

The bovine spongiform encephalopathy (BSE, commonly known as mad-cow disease) is a TSE of cattle. The disease may be most easily transmitted to human beings by eating food contaminated with the brain or spinal cord or digestive tract of infected carcasses. In the EU, farmed and free ranging deer were almost certainly exposed to BSE infected in proprietary feeds prior to legislation banning its inclusion. The European red deer are susceptible to intra-cerebral challenge with BSE positive cattle brain pool material and the clinical signs are indistinguishable to those reported in deer with CWD, and one of six red deer orally dosed with BSE developed clinical disease (Martin et al. 2005, 2009) although no BSE wild deer has been diagnosed. By 2002, BSE crisis led to prohibition on abandoning the carcasses of extensively grazed animals in the wild in the EU, and subsequently, led to a scarcity of this type of carrion. EU regulations distinguish between carcasses that are safe and those that may be a potential source of BSE transmission /EU regulations: EU999/2001, EU1774/2002, EU32272003, EU830/2005), despite the lack of any evidence to suggest that dead animals left in the wild suppose any risk of BSE transmission (CMIEET 2001, Crozet & Lehmann 2007). Over 90% of European vultures live in Mediterranean areas (e.g. Donazar et al. 2009a), most on them in the Iberian Peninsula, and these species are of conservation interest. As a result, avian scavengers have been forced to congregate at artificial feeding points (fenced ‘vulture restaurants’) supplied with carcasses from stabled animals under intensive exploitation that are thought not to represent a risk BSE transmission. In theory this is a good system for avoiding public health problems caused by the ingestion of carcasses of animals that can be vectors for the spread of disease, such as generalist carnivores. This feeding strategy has led secondly to the obligate concentration of birds at a small number of feeding points supplied with carrion originating from stabled animals, which are though not to be at risk in the transmission of BSE. It can be counterproductive in certain aspects, such as avian behaviour and dynamics, and impact in local habitat quality. Additionally, as changes in commented below, this management conveys infectious and toxicological risks to scavengers (Lemus et al. 2008, Blanco et al. 2009). Because of the dilemma between the application of sanitary and conservation strategies, managers and policy makers must solve a problem of lack of food in one of the most threatened wildlife groups and at the same time Being compatible with food security policies. Wild ungulate waste may mitigate these effects; although this does not mean encouraging at the same time their densities, because overabundance (and its consequences) is already a serious concern. A sufficient amount of natural of carrion originating from big game is already generated in large areas (in Mediterranean areas from Spain may reach about 40% of obligate scavenger diet, and nearly 100% during hunting season), and the question is how ensuring the supply of big game waste management with not sanitary risks for other groups of animals and humans. In fact, vultures are sanitary filters because they clean up the environment by eating carrion.

An additional problem is that wild ungulate remains left by hunters may result in lead-shot poisoning with harmful effects on individuals and populations (e. g. Hernández & Margalida 2009). There are possible measures that have not yet been developed which should be implemented by the regional authorities. It appears advisable for the networks of vulture restaurants to be well spaced out and include a large number of feeding points, to promote them in big game exploitations, always implemented and assessed on the basis of scientific evidence. These feeding points should be adequately managed in order to avoid excessive spatial and temporal concentration of avian scavengers and big game remains, which is produced as large pulses in time and space in many hunting areas (see Box 1).

Interestingly, and contrary to extensive livestock, it has been found that the presence of carcasses originating from intensive livestock (especially pigs and chickens) carry a high number of pathogens and pharmaceutical products that can be transmitted via carrion consumption to avian scavengers at feeding sites (Lemus et al. 2008, Blanco et al 2009). If sanitary programs are applied in the same way in some game farms, attention should be paid to the remains originated from these animals. Carcasses originating from extensive livestock and natural big hunting left in the wild should be prioritised by management and conservation programmes targeting avian scavengers as a strategy that does not put the health of birds or the environment in general at risk. Nonetheless this situation may present particularities in wild ungulates. This is so because overabundance situation conveys high prevalence of disease in populations that are not under sanitary programs as extensive livestock is, this remarks the importance of the veterinary inspection of the carcasses and hunting remains. Similar to intensive livestock, pharmaceutical products used in farmed big game might have negative consequences for the health of avian scavengers and population dynamics (Oaks et al. 2004 concerning the collapse in avian scavenger populations in South-east Asia, Lemus et al. 2008, Blanco et al. 2009 concerning non-steroidal anti-inflammatory and anti-parasitic agents and Iberian vultures that feed predominantly at feeding points supplied with animal remains). As a consequence, the pathogenic and bacterial flora may develop resistance to these products. It has been found that the presence in the blood of the residues of antibiotics coming form intensive livestock carrion is associated with the infection by opportunistic pathogens of the nestlings and damage to internal organs in vultures from Spain (Lemus et al. 2008, Blanco et al. 2009). Scavengers may also encounter metal toxics in big game carrion, for example metals derived from bullets. Metals (Pb, Hg, Cd) belong to the most toxic substances present in the natural environment. Lead, as a metal characterized by a very high accumulation rate in the environment, presents a particular strong threat of disturbing the chemical equilibrium in the biosphere. A serious problem is presented by bullet-derived lead contamination of large game carcasses. Monitoring of lead contents in tissues of large game frequently shows high lead levels, exceeding the admissible contents several tens or—on occasion—several hundred-fold (e.g. Dobrowolska & Melosik 2008). Careless removal of tissues from around the bullet pathway in the animal body results in elevated lead doses being ingested by humans. Lead-shot poisoning may harms scavengers, as has been demonstrated for avian scavenger (Hernandez & Margalida 2009). For example, bullet fragments in rifle-killed deer (*Odocoileus* spp.) carrion and offal have been implicated as agents of lead intoxication and death in bald eagles (*Haliaeetus leucocephalus*), golden eagles (*Aquila chrysaetos*), California condors (*Gymnogyps californianus*), and other avian scavengers (Craighead & Bedrosian 2006, Hunt et al. 2006). For example, 94% of deer samples killed with lead-based bullets contained fragments, and 90% of 20 offal piles showed fragments: 5 with 0–9 fragments, 5 with 10–100, 5 with 100–199, and 5 showing 200 fragments (Hunt et al. 2006). In contrast the authors counted a total of only 6 fragments in 4 whole deer killed with copper expanding bullets. These findings suggest a high potential for scavenger exposure to lead. Mammalian carnivores in areas of high hunting density may exhibit the same temporal pattern of lead exposure from ingestion of rifle bullet fragments during the hunting season as avian scavengers, for example grizzlies feasting eating the remains of the hunt in the Greater Yellowstone on gut piles get lead and show blood elevated levels of toxic lead (Rogers et al. 2008). Finally, bullet-derived lead in game food products is also an important source of human contamination. In summary, research is needed on big game remains that are left for scavengers without any previous toxicological control

Box 2. The example of wild ungulate waste consumption by vertebrate scavengers in Mediterranean areas: ecological, conservation and sanitary implications

The carcasses and gut piles of wild ungulates are becoming gradually more available in the Iberian ecosystems, and may constitute an important food source of feeding for the vertebrate scavengers (see Box 1). This is because overabundant ungulate populations (mainly wild boar and red deer) do exist in many areas, where hunting is an important socio-economic activity, and this contributes to generate large amounts of available carrion. Overabundance consequences, due to big game industry, in form of diseases (Gortazar et al. 2006, Acevedo et al. 2007) occur, and subsequently the consumption of infected ungulate carrion may contribute to disease maintenance and spread in the wildlife reservoir, especially if facultative scavengers have access and are directly exposed to potentially infectious ungulate material. The importance given to the management of big game waste when is capable of transmitting diseases depend on whether wildlife has a high probability of substantially affecting regional disease status, and the disease has a strong impact on human health, economy, wildlife management and conservation. One case is bTB in Spain. Carcasses from individuals with chronic bTB are particularly infective owing to the large extension of infected tissues (Martin-Hernando et al. 2007). In South Central Spain bTB averages 45% of prevalence based on macroscopic lesions in wild boar (ranged up to 100% in local populations of wild boar, Vicente et al. 2006), reaching prevalences that has been estimated up to 60% based on culture (Gortazar et al. 2008). The importance of the intraspecific transmission of bTB may be especially relevant in areas with absence of vultures (Gortazar et al. 2008), and the fact that wild boar, the main carriers of the bTB in Spain (Naranjo et al. 2008) are the principal ungulate carrion consumers (see below). Figures for deer in south-central Spain are also high, averaging 15% prevalence (Vicente et al. 2006), and about 30% and 20% in red deer and fallow deer, respectively, in Doñana National Park, South Spain (Gortazar et al. 2008). The isolation of bTB strains from these estates strongly suggests that the *M. tuberculosis* complex is able to survive in these high-density wildlife populations in the absence of livestock hosts (Gortazar et al. 2005). Cannibalism may be one mechanism by which TB is transmitted within wild boar populations (Ragg et al. 2000, Gortazar et al. 2002). Predators in Spain, such as the endangered Iberian lynx, the badger and the fox, have been considered to be possibly infected by TB as a result of their consuming infected prey or carrion (Perez et al. 2001, Millan et al. 2008, Sobrino et al. 2008). As long as bovine bTB exists in ungulate populations (the main reservoirs in many areas), there will be some risk to local wildlife species that feed on bovine bTB-infected carcasses or gut piles.

As a basis through which to determine the ecological, conservational and sanitary relevance of ungulate carrion and waste for the vertebrate scavenging community (specialized and facultative) in European Southwest Mediterranean areas during 2007, we describe the guild of vertebrate scavengers. We compared one site on which obligate scavengers (vultures) were absent during the study period (Doñana National Park) with another area in which they were present and abundant all year round (Central Spain). By using automatic photo-trapping on 47 carcasses (Figure 2), the frequencies of scavenging for different species per carrion and study area are shown in Table 1. As expected, vultures were only detected in Central Spain. We evidenced that, even in the presence of abundant specialized scavengers, wild boar, red fox and facultative scavengers accounted for a relevant proportion of the scavenging activity, to the extent of becoming locally predominant in the absence of vultures. Also, own data on the frequency of scavenging on gut piles generated after big hunting events in Central Spain evidence that remains left in habitats with high vegetation cover are more often scavenged by facultative scavenger (wild boar and foxes) than by vultures. The high occurrence of wild boar and foxes at carrion throughout our study areas is indicative of their ubiquity and abundance, whereas for other species the low and/or solely local occurrence would reflect their more restricted distribution and/or abundance (e. g. Imperial eagle, kite, Egyptian vulture, Egyptian mongoose, etc.). Wild boar are frequently managed in order to promote high densities on hunting Estates in South Central Spain, and also reach considerably high densities in protected areas (Acevedo et al. 2007). The communal feeding by facultative scavengers on ungulate carrions may facilitate the intraspecific and interspecific transmission of diseases, since scavengers are directly exposed to potentially infectious ungulate material. This risk is increased by the sanitary consequences of overabundance caused by intensive management in South Central Spain, which favour the transmission of disease and its persistence in ungulates (Gortazar et al. 2006, Acevedo et al. 2007). The importance of the intraspecific transmission of TB may be especially relevant in certain areas because of the absence of vultures, and the fact that wild boar, the main carriers of the disease in Spain (Naranjo et al. 2008), are the principal ungulate carrion consumers. This example provides support for the influence on the environment on carcass consumption and the direct or indirect competitive relationships between scavengers, and focuses the discussion on ecological, conservational and disease management considerations. The consumption of infected ungulate carrion and waste may subsequently contribute to the spread and persistence of bTB in wildlife with regard to carnivorous and omnivorous species rather than avian scavengers, which effectively remove infectious sources. In this context, controversy has arisen concerning vulture conservation, since current European policies encourage the destruction of domestic animal carcasses, rather than their being left in the open (Donazar et al. 2009b), and this could also apply to the management of big game carcasses and hunting remains. Table 2 describes the guild of vertebrate scavengers present during the monitoring of 18 gut piles in the South Central area, during the 2008-2009 hunting seasons. Gut piles where obtained from hunting were constituted by non-trophy heads, hoofs and thoracic and abdominal viscera (red deer and wild boar).

Visitant species	Total (n=47)	Doñana N. P. (n=10)	Central Spain (n=37)
	%Pres/%Scav[1]	%Pres/%Scav	%Pres/%Scav
Wild boar	80.9/55.3	100/90	80/50
Red fox	85.1/55.3	80/30	90/70*
Griffon vulture	40.4/40.4	0/0	65/65
Monk vulture	36.2/31.9	0/0	55/55
Raven	12.8/6.4	40/20	0/0
Magpie	12.8/6.4	30/20	15/5
Jackdaw	8.5/2.1	30/0	0/0
Kite	14.9/10.6	70/50	0/0
Egyptian mongoose	2.1/2.1	10/10	0/0
Imperial eagle	2.1/2.1	0/0	5/5
Egyptian vulture	2.1/0	0/0	5/0
Eurasian jay	2.1/0	0/0	0/0
Red deer	6.4/0	0/0	0/0
Cattle	8.5/0	40/0	0/0
Horse	4.3/0	20/0	0/0

Table 1. Percentages of presence and scavenging occurrence per carcass (%) of the scavenging community. [1]Percentage of carrions at which the species was detected/Percentage of carrions at which scavenging by the species was detected.

Species detected	Nº of gut piles visited		Nº of gut piles scavenged		Mean group size (± SD)
	open	woodland	open	woodland	
Griffon vulture	9	1	9	1	36.03 ± 22.67
Monk vulture	7	0	7	0	4.15 ± 2.94
Raven	8	2	6	1	4.29 ± 3.51
Magpie	2	0	2	0	6.14 ± 4.09
Azure-winged magpie	5	1	5	1	4.24 ± 3.09
Egyptian vulture	1	0	0	0	1
Imperial eagle	2	0	2	0	1.26 ± 0.45
Golden eagle	1	0	1	0	1
Wild boar	4	4	4	3	1.58 ± 0.88
Red fox	9	5	7	4	1.02 ± 0.12
Common genet	0	1	0	0	1
Dog	3	1	3	1	1.02 ± 0.15
Red deer	2	0	2	0	1

Table 2. Scavenging community and general parameters of activity detected in 18 gut piles from South Central Spain.

5. Normative

Here we briefly expose and update the current normative that applies to big game waste management from a European point of view, from the EU Regulation to some regional normative that is exemplified with particular cases given the current diversity occurring among countries and regions. Regulation (EC) No 1774/2002 introduced the classification of animal by-products into three categories according to the degree of risk involved. Pursuant to that Regulation, only material from animals which have undergone veterinary inspection is to enter the feed chain. In addition, it lays down rules for processing standards which ensure the reduction of risks. The regulation (EC) no 1069/2009 of the European Parliament and of the Council of 21 October 2009 lays down health rules as regards animal by-products and derived products not intended for human consumption and repealing Regulation (EC) No 1774/2002 (Animal by-products Regulation). In order to prevent risks arising from wild animals, bodies or parts of bodies of such animals suspected of being infected with a transmissible disease should be subject to the rules laid down in this Regulation. This inclusion should not imply an obligation to collect and dispose of bodies of wild animals that have died or that are hunted in their natural habitat. If good hunting practices are observed, intestines and other body parts of wild game may be disposed of safely on site. Such practices for the mitigation of risks are well-established in Member States and are in

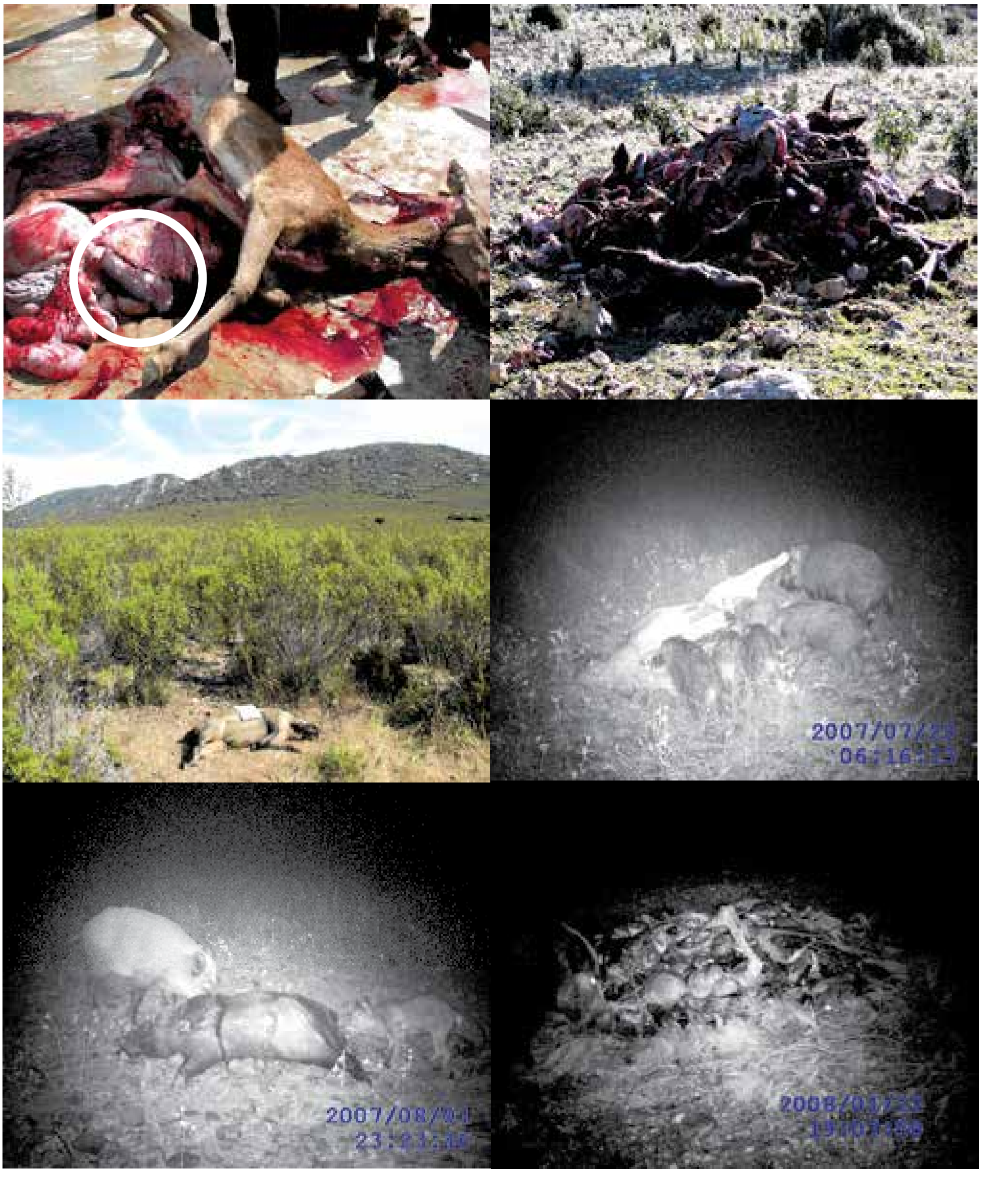

Fig. 2. This figure illustrates big game waste production; handling and gut pile disposal after a hunting event (top right and left) in south central Spain. Research on carcass and waste use by scavengers can be done by means of automatic recording systems after the carcass or gut pile is disposed. Pictures illustrate carrion and gut pile consumption (either deer or wild boar origin) by wild boar in south central Spain. Facultative scavengers may proportionally account for a proportion of the scavenging activity and under certain circumstances; big game waste consumption by them favours the feed back on the transmission chain and the maintenance of diseases, such as bovine tuberculosis. Observe bovine tuberculosis compatible lesions in the liver of the gut pile obtained from the deer (top left).

some cases based on cultural traditions or on national legislation which regulates the activities of hunters. This Regulation shall not apply to the following animal by-products: (a) entire bodies or parts of wild animals, other than wild game, which are not suspected of being infected or affected with a disease communicable to humans or animals, except for aquatic animals landed for commercial purposes; (b) entire bodies or parts of wild game which are not collected after killing, in accordance with good hunting practice, without prejudice to Regulation (EC) No 853/2004; (c) animal by-products from wild game and from wild game meat referred to in Article 1(3)(e) of Regulation (EC) No 853/2004. Community legislation Regulation (EC) No 853/2004 of the European Parliament and of the Council of 29 April 2004 laying down specific hygiene rules for food of animal origin, lays down rules for handling of meat and animal by-products from wild game. Those rules also place the responsibility for the prevention of risks on trained persons such as hunters. In view of the potential risks for the food chain, animal by-products from killed wild game should only be subject to this Regulation in so far as food hygiene legislation applies to the placing on the market of such game and involves operations carried out by game-handling establishments. Following with the example at regional scale, the Castilla-La Mancha regional normative 14607 from may 2008, complete regulation (CE) nº 852/2004, and normative lay down specific regulations on the collection, transport, inspection and sanitary control of game (big sands mall) intended to commercialization for human consumption. Concerning the inspection, before transport of the carcass to an authorized processing plants it is compulsory a first inspection by an authorized veterinarian (private in this cases, in other Spanish region the veterinarian must be official). It lay down that all the shot animals must be presented to the veterinarian in an authorized evisceration place, and there will be auxiliary staff to eviscerate and mark the carcasses. Even for big hunting intended for self-consumption, it is not allowed that the responsibility for the prevention of risks falls on not veterinarian persons, such as hunters, and it laid down the conditions that are required for big game authorized inspection locals (D65/2008 from 6 May 2008). The examination of the animals must include the body and the offal.

Animal by-products are classified into three categories, in accordance with their potential risk to public and animal health. Thus, depending on their age (due to BSE risk), despite the lack of any evidence to suggest that dead animals left in the wild suppose any risk of BSE transmission (CMIEET 2001, Crozet & Lehmann 2007), the carcasses of domestic ruminants are mainly classified as belonging to Category 1, the carcasses of monogastric species belong to a lower risk category (Category 2 materials) and Category 3 materials are those intended for human consumption but which are not used for this purpose. The Regulation considers as an exception the possibility of authorizing animal by-products in Categories 2 and 3 for the feeding of wild animals and, in the case of necrophagous birds, also authorizes the feeding of carcasses that are classed as Category 1 materials (cows, goats and sheep). Traditionally, gut piles are left in the own hunting states and remain available not only for obligate scavenger species but can also be used by the facultative scavenger, such as terrestrial carnivorous and omnivorous mammals. Nonetheless there is a variety of national traditions, legislative framework and regulations, ownership, which have a profound effect on the organisation and integration of management activities. Game management and hunting in Spain is regulated by autonomous regional governments. As a result there are 15 distinct regulatory and legal frameworks within continental Spain with little or no coordination at the level of the country. Due sanitary concerns (see Box 2), some Spanish regions have recently developed legislation concerning the disposal of big game remains. In

Castilla-La Mancha region, the options to manage big game by-products generated alter hunting activities (Resolution 10/02/2009 on the management of big hunting by-products) are (i) burial of gut piles in the same hunting place under certain conditions (location, deep o burial, cal, burial registration book, and authorisation previous visit an inspection by the Animal heath authorities), (ii) the move of them to be processed and destructed in authorised plants by special authorised vehicles and (iii) the move of them to legal vulture restaurant (fenced places with a constant carcass supply), laying dawn the conditions that these feeding points must meet. There are also national and regional normative creating a net of modern feeding points for vultures and regulating the feeding of necrophagous birds with animal by-products not intended to human consumption (regional: D108/2006, national: RD664/2007).

Since Regulation (EC) No. 1774/2002, all dead livestock had to be removed, using specially authorized vehicles, in order to be transformed or destroyed at approved, designated plants. This affected big game waste use because it has been seen as an alternative food supply for avian scavenger, especially in Southern Europe where this endangered animal community is more abundant. Conservation strategies (dispositions 2003/322/CE, 2005/830/CE, in Spain RD 342/2010 which modifies RD664/2007) regulating the use of animal by-products as food for necrophagous birds have been incorporated into European Commission regulations since the 2002 BSE crisis, in order to be more flexible and trying to make compatible sanitary and environmental interests. All these legal derogations aimed granting exemption from removing certain animal by-products from the carcasses, once a series of specific conditions have been met. One requirement is that the animal by-products have to be made available to animals within fenced-in sites, although in our experience there are administrative restrictions on the creation of feeding stations and numerous small feeding sites are needed to avoid excessive concentration of waste in just a few places (see comments above on the repercussions of these changes on individuals, populations and communities of avian scavengers.

6. Management

Big game waste must be properly managed to protect the continuation of natural ecological processes, to reduce their effect on animal/human health and the environment under certain circumstances, and to a lesser extent, for aesthetics purposes or to recover resources from it (such as composting). Management to control hazards associated to waste disposal in the wild starts at a pre-harvest phase. First we must known the prevalence, spatial and temporal distribution of big game pathogens (and quantifying any other hazard) in the area, what is called surveillance and monitoring (Wobeser 1994, 2002, Artois 2001, 2003). In Europe, wildlife disease surveillance is addressed by a series of regional, national and international schemes. Currently, parts of Europe benefit from wildlife surveillance efforts (frequently limited to a few diseases), while in other parts no surveillance is done at all. Proper implementation of a complete surveillance effort must be a priority of the Veterinary Authorities, since it is accepted that those countries which conduct disease surveillance of their wild animal populations are more likely to detect the presence of infectious and zoonotic diseases and to swiftly adopt counter measures (Mörner et al. 2002). We need identifying risk factors of relevant wildlife diseases (i.e. those affecting animal and human health) to effectively manage them and reduce the generation of hazardous solid waste from big game. Among risk factors, the most frequent one is the introduction of diseases through movements or translocations of wild or domestic animals. Examples include food and mouth disease in the UK (involuntary disease translocation). Introduction of exotic species (i.e. wapiti and Barbary sheep in Europe), conveys important risks of introduction of previously inexistent diseases, especially when foreign species range in or share habitats with native susceptible species, leading to situations where the native big game species become endangered (McInnes et al. 2006). For example, this is the case of the North American wapiti (carrying the trematode *Fascioloides magna*, highly pathogenic in the red deer, Novobilsky et al. 2006). Finally, the expansion or introduction of hosts has been linked to disease risks in several occasions. For example the newly established (escaped) wild boar populations in the UK and tuberculosis (Williams & Wilkinson 1998). Overabundance of wildlife is a second relevant risk factor for wildlife diseases. We above highlighted the relationship between overabundance and disease in big game species, and therefore control of the situation will lead to limiting the associated risks. The assessment of overabundance and the available management tools have been discussed recently (Gortazar et al. 2006). Management tools to estimate big game overabundance are needed for legislative purposes and for the monitoring of wildlife populations. A multidisciplinary approach is needed to diagnose if a given wildlife population is overabundant, which includes signs such as adverse effects on the soil, vegetation or fauna, poor body condition scores, low reproductive performance or increased parasite burdens infectious disease prevalences. A close monitoring of wildlife densities and diseases, the establishment of reference values for all signs of overabundance, and the mapping of the disease and density hotspots will be needed to design adequate management for each particular situation.

In addition to surveillance and monitoring, three basic forms of disease management strategies for wildlife are known: prevention of introduction of disease, control of existing disease or almost impossible eradication (Wobeser 2002). Preventive veterinary medicine is a key discipline in the prevention and control of disease in the wild and at wildlife/domestic livestock/human interface. Among the preventive actions, the most important one is by restricting translocation of wild animals to prevent movement of disease (Wobeser 2002). This includes the movement and release of farm-bred "wildlife", an increasingly popular game management tool that needs a careful sanitary control (Fernández de Mera et al. 2003). But even the movements of domestic animals can easily cause the introduction of new diseases or new vectors. A close monitoring of both wildlife densities and wildlife diseases and the mapping of the disease and density hotspots will be needed to design adequate risk control measures for each particular situation. In addition, it must be noted that any risks are not necessarily directly related to actual population size because aggregation of animals, usually due to human activities (i.e. artificial feeding), can increase the risk of pathogen transmission even in populations with a not particularly high density (Acevedo et al. 2007). European examples include bTB in wild boar, red and fallow deer (Vicente et al. 2007) and classical swine fever (Rossi et al. 2005) in wild boar, where an increase in transmission rates has been associated with artificial feeding. In many cases bans on supplementary feeding (Miller et al. 2003) can reduce the carrying capacity for ungulates and reduce population density and, just as significant, aggregation, two key factors in infectious disease transmission (Acevedo et al. 2007, Vicente et al. 2007). Once a given wildlife population is defined as overabundant, and even if there is specific evidence for disease problems, it is difficult to establish corrective management actions. One future direction is to convince hunters and wildlife managers of the benefits of management for quality, with lower densities and less detrimental consequences on habitat, wildlife and animal and human health. Moreover wildlife managers and the public may perceive animal health authorities as purveyors of bad news with no positive counterpart (Ferroglio 2003). Concerning the problem of the use of lead gunshot in terrestrial ecosystems, and particularly in big game, copper bullets have been trialled, and their accuracy and killing power compared to that of traditional lead bullets, which suggest that copper bullets are a viable alternative to lead bullets (e.g. Knot et al. 2009). In problematic areas, it can be considered restrictions on the use of lead ammunition, designed to encourage a switch to non-toxic ammunition across terrestrial habitats, to be a proportionate response to the problems associated with lead ingestion.

Attempts to control disease in wildlife populations have been based on a variety of methods. These include setting up barriers, improving hygienic measures, culling, habitat management and feeding bans, vector control, treatments, and vaccination (Wobeser 2002, Artois et al. 2003, Karesh et al. 2005, Gortazar et al. 2006). Wildlife culling is almost never an effective means of controlling a wildlife-related disease. Moreover, it is a subject of intense scientific and social debate (e.g. badger culling for TB control, Donnelly et at. 2006). Only in the case of island populations, when geographical barriers limit animal dispersal, or in the case of introduced species (pest species, where legal and social constraints to culling are minimal), or to cope with a point-source wildlife disease outbreak (centering culling on the disease focus, plus an outer ring of vaccination), is culling and eradication an option. By contrast, population reduction is a goal in many disease control efforts. This is a temporary measure, except if habitat modification is used to reduce host density more permanently, or to alter host distribution or exposure to disease agents (Wobeser 2002, Gortazar et al. 2006, Acevedo et al. 2007). Selective culling is limited to situations in which affected individuals are readily identifiable (Wobeser 2002). This has been used in several attempts to control mange in wild ungulates, but obviously with little success since not all individuals with visible lesions are detected and not all infected individuals show visible lesions. Social structure disruption with increased movement and therefore increased contact rate (at intra or interspecific level, Donnelly et al. 2006) may be counter-productive consequences of depopulation, which could be followed by rapid recovery of population size, even with increased population turnover through compensatory reproduction (e. g. wild boar and swine fever after high hunting pressure, Guberti et al. 1998). Recently, wildlife contraception has been considered, but there is still little information on the reliability of this method under field conditions (Ramsey et al. 2007). Ultimately the best management choice must run in parallel with intense campaigns that convince and inform the society (hunters, wildlife managers, etc.). Habitat management to cope with diseases may have opposite goals. For example, feeding bans can reduce the habitat carrying capacity for ungulates and eventually reduce population density and aggregation, two key factors in infectious disease transmission (Acevedo et al. 2007, Vicente et al. 2007). The identification and correction of overabundance situations is a key action in the control of many infectious diseases (Gortazar et al. 2006). Vector control has sometimes proven helpful in field experiments, but vector diversity and other factors may limit the effectiveness of this management in more complex environments. Treatment of wildlife is increasingly frequent, especially against parasites in economically valuable game species. For example, Iberian wild goats have been treated against sarcoptic mange with ivermectin (León Vizcaino et al. 2001). Anthelminthic treatments are frequent in ungulates (Fernández-de-Mera et al. 2004, Rodríguez et al. 2006). In many cases however, the actual effectiveness of these treatments is unclear and ethical and public health issues need to be addressed. For example, the use of antibiotics in game species may affect meat hygiene. Wildlife vaccination is exceptional, and it is normally limited to the most relevant diseases (those that cause serious economic losses, are almost under control in domestics, and where wildlife reservoirs are paramount). In Europe, this is the case of fox rabies (Artois et al. 1993), classical swine fever (Kaden et al. 2002), and probably soon bTB. In contrast to culling, oral vaccination has the advantages of being painless, thus avoiding animal welfare problems, and does not cause behavioural problems such as increased dispersal or immigration. Vaccination makes sense if the huge investment is the only way to control a disease in its wildlife reservoir, if the costs are clearly outbalanced by the costs of remaining passive, and provided the effectiveness and safety of the vaccine have been tested in captivity. In most cases, vaccination needs to be combined with other management measures, and the ecology of the host species needs to be considered carefully.

In summary, management of diseases and subsequent reduction in the generation of hazardous big game waste usually require a change in human activities (Wobeser 2002), and a sound scientific basis is strongly needed before suggesting any corrective measures that can create or increase conflicts between the different stakeholders: veterinary authorities, hunters, conservationists, livestock breeders and the general public. Moreover, the success of any wildlife disease and subsequent hazardous waste management action must be assessed critically, including an analysis of the costs, of the ecological consequences, and of the animal and human health and welfare benefits.

Post harvest operations occurs once big game waste is generated, we can firstly act by inspecting and identifying any hazard present in such remains. Then, the uses that can be given to the waste should depend on that. Harvest and post-harvest actions are mainly responsibility of hunters, hunting managers and land owners, with supervision of wildlife and sanitary authorities. Big game waste not suspected of being infected with a transmissible disease can also be managed in order to favour conservation and ecological processes. The disposal in the wild must meet sanitary and ecological/conservation interests and therefore the question that arise is how it should be done. Big game management is consubstantial to the production of solid residues and its destruction or disposal depends on practical, cultural, conservation, sanitary and legislative considerations. For non-hazardous big game waste the responsibility is usually of the generator. In order to prevent risks arising from wild animals, bodies or parts of bodies of such animals suspected of being infected with a transmissible disease should be subject to the rules laid down in regulations. Under certain regulations, for example those of the UE, this inclusion should not imply an obligation to collect and dispose of bodies of wild animals that have died or that are hunted in their natural habitat. Such practices for the mitigation of risks are well-established on national and regional legislation in the UE. Another consideration is that waste from hunter kills is highly aggregated in time and space, and usually takes place in a tightly circumscribed area over a narrow period of time. This very often conveys that a large amount of biomass residues have to been managed safety and efficiently in a short time and the accumulation of large amount of decomposed material environmental, sanitary or aesthetical problems. The spatial distribution of feeding stations has to be very patchy and with numerous small feeding sites, to avoid being predictable because of the repercussions of aggregated food availability changes on individuals, populations and communities of avian scavengers (Carrete et al. 2006). Future legislation should encourage the opening of numerous feeding stations supplied with low quantities of food to mimic the original condition of temporal and spatial unpredictability of carcasses and to maintain ecological relationships within the scavenger guild as suggested by Cortes-Avizanda et al. (2010), who propose: (i) carcass-size manipulation may be a valuable tool for directing supplementary feeding towards species of interest, (ii) the spatial position of the feeding station must also be taken into account in any attempt to increase the probabilities of use by breeding threatened species, (iii) diversifying the time at which carrion is left at feeding sites may be an efficient way of avoiding direct competition and small scavengers may benefit from food supplied when the predominant vultures are less active. Finally, we remark that cost studies suggest that the economic cost of this type of conservation measure is high (the creation of a new feeding station will cost between 30000 and 50000 Euros), to which must be added the running and maintenance costs of the site of around 20000 Euros/year (Donazar et al. 2009b, Margalida et al. 2010). The abandonment of safety livestock carcasses (once demonstrated that risks for BSE) is more recommendable than favours populations of wild herbivores to help to maintain populations of avian scavengers given the consequence of overpopulation discussed along this chapter.

Hygienic measures, such as correct disposal of the hunting carcasses and carcass remains should become compulsory in every country and for every hunting modality. Leaving it *in situ* is an option when big game animals that are hunted in their natural habitat are not suspected of being infected with disease transmissible to humans or animals or affected with any potential hazard ad good hunting practices are observed. This is recommendable to protect natural ecological processes, although it should follow previous veterinarian inspection of the offal. In this case, it becomes especially essential disease surveillance and monitoring of big game diseases (see above), and in general of wildlife in the area. Completely limiting access to this material would probably have important conservation consequences (Donazar et al. 2009a). We therefore need research assessing which disposal regime is most beneficial to obligate scavengers, guaranteeing their food supply, while reducing the exposure of susceptible animals to potentially infectious material. One solution that is being tested is the use of mammal proof enclosures or other effective barriers, although fences can injure birds, especially during landing and take-off, and therefore the feeding point should have a minimum area high game fences should be sufficiently separated from the disposal site.

7. Conclusion

This review shows that management hazards associated to big game waste production require a change in human activities, and a sound scientific basis is strongly needed before suggesting any corrective measures that can create or increase conflicts between the different stakeholders. Big game solid waste is produced in large amounts and worldwide. The production of high ungulate biomass is in many cases aggregated in space and time, and ususally associated to overabundance situations, which involves among others infectious hazards for scavengers, animals and humans. Proper implementation of a complete wildlife disease surveillance, monitoring and risk assessment efforts must be a priority of the sanitary authorities. Countries which conduct disease surveillance of their wild animal populations are more likely to detect the presence of infectious diseases and to swiftly adopt counter measures to effectively manage and reduce the effects of hazardous solid waste from big game. Hazardous big game waste disposal may contribute to the establishment and subsequent maintenance of pathogens and disease. Scientific knowledge (epidemiological and ecological) is essential to support equilibrated regulations and decisions on big game waste use, balancing sanity and conservation priorities. A mayor problem worldwide is bovine tuberculosis, for which transmission may get magnified among scavengers (especially vertebrate facultative ones when) due to infected gut pile disposal. This kind of situations secondary increases the risk of disease transmission from wildlife to the domestic flock and humans, which can also undermine conservation efforts if wildlife. Because obligate scavengers effectively remove infectious tissue from big game it is therefore desirable that legislation be applied in a way that allows for the selective access of avian scavengers to the resource. We need assessing which disposal regime is most beneficial to obligate scavengers while reducing the exposure of susceptible animals to potentially infectious/hazardous material. One solution that is currently being tested is the use of mammal proof enclosures or other effective barriers. At the same time, sanitary authorities should consider the removal of infected hunted animals and viscera to limit potential pathogen contamination. This arises the question on whether rigorous inspection compatible with food security policies can be done by not veterinarian persons, at least in vast areas where the prevalence of relevant diseases in many wild ungulate recommend a proper veterinary inspection. There is need also at international and national level of coordinating distinct regulatory and legal frameworks (e. g. presence of some diseases in boundary areas). Hygienic measures, such as correct disposal of the hunting carcasses and carcass remains should become compulsory in every country and for every hunting modality. Leaving it *in situ* is an option when big game animals that are hunted in their natural habitat are not suspected of being infected with disease transmissible to humans or animals or affected with any potential hazard ad good hunting practices are observed. This is recommendable to protect natural ecological processes, although it should follow previous veterinarian inspection of the offal.

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Part 2

Processing of Solid Waste

8

Vermicomposting

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1. Introduction

Million tons of organic wastes are disposed in landfill or incinerated annually. Each of these methods can make threat to environment and public health by emission of various pollutants to atmosphere, water resources and soil. Also, gathering, landfill or incineration of organic wastes imposes heavy costs to responsible organizations. In wastes landfill in addition to its restrictions such as costs and ground occupying for a long time, odor, flies and rodents, there is a threat of nitrate and other contaminants infiltration to groundwater (Primo et al., 2009; Sawyer, 1978). Air pollution is a problem in many parts of world and a loud alarm for health safety. Although waste incineration almost exterminates the organic wastes and may be a source for thermal energy, but air pollution is its serious threat and nowadays health and environmental protection organizations set so narrow emission standards and approach to these standards in landfill and incineration is costly and with some technical difficulty. Herein challenges for solving the problem of organic wastes safe disposal a biological environment friendly method can be a reliable response. For a long time composting is applied as a biological process of organic waste in many parts of world and in recent decades using some species of red worms in compost process as vermicomposting makes many advantages for the process of organic wastes biological degradation and for the finally obtained fertilizer. The organic wastes passing through the gut of the earthworm, recycled organic wastes are excreted as castings, or worm manure, an organic material rich in nutrients that looks like fine-textured soil (Dickerson, 2001).

2. Importance of vermicompost

Organic waste and especially fast degradable food waste is a considerable fraction of municipal agricultural and some industrial wastes. In many countries food waste is a big part of daily produced municipal wastes for an example the result of a study showed that Iran has a potential for production of 4 million tons compost from municipal solid wastes, annually (Faraji, 2007). Nowadays, public understanding of vermicompost process increased and its deployment to convert organic waste into vermicompost has been increasingly expanded (Tejada et al., 2009). Ease of the vermicompost process and ability of its application in various scales made the vermicomposting a popular issue almost everywhere. This developed application of vermicompost requires much knowledge of the process and its effect on quality of the obtaining fertilizer from the raw waste.

3. Vermicomposting, advantages and limitations

In vermicomposting, worms are feed by organic wastes and the worms change it to fertilizer. In this process, by feeding the worms with organic materials, some of the bacteria that have useful role in decomposition of organic wastes, added to them and expedite the organic materials' decomposition. Also these bacteria have positive effects on stabilization and making minerals applicable for plants (Asgharnia, 2003; William, 2000). Positive effect of adding vermicompost to soil for tomato had shown by Federico (Federico et al., 2006). In another research the increasing growth of rice stalks and soil fertility obtained by adding vermicompost (Jeyabal and Kuppuswamy, 2001). The worms used in the process can also as a byproduct in the process are discussed because; they do grow and multiply during the process and these organisms can used for produce various products, especially in the production of poultry and fish meal. Each earthworm body is composed of about 60-70% of protein and has much levels of essential amino acids like methionine and Lysine which the quantities is even much than livestock and fish. Worms body are consists of 6-11% fat, 5-21% carbohydrate, 2-3% minerals and some vitamins, particularly niacin and vitamin B12 are notable (Edwards, 1985). The worms' activity has negative effect on pathogens and some researchers have shown that the vermicompost is healthier than other organic fertilizers such as compost and manure (Asgharnia, 2003). Some problem associated with vermicompost is about the worms, the worms are sensitive to pH, temperature and moisture content which must be controlled during the process.

4. The worms of vermicomposting

4.1 The worm genus for vermicomposting

There are More than 3000 species of earthworm in the world which roughly found in most parts of the planet (Cook, 1996). Among these species, the ability and play an active role of *Eisenia foetida* to convert waste to vermicompost has been proven in many studies (Bansal & Kapoor, 2000). Other species of red worms or red wigglers such as *Lumbricus rubellus*, *Perionyx sansibaricus, Perionyx excavatus*, *Eisenia andreii* and some other species successfully are used in vermicompost production. They often found in aged manure piles, they generally have alternating red and buff-colored stripes and prefer the compost or manure environment. While common garden or field earthworm species such as *Allolobophora caliginosa* prefer ordinary soil and occasionally found in compost pile (Dickerson, 2001).

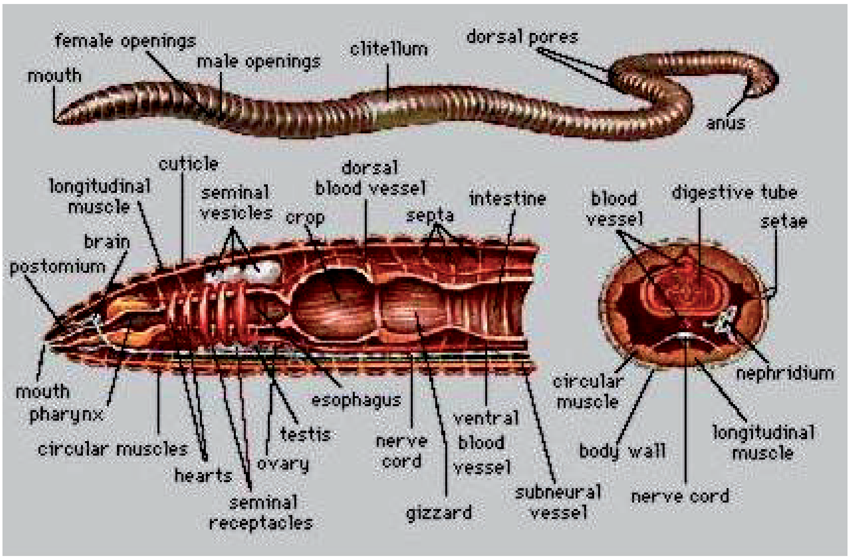
4.2 Physiology of worms and its life conditions

Earthworm body is almost cylindrical shape but may has end cross-sectional area of quadrilateral, octagonal or trapezoidal and in some species may be flat shape. Body length varies from 15 mm to 300 mm and its diameter varies from 1- 10 mm. External grooves, Furrow, on the worm body specify the place of internal curtains ,Septa,. These curtains divide the body into a series of similar parts which called Somite or Metamere. External secondary grooves, Annuli, often form three rings. The secondary grooves is a virtual division and do not exist in internal anatomy of the body. The first body segment, Peristomiom, surrounds the mouth and on the dorsal area has a lobe which called Prostmium. How to connect the mouth and Prostmium in earthworm is variable depending on the species and are used for their classification. Earth worms are androgyny and have

both male and female reproductive system which is mainly limited to the front parts of body. Earth worms have a simple digestive system. Earthworms eat almost everything such as plant roots, leaves and seeds, microscopic organisms such as protozoa, Larvae, the Rotifers, bacteria, fungi, and larger animals, especially cattle, feces. The food ingested with soil and passes along from the earthworms digestive canal. Earth worms continuously or semi-continuous are do egg-laying most often along the year. Worm eggs are placed in the cocoon. The cocoon shape is different depending on the species of worm. In moist conditions and the temperature of 16 to 27 ° C for the eggs, within 14 to 20 days the small worms come forth. Natural life of many earthworms is short and some species in case of being protected from natural hazards live longer more than 1.5 Year.
Activity, metabolism, growth and reproduction of worms are strongly affected by the temperature. Temperature and humidity usually have an inverse relation. High temperature and dry environment are more limiting than low temperatures and water saturated environment, for the worms. Earth worms setting cocoon and coming out of egg are also affected by temperature. For example, setting cocoon in *Eisenia foetida* increases linearly with increasing temperature from 10 to 25 ° C, although the number of worms per cocoon out in 25°C is less than 20 °C. Cocoon opening period also is depends on temperature. Growth of new worm out of the eggs to mature at 18 ° C reaching in 9.5 weeks and at 28 ºC only 6.5 weeks is needed (Gupta, 2004).
Worms are sensitive to hydrogen ion concentration which is stated as pH. According to sensitivity to pH in some texts have been divided them in three categories: resistant to soil acidity, sensitive and to soil acidity and a variety that can live in wide range of pH. However, many researchers have expressed that more species of earthworms show interest to live in neutral pH. *Eisenia foetida* is preferred life in the soils that pH is between 7 and 8.
The role of organic carbon and inorganic nitrogen for synthesis of cell, growth and metabolism is essential in all organisms. Proper ratio of carbon to nitrogen is needed for optimal growth of earthworms.

5. The methods of vermicomposting

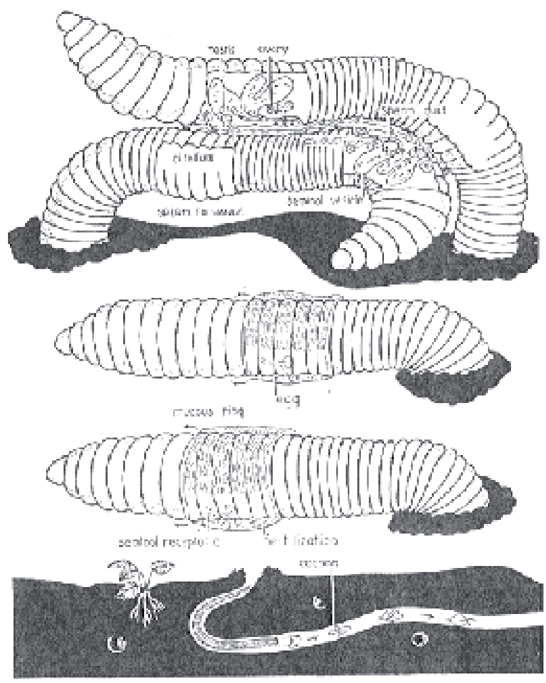
There are two major methods of vermicomposting, vermicomposting in bin and vermicomposting in vermicompost pile. The bin method is prepared to use in small scale such as home composting, in kitchen or garage and so on. The bin can be made of various materials, but wood and plastic ones are popular. Plastic bins, because of lightness, are more preferred in home composting. A vermicompost bin may be in different sizes and shapes, but its height most be more than 30 cm. bins with a height of 30-50 cm, and not so more than it, are prefect. Draining some holes in bottom, sides and cap of bin is so helpful to aeration and drainage. Around 10 holes with 1-1.5 cm in diameter is a good choice. Before feeding the worms by wastes it's needed to apply a worm's bed. A height of 20-25 cm bedding is appropriate. It may be a mixture of shredded paper, mature compost, old cow or horse manure with some soil.
Pile method mostly is used for vermicomposting in larger scale rather than bin method. Where the vermicompost is the chosen way to processing a Large amount of wastes, application of piles is cost beneficial. The piles can be made in porch place like greenhouse or in a floor with some facilities for drainage in warm climates. Although the pile size may be so various in width and length, however, it can't be so high and is better to follow the height of bin method.



(a)

(c)

(c)



(b)

Fig. 1. a and b, show the worm's body anatomy, intercourse and c and d shows the cocoon (Vermica, 2008).

6. The effect of ambient conditions and wastes on vermicompost

6.1 Effect of temperature

According to kinetics of biochemical reaction and amount of energy production in bio organisms, biological activity is depended on temperature. Bacteria activity is greatly depended on temperature. It plays a vital function in compost and vermicompost process. Also, the worms' activities are widely affected by temperature. Whole the process which named vermicomposting indeed is playing natural role of worms and bacteria in their life to live. So, this process depends on temperature. We know that the bacterial activity multiplies by two per each 10°C increase in temperature and the worms have well activity around 15-30°C. Several studies showed that a temperature range around 15-25 °C is more appropriate for vermicomposting. The most decrease in carbon percentage and C:N ratio have obtained in this range of temperature in a study among three temperature ranges of 5-15, 15-25 and 25-35°C. Also it has been the best temperature for the worms' growth (Rostami et al., 2009.a).

6.2 Effect of moisture content

The bacteria need water to proceeding biochemical reactions and many of essential substances are solved in water for transmission through membrane into bacterial cytoplasm. It's known that, the bacterial activity extremely decreases in a moisture content lower than 40% in a composting process and it almost stops in lower than 10% (Tchobanoglous et al., 1993). Also we know, the worms need to be in a moist ambient because they need to keep their skin wet for respiration through it. Recommended moisture for bacterial activity in compost process is around 55%, but the worms need some more moist to have their maximum growth and activity. It's known that, there is a relationship between moisture content and temperature in effecting on vermicompost process. In a comparative study on vermicomposting process and the worms' growth in various ranges of temperature and moisture, results showed 65-75% is a suitable range of moisture for all ranges of vermicomposting temperature (Rostami et al., 2010. a).

6.3 Effect of pH

Many kinds of bacteria can live in low pH and some live in a pH as low as 2 or even lower. Other kinds of microorganisms which are active in compost and vermicomposting are fungi which can keep their activity in lower pH around 4. Also some bacteria tolerate higher pH than neutral. However, recommended pH range for compost is around 6.5-7.5. In vermicomposting the worms are sensitive to pH and they don't tolerate a wide range of pH and they prefer neutral pHs. Although, some studies showed that the worms can be alive in some higher or lower pHs, but the recommended pH for vermicomposting is around 6-7 (Dickerson, 2001). In lower pH the bacterial activity decrease and worms which don't like it will escape to a place with better condition if they can find or most probably die.

6.4 Effect of C:N ratio

The major effect of C:N ratio in vermicompost is on bacterial activity, high C:N ratio decrease bacterial activity because of nitrogen shortage that is essential for bacteria and takes part in proteins, amino acids and other structural substances of bacteria. On the other hand low C:N ratio will led to loss of the nitrogen as in form of NH_3 to atmosphere. The worms also hate the high concentration of ammonia and will escape from it. Vermicompost process will progress properly by starting the process with a C:N ratio around 25-30 and it will decrease during the process. Carbon reduces because heterotrophic bacteria use organic material as source of electron and carbon is oxidized to CO_2 and releases to atmosphere (Tchobanoglous et al., 1993). However, bacterial nitrogen usage is so less than carbon and some kind of bacteria can stabilize atmospheric nitrogen into compost such as Rhizobium. Also, autotrophic bacteria use ammonia as source of electron and convert it to nitrite and nitrate which remain in compost unless an anoxic condition occurs. In this condition nitrate and nitrite reduced and nitrogen releases to atmosphere as N_2 (Bitton, 2005).

7. Effect of preparation time on vermicompost

Before feeding the worms with organic waste materials, organic materials are composted for a while without worms. This causes the of organic matter decomposition spent thermophilic level and the worms which are sensitive to high temperature will not damage. Also, the compost production process forward faster, and many of pathogens are destroyed in thermophilic phase. Duration of the preparation is impressive on quality of the resulting compost, vermicomposting process and space and facilities needed for preparation. Results of some studies showed that a nine-day preparation is proper (Nair et al. 2006). This time seems to be enough for pass the initial composting thermophilic period and also for loss of most pathogens (Bansal & Kapoor, 2000). In another study, the impact of preparing time on vermicompost was investigated in food waste that no amendment had been made on it (Rostami et al., 2009. b). Sometimes for better aeration or adjust C:N ratio the balking agents or other materials, such as wood chips, sawdust, manure, sludge and so on may be added to wastes as amendment. In this study, food wastes with preparation time of 0, 6, 12 and 18 days has entered in Vermicompost process and were monitored during the process. Results showed that, duration of preparation is effective on changes in C:N ratio during the vermicompost process. Best results and lowest C:N ration obtained along 6-12 days of preparation. Fig. 2 is presentation of the result. In this kind of not amended materials more preparation duration may redound on anaerobic process and as a result of the acidification phase, pH is reduced and these conditions are unfavorable for worms to live and activate. Thus, reducing the activity of aerobic bacteria and worms, the C:N ratio reducing speed is decreased. Fig. 3, Shows the trend of pH reduction consistent with increased preparation duration. It's clear that, if sufficient aeration and well composting conditions provided during preparation by material amendment, aeration or by any means, anaerobic condition will not occurs in longer preparation duration. But it needs proficiency and some cost.

8. The effect of worm population on vermicompost

In vermicompost process the worms have a vital function. So, the worms' population in waste is effective on vermicompost process and quality. So, a question about vermicompost is, how many worms most be applied for vermicomposting to get a prefect process and fine vermicompost? Some researchers have done efforts to find the answer. It is clear that, each species of worms have individual properties and the answer may be different. Some

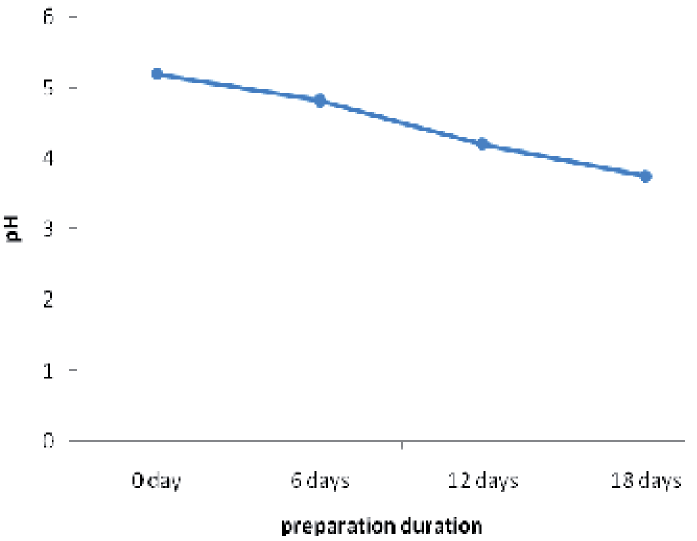


Fig. 2. Mean pH of the wastes with various preparation durations, within vermicomposting process.

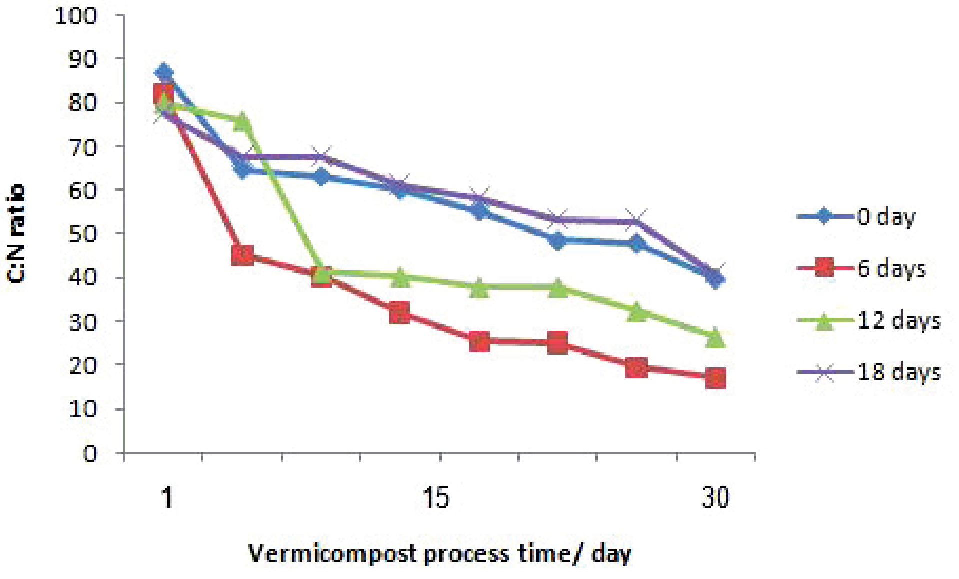


Fig. 3. C:N ratio during vermicomposting process for wastes with the various preparation durations.

studies declared that a worm can eat around as much as half weight of its body per day (Jicong, 2005). Also, some texts suggest a 1:1 ratio of worms and wastes, by weight, for vermicomposting. In a study the effect of *Eisenia foetida* species population was investigated on vermicomposting of food waste. In this study four populations of worms including, 6, 12, 18 and 24 worms set in 70g of food waste and a blank, food waste with no worm, were monitored for a month of vermicomposting. The results showed that, increasing in number of worms can be effective in maintenance of pH around neutral range. It is important during vermicomposting process. Also, it is important for the obtaining vermicompost to be at the standard range of A class's range, 6.5-8.4, (Brinton, 2000). More number of worms can much aerate the waste and prevent process from anaerobic condition which reduces pH. Also, in aerobic condition ammonia is consumed and this can prevent from much pH increasing. Best result about C:N ratio in this study has seen in the population of 18 worms per 70g of waste (Rostami et al., 2010. b). According Fig. 4, the C:N ratio declined with increasing of worm population until 18 worm and then increased in population of 24 worms. This result may be due to no more increasing of number and activity of bacteria in presence of more worms, or slaking of worms' activity which some limiting factors such as food or other factors can be causes of that.

9. Application of vermicompost

Vermicompost can be applied everywhere which wanted to help nutrition and growth of plants. There are many reports of vermicompost successful application for various plants. There are many methods to add a fertilizer. A simple method for using vermicompost is adding it as a thin layer to soil around the plant and mixing with the soil. It is very mild and overfertilizing will not result in burning the plant. Amount of using vermicompost depends on its quality and containing elements. But, there are some recommended normally amounts for different plants. An example is table 1. Period of fertilizing can be 2-6 month according to plant's demand.

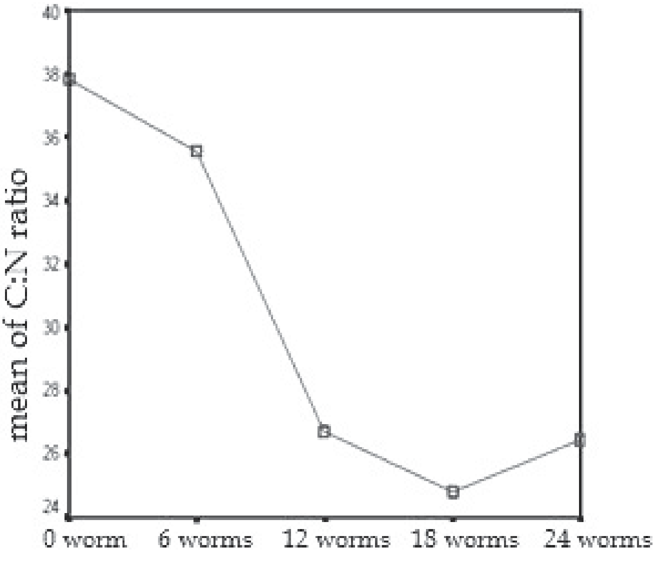


Fig. 4. The mean C:N ratio of vermicompost with various worm population.

10. Vermicompost tea

The vermicompost tea is a mixture of aerobic microorganisms which extracted form vermicompost in highly aerated water. This liquid contains beneficial bacteria and fungi which help to enrich the soil, which may be poor of microorganism in result of pesticide and inorganic fertilizer application, with these microorganisms. The aerobic microorganisms also are disease-suppressive for plant. It most noted that the leachate of vermicompost during vermicomposting process is not tea it is just vermicompost leachate and may contains significant amount of not decomposed organic material.

Plant	Amount of vermicompost (g)
Fruit Tree	1000-3000 According to age of the tree
Per each sapling and seedling forestry tree	100
Per each square meter of ornamental shrubs and grass	500
For ornamental plants, per square meter (flower types)	400
For each pot ,The average pots	80
For each pot, The Large pots	150

Table 1. Amount of vermicompost that is applied for plants

10.1 Method of tea making

The tea making commonly is performed by using a tea brewer. It is a set which aerate the water and extract tea from compost. There are many kinds of brewers in various sizes and types. Fig. 4 shows a brewer.

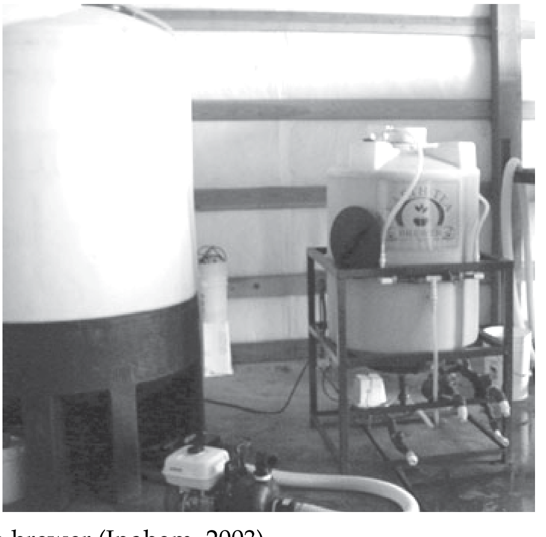


Fig. 4. A 100-gallon tea brewer (Ingham, 2003).

It is important to choice an appropriate vermicompost for tea extracting. Whatever the using vermicompost be fresh and contains more microorganism. So, the tea will be better. An incomplete and not perfectly stabilized vermicompost contains not decomposed organic materials that will be cause of quickly turning the tea to anaerobic condition and it poorly contains nutrient and microorganisms than a finished vermicompost. Nutrition of the microorganisms after brewing is substantial to keep them alive. For this purpose something such as brown sugar, honey, and black strap molasses can be added to the tea.

10.2 Advantages and limitations

Vermicompost tea has the nutrients of vermicompost. It is liquid and quickly reaches the plants' root. The tea enriches soil with bacteria and helps to soil bacterial activity. The bacteria cover roots, leafs and stalks' surface and terminate the anaerobic bacteria, pests and pathogens in a compotation. It helps plants to resist against many diseases. A limitation of tea is that, it can't be stored for a long time because bacteria in the tea need food and oxygen. Tea is a liquid rich of bacteria and its food and oxygen demand is high. So, the bacteria will die and tea turns to anaerobic in less than a day unless the food and oxygen provided.

10.3 Tea application

Tea can be applied for various kinds of plants not only for fertilizing but also for protection of plants against diseases and pests. It commonly is applied by spraying onto both sides of plants' leaves and stalk and drenching into the root zone and used as root dip for bare root. It may be applied almost any time, except in cold weather conditions when soil is below 5°C. The UV radiation harms the microorganisms and it's better to avoid times with intense sunlight. Some plants prefer bacteria dominated soil and some prefer fungi dominated soil, it's better to use vermicompost tea for the plants which prefer bacteria dominated soil because in vermicompost tea the bacteria are dominant (Ingham, 2003).

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9

Animal Manure as Alternatives to Commercial Fertilizers in the Southern High Plains of the United States: How Oklahoma Can Manage Animal Waste

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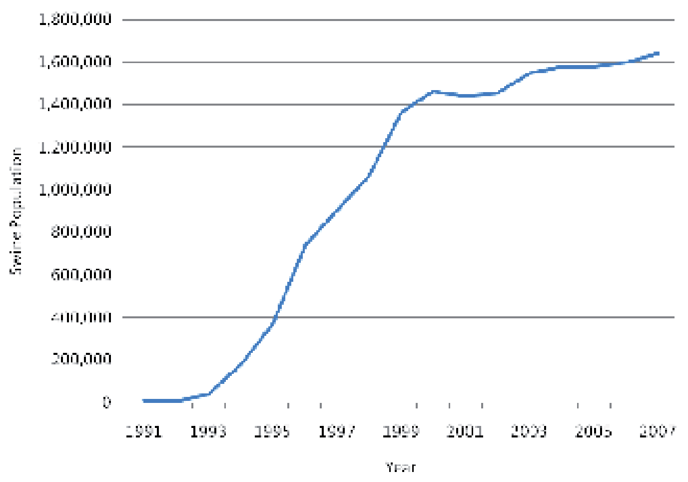
1. Introduction

The Southern High Plains (SHP) in the United States is one of the leading livestock producing regions in the US (Wright et al., 2010). More than 7 million fed cattle, which accounts for about 30% of the nation's production, are currently marketed annually in this region (Biermacher et al., 2005). Most recognize the Oklahoma Panhandle as the epicenter of the 1930's Dust Bowl in the U.S., but over the past two decades swine production in the Oklahoma Panhandle has increased 164 fold as illustrated in Figure 1 (Lowitt, 2006). Today the Panhandle is one of the more important swine producing regions in the U.S (Park et al., 2010). As elsewhere in the U.S., e.g. Iowa and North Carolina, the exponential rise in swine numbers was from the intensification of swine production, i.e. including concentrated animal feeding operations (CAFOs) and other large scale feeding operations (Williams, 2006). The Oklahoma Senate Bill 518 was passed in 1991, which eased restrictions on large concentrated animal feeding operations (Carreira et al., 2006).

A similar story has taken place in Eastern Oklahoma, which experienced a similar exponential growth of poultry production in the 1990's (Fochta, 2002). Approximately 48.2 million birds were produced in Oklahoma during 2007 (NASS, 2007). Over the past two decades, the continuous application of poultry litter, a mixture of bedding material and manure, on some poultry farm's soils has led to a build-up of phosphorus (M3-P), at times exceeding 150 and 200 mg kg^{-1} (Penn et al., 2011). Because of current environmental regulations that prevent further P application once thresholds are met, there now exists a need to move the poultry litter off-farm (Van Horn, et al., 1996; Collins and Basden, 2006).

The large-scale animal feeding operations in beef cattle, swine, and poultry production have played a major role in the economy of the southern high plains region (Carreira et al., 2007). The introduction of the animal production industries has provided a more profitable alternative to traditional agricultural enterprises in the region, such as wheat and stocker cattle, which have struggled to remain competitive with producers in the more profitable Corn Belt. For instance, the swine industry's economic importance in the Oklahoma

Panhandle includes generating more than $600 million in revenues and the creation of about 16 thousand jobs within the region. Likewise, the poultry industry in Eastern Oklahoma has generated 11,000 jobs over the past two decades and in an average year accounts for an added $700 million in revenue to the local economy.



Source: NASS (2008).

Fig. 1. Swine population numbers in Oklahoma: 1991-2007.

The growth of the swine and livestock industries in the southern high plains region has led to unintended consequences, i.e. palpable discontent and apprehension over the management of animal waste by local citizenry and environmental groups (Fochta, 2002). Environmental concerns associated with the improper management of animal waste include surface and subsurface water quality degradation (eutrophication and nitrogen leaching) and air emissions (Williams, 2006). As early as 1998, before the swine and livestock industries had yet to reach their peak numbers, citizen groups had already lobbied state government to limit further expansion of CAFOs in the Oklahoma Panhandle (Hinton, 1998). In Eastern Oklahoma, even greater opposition has surfaced as waterways have become impaired, affecting drinking water and recreational uses (DeLaune et al., 2006). The public outcry led to a series of public laws that placed stricter guidelines on the handling and use of animal wastes. The link between mismanagement of animal waste and increased phosphorus reaching waterways has led to regulations regarding the land application of animal wastes such as poultry litter (Britton and Bullard, 1998).

In the past, animal waste has been managed by applying it as fertilizer at rates that satisfy crop nitrogen recommendations, which has provided operators in areas of intensive livestock and poultry production with a means to utilize animal waste in a beneficial manner (Reddy et al., 2008; Eghball and Power, 1999). Because the nutrient ratio in litter is different from plant nutrient ratio requirements, careful consideration must be taken when land applying animal waste to avoid over-application of certain nutrients (Penn et al., 2011).

In Oklahoma, phosphorus is likely to be over-applied if animal waste is applied on the basis of satisfying nitrogen levels. Continuous application of poultry litter to plants at N recommended rates has been shown to cause an increase in soil test phosphorus (STP) beyond agronomic optimum (Sistani et al., 2004; Maguire et al., 2008). For Oklahoma, this agronomic optimum is 32.5 mg kg^{-1} Mehlich-3 P (M3-P). One consequence of increased STP is a greater potential for non-point transport of phosphorus to surface water bodies through overland flow (Johnson et al., 2004; Daniel et al., 1994). Input of phosphorus into surface waters can cause eutrophication (Williams et al., 1999; Boesch et al., 2001). Eutrophication is characterized by excess plant growth and oxygen depletion in water and can result in algal blooms, taste and odor problems, and fish kills. This not only reduces attractiveness for recreation, but creates water quality concerns for drinking water supplies. Moreover, the effects of over-application can take a few years to cause a problem.

The link between STP and increased potential transport of phosphorus to surface waters has led to regulations regarding the land application of animal wastes such as poultry litter. For example, in Oklahoma, soils within "nutrient limited watersheds" (such as the Illinois River Basin) possessing M3-P values greater than 150 mg kg^{-1} are not permitted to receive phosphorus applications. For non nutrient limited watersheds, soils with greater than 200 mg kg^{-1} M3-P are only permitted to receive a maximum phosphorus application equal to plant phosphorus removal rates (NRCS, 2007). Much of the Oklahoma poultry production is located in the eastern portion of the state where nutrient limited watersheds are abundant (Britton and Bullard, 1998).

Marketing poultry litter outside of impacted watersheds to nutrient-deficient areas offers one solution to the litter surplus problem associated with intensive animal production. Animal manure can increase farmers' profitability by providing a lower cost alternative supply of soil nutrients and usually enhances soil biophysical characteristics (McGrath et al., 2010). According to many previous agronomic studies, animal manure was found to be equally effective as commercial fertilizers for the row crops and forage production (Kwaw-Mensah and Al-Kaisi, 2006; McAndrews et al., 2006; Loria et al., 2007; Paschold et al., 2008; Chantigny et al., 2008). Agronomic benefits from applying swine effluent have also been reported, including the build-up of macro- and micro-nutrients (N, P, K, S, Ca, Mg), increased soil organic carbon, enhanced soil fertility and soil aeration, and increased beneficial microorganisms. Moreover, some studies on row crops and forages found that animal manure can be an agronomically viable substitute for inorganic fertilizers (Adeli and Varco, 2001; Brink et al., 2003; and Adeli et al., 2005).

In Oklahoma, areas outside of these nutrient-dense watersheds are typically composed of soils that are nutrient poor and low in organic matter and pH, resulting in overall poor agronomic conditions; thus, such soils in these nutrient deficient areas would benefit most from litter applications (McGrath et al., 2010; Adeli et al., 2009). However, the cost of transportation is the most limiting factor to movement of litter to nutrient-deficient areas since manure is typically too bulky to transport over long distances (Payne and Smolen, 2006). Liquid swine manure often cannot be hauled more than 25 miles, after which other manure or commercial fertilizer becomes a more economical choice. A study conducted in Alabama determined that litter can only be cost effectively transported up to 263 km from the production facility. The Alabama study showed that the 29-county region could not utilize the amount of litter produced due to high shipping costs that constrained litter movement (Paudel, 2004). Cost-share programs have been successfully implemented in both Arkansas and Oklahoma to help defray litter transportation costs. However, due to state and federal budget cuts and successful development of markets for litter, these programs are being phased out. Poultry litter has longer distances over which it can be profitably shipped compared to liquid swine manure.

One potential solution to help decrease the cost of litter transportation and allow for greater hauling distances is reducing litter mass. Traditional composting of animal manure will cause a mass reduction of 30 to 50% (Eghball et al., 1997; Rynk, 1992) due to organic carbon (C) oxidation to carbon dioxide (CO_2). However, traditional composting of litter is not always a viable option since this is a time, energy, and labor consuming process, in addition to application of C rich materials intended to decrease N volatilization. An increase in the C:N ratio occurs due to the typical application of materials with C:N ratios higher than the litter (i.e. "bulking agents"); this increase in C:N makes the material less desirable as an agronomic fertilizer by reducing the plant available nitrogen (PAN) content of the material. Since litter value (monetary) is currently based on the amounts of N, P, and K contained in "as is" litter, any increase in nutrient concentration and reduction in moisture content will increase litter value on a weight basis and increase the efficiency in which nutrients could be transported (Carreira et al., 2007).

This increase in value would allow for greater transport distances per unit mass of litter. In addition, a decrease in litter mass or increase in P concentrations via drying or organic matter decomposition would simply reduce the total mass of material needed to be transported. Thus, for poultry litter there is an opportunity to reduce litter mass and increase nutrient concentrations with little monetary and labor inputs for the purpose of reducing litter transport costs and increasing hauling distances.

Although the profitability of manure is critical to ensure that producers would be willing to apply animal waste, there has been only limited research in semiarid agroecosystems on the profitability of animal waste application. In particular, there has been limited testing on the long-term, repeated applications of animal manures in cropping systems. One objective of the chapter will present the findings from field experiments in Oklahoma that measured the yield efficacy of swine manure and beef manure, and poultry litter relative to commercial fertilizers. An economic model will be constructed for each type of manure to test its profitability, i.e. measuring its economic viability as a substitute for commercial fertilizer. Results will be presented and discussed, including a cross-cutting assessment of the differences among the alternative types of manure.

A second objective of the chapter is to determine the potential for transporting animal waste to producers in the surrounding area. To fill in this gap, a transportation model was developed using GIS that predicts animal waste movements in Oklahoma based on the supply of animal manure and demand centers. The transportation model was parameterized using the results of the field trials. Our chapter also presents findings from a poultry litter study that tested composted poultry litter, which is a less bulky form of litter that can be transported over longer distances.

The issues to be explored in this chapter, while having regional significance and importance in Oklahoma, will also resonate with national and international readers as well. Issues of animal waste management are present in other parts of the U.S., e.g. Iowa and North Carolina, and increasingly in other parts of the world such as China. At the regional level, the chapter has importance since the Oklahoma Panhandle has a limited and irregular surface water source, and elsewhere in Oklahoma groundwater is getting competitive among alternative users such as livestock production, crop irrigation, and human consumption. It is important to utilize the water and the nutrients in the manure by developing the proper animal waste management and application practices to protect waterways. So, the third objective of this chapter is to present how Oklahoma has managed animal waste over the past two decades. The comparison among the alternative sources of animal manure will be of interest to policy makers in other regions since the issues of animal waste management are present in most parts of the world.

2. Methodology

This section utilizes results from field experiments conducted at several sites in Oklahoma that tested the efficacy of manure when applied on different types of crops and forage grasses. This includes experiments on animal waste from swine, beef, and poultry producers. The data collected from the field experiments enables a direct comparison between animal waste and inorganic fertilizers.

2.1 Swine and beef manure efficacy trials: Western Oklahoma

A long-term field experiment was conducted from 1995 to 2007 at the Oklahoma Panhandle Research and Extension Center (OPREC) near Goodwell, Oklahoma (36°35 N, 101°37 W; elevation) to test the efficacy of applying alternative nutrient sources on corn and four types of forage grasses (Park et al., 2010). Annual precipitation and temperature at the Goodwell station are well representative of the climate in the Southern High Plains, with an average rainfall of 435 mm per year and an average temperature of 13.2°C, respectively. The field experiment was established on a Gruver soil series, which is classified as a fine, mixed, superactive, and mesic Aridic Paleustoll soil with a 0 to 2% slope. The Gruver soils are also typical of conditions prevailing in the region in and around the experiment station.

The experimental design for corn was a randomized, complete block design with three replications of each of the main treatment effects, nitrogen source (NS) and nitrogen rate (NR). Each of three N sources, anhydrous ammonia (AA), beef manure (BM), and swine effluent (SE), were applied at equivalent nitrogen rates of 0, 56, 168, and 504 kg N ha^{-1} yr^{-1}. Nitrogen application levels were selected on a maximum amount of swine effluent applied at 0.0205 ha-m yr^{-1} as part of the waste management system for swine confined animal feeding operation units in the region, which supplied approximately 504 kg N ha^{-1} yr^{-1}. Equivalent N rates of 504 kg N ha^{-1} yr^{-1} for AA and BM were also included in the experiment to maintain a balanced design, even though they are higher than recommended application rates. Hence, to provide meaningful comparisons with AA and BE, other NR were included. The N rate of 168 kg N ha^{-1} yr^{-1} is consistent with recommended N rates to satisfy yield goals in the region (Zhang and Raun, 2006), and a low N rate of 56 kg N ha^{-1} yr^{-1} was included to provide additional NS comparisons.

The main treatment effects were arranged in a split-plot design, with NS on each of the main plots, and the equivalent NR on the corresponding subplots. Before the experiment, continuous wheat had been grown on the test plots for several years. Nutrient levels for macronutrients (P and K) and micronutrients (Mg, Ca, S, Fe, and Mn) were found to meet or exceed plant requirements, so these nutrients were not added. Before the start of the experiment in 1995, soil P was sufficient, with an initial value of 73 kg ha^{-1}, which exceeded the recommended P level of 32 kg ha^{-1}, and remained above this level throughout the experiment (Zhang and Raun, 2006). Soil N levels were 141 kg ha^{-1} before the start of the experiment, which were about 50 kg ha^{-1} below the recommended soil N level of 190 kg ha^{-1} (Zhang and Raun, 2006). Soil pH levels were not adjusted because they would interfere with one of the long-term objectives of the experiment, which was to evaluate the cumulative effects of repeated nutrient applications on crop yields and soil properties (including pH) across different NS.

The experimental design for the grass forage study was a randomized complete block design with three replications of each of the main treatment effects, NS and NR. Each of the nitrogen sources, anhydrous ammonia (NH3), BM, and SE, were applied at equivalent nitrogen rates of 0, 56, 168, and 504 kg N ha^{-1} yr^{-1}. A total of 28 grass forage production strategies were also tested using an experimental design that included combinations of three factors: forage type, N source, and N rate. This design included four grass species (Bermuda grass, buffalo grass, orchard grass, and wheatgrass), four N application rates (0, 56, 168, and 504 kg N per ha), and two sources of nitrogen fertilizers (swine effluent and urea).

The experimental plots used a split-plot design with four replications for each of the 28 grass production strategies. In the first year of the experiment, each plot was randomly assigned to one of the strategies. Since residual effects (e.g. nutrient carry-over) were expected to have a significant effect on production outcomes, each strategy was maintained in the same plot throughout all eight years of the experiment. Swine effluent was obtained from a local anaerobic single stage lagoon near the research station, the same type of effluent available to producers. Swine effluent and urea were applied at equivalent N rates of 56 and 168 kg N per ha after the first monthly cutting in June. The 504 kg N per ha rate was split into two applications; the first application came after the first cutting in June and the second just after the second cutting in July. All plots were fully irrigated under a center-pivot irrigation system following standard practices used by producers in the region. The swine effluent was field applied through the center pivot system as part of the June and July irrigation water applications.

2.2 Poultry litter efficacy trials: Eastern Oklahoma

A short-term (3 yr) study was established at two distinct locations in the spring of 2007 with two locations: Oklahoma State University's Eastern Oklahoma Research Station located south of Haskell, OK (35 44′ 46″ N, -95 38′ 23″W) and at a site located west of Aline, OK in Woods County (36 29′ 25″ N, -98 40′ 24″ W). The three year experiments tested the efficacy of poultry litter on sweet sorghum and bermudagrass, each having an importance in the region as a source of animal feed and potential biofuel feedstock (Penn et al., 2011). Test plots were established on a Taloka fine mixed, active, thermic, mollic, Albaqualfs at Haskell and Eda mixed, thermic Lamellic Ustipsamments at Woods. A randomized split-plot design was employed similar to design of the swine effluent trials in Western Oklahoma discussed previously. Inorganic commercial fertilizer was applied at equivalent N, P, and K rates of the poultry litter based on the prior analysis of the litter using urea, di-ammonium phosphate, and potash. Degraded litter was applied at the same N, P, and K rate of the fresh poultry litter. The Haskell site has on average 215 growing days, with an average temperature of 15.5^{o}C, and 1130 mm of precipitation a year. The Aline site has on average 191 growing days, with an average temperature of 14.3^{o}C, and 683 mm of precipitation a year.

2.3 Transportation model of animal waste movements

An animal waste transportation model was constructed to evaluate the economic benefits of applying litter, swine effluent, and beef manure as a substitute for chemical fertilizer (Penn et al., 2011). The transportation model was constructed utilizing the results of the field trials discussed above. Animal waste movements are projected in the model by minimizing shipping costs between source and destination, i.e. poultry producers supplying litter and farms demanding the contained nutrients. In addition to the transportation costs, handling and applications costs are also included in the model for animal waste movement. Benefits are measured as the cost of applying macronutrients (NPK) using animal waste compared to chemical fertilizers. Field application rates for animal waste were obtained based on the results of the field efficacy trials.

The transportation model projects animal waste movements by minimizing shipping costs between source and destination, i.e. between animal producers supplying animal waste and farms demanding the contained nutrients. The cost of transporting animal waste from source i to destination j is given by the following equation:

$$\text{TRNSP COST}=\sum_i\sum_j\sum_t D_{ij}C_{ij}Q_{ijt}X_{ijt} \tag{1}$$

Where D_{ij} is the distance from i to j, X_{ijt} is the binary decision variable that determines whether animal waste is shipped in year t (X_{ijt}=0 no shipment, X_{ijt}=1 shipment), Q_{ijt} is the quantity of animal waste shipped in year t, and C_{ij} is the unit cost of transporting animal waste from i to j in year t. In Oklahoma, this requires moving litter from the eastern part of the state where poultry operations are concentrated to producers in the central part of the state where wheat and hay production is primarily located, and likewise moving swine effluent and beef manure in the western part of the state to farms in and around the Panhandle. In addition to the transportation costs, handling and applications costs are also included in the model for poultry litter, swine effluent, and beef manure. When combined with the transportation costs from Equation 2, the total cost of transporting, handling, and field applying animal waste is given by the following:

$$\text{TOTAL COST}=\sum_i\sum_j\sum_t (D_{ij}C_{ij}+H_{ij}+A_{ij})Q_{ijt}X_{ijt} \tag{2}$$

where H_{ij} and A_{ij} are the handling and field application costs for poultry litter for each unit of poultry litter shipped from source i to destination j.

Constraint relationships were included in the model to ensure compliance such that the accumulated soil P levels from applied animal waste were held under 32.5 mg kg^{-1}. Using similar notation to Equation 2, the soil phosphorus constraint equation is given by the following inequality:

$$\sum_i\sum_t X_{ijt}Q_{ijt}\text{PHOS}\le\overline{P_{soil}}\quad \text{for all j} \tag{3}$$

Where PHOS is a coefficient that relates the quantity of animal waste applied at site j in year t to the long-run accumulation of phosphorus in the soil and $\overline{P_{soil}}$ is the upper limit on soil phosphorus levels. Optimum soil test P concentration for agronomic production in OK is 32.5 mg kg^{-1} (M3-P soil extraction; Mehlich, 1984). For P demand and crop production, it was assumed that no P would leave the farms receiving litter; this provided a “worst case scenario” for moving either poultry litter, swine effluent, or beef manure. The increase in soil test P with litter applications was estimated using relationships developed for Oklahoma soils (Davis et al., 2005).

Economic benefits were determined in the transportation model by the cost savings in applying equivalent nutrient levels from animal waste versus commercial fertilizer sources according to the following equation:

$$\text{BENEFITS} = \sum_{i}\sum_{j}\sum_{t} Q_{ijt}\Phi_{NPK}\text{PRC}_{NPK}X_{ijt} \tag{4}$$

Where Φ_{NPK} is the transformation coefficient governing the content of NPK per unit of animal waste and PRC_{NPK} is the vector of N, P, and K prices. This valuing approach also enabled a direct comparison between commercial fertilizer and animal waste. Animal manure demand was estimated based on its use as a substitute for N, P, and K from commercial fertilizer. Poultry litter applications were applied in the model based on observed crop and hay acreage at the county level (NASS 2009) and achievement of 32.5 mg kg^{-1} M3-P, which established an aggregate demand for P. Current average soil P levels were estimated using soil test samples from Oklahoma State University's Soil Testing Laboratory, which contains records of 65,000 soil samples.

Consistent with Carreira et al. (2007), poultry litter, swine effluent, and beef manure was valued using commercial fertilizer prices to establish nutrient prices for N, P, and K (Oklahoma State University Nutrient Management; NPK, 2010). The model used results from previous field experiments discussed in the previous section to value poultry litter, swine effluent, and beef manure on a weight basis (i.e. the estimate of Φ_{NPK}) based on the measured concentrations of N, P, and K in each type of animal waste. Poultry litter's macronutrient contents for NPK (77% dry matter) were measured at 3.15% for nitrogen, 3.05% for phosphorous, and 2.50% for potassium. For swine effluent, NPK contents were measured at 0.21%, 0.05%, and 0.25%, respectively. At 62% dry matter content, beef feedlot manure had NPK contents of 1.2%, 1.05%, and 1.25%. Transportation, handling, and field application costs for poultry litter used in the economic model (see Equation 3) were obtained from Carreira et al. (2007), with values of C_{ij} = \$0.10 Mg^{-1} km^{-1}, H_{ij} = \$18.73 Mg^{-1}, and A_{ij} = \$ 7.72 Mg^{-1}. The corresponding values for swine effluent and beef manure were obtained from Park et al. (2010).

The transportation model was solved by maximizing the difference between the BENEFITS and COSTS equations subject to maintaining soil P levels within the prescribed limits dictated by Equation 3. The General Algebraic Modeling Systems (GAMS) software package was used to find the optimal solutions to the transportation modeling formulation given by Equation 1 through Equation 4. Results were then linked to the Arc-Maps GIS system where maps were created to present results of the transportation flows. The transportation model was solved under two scenarios. In the first scenario, poultry litter was prepared conventionally. In the Compost Scenario, all of the poultry litter was presumed to be prepared as compost. While complete adoption of compost is not anticipated, the scenario establishes the upper limit on the benefits of compost.

2.4 Statistical analysis

Analysis of variance models (ANOVA) were constructed for the crop and forage yields and corresponding economic returns using the SAS PROC MIXED routine (SAS Institute, 2002). For all of field studies, ANOVA models were used to determine if there were significant differences among main treatment effects, which varied between each study. In the poultry litter study, chemical properties ($p = 0.05$) between the fresh litter (day 0) and degraded litter (i.e. day 60) were tested. The effects of nitrogen source and nitrogen rate were included in the model as fixed effects along with covariates rainfall and irrigation. The small scale poultry litter storage had year as the blocking factor (three years) and two different treatments; normal litter and alum amended litter. The swine effluent and beef manure experiments in Western Oklahoma tested nitrogen source and nitrogen rate. The economic profitability of each nitrogen source was calculated as the gross income (corn price × yield) minus total specified costs. Sensitivity analysis on the economic models was also obtained to illustrate using break-even analysis how alternative prices would affect profitability.

3. Results

The results from the field trials in Western Oklahoma showed that both swine effluent and beef manure generated significantly higher corn yields than anhydrous ammonia (Figure 2). The highest corn yield was found when beef manure was field applied at a rate of 168 kg per ha, but the yield was not statistically different from those with swine effluent when applied at that same rate (Figure 2). While at the lower nitrogen rate of 56 Kg N per ha no effect of nitrogen source on corn yield was found, greater mean separations were found among the nitrogen sources when the rate of nitrogen application was increased. At the highest nitrogen application rate of 504 Kg N per ha, swine effluent generated the highest corn yield, followed by beef manure and then anhydrous ammonia. The superior performance of the animal manures (swine effluent and beef manure) over commercial fertilizers can be explained by enhanced soil components such as the addition of micronutrients and organic matter, and improved soil pH levels.

In terms of forage production systems, higher dry matter yields were observed in urea than swine effluent for the summer forage grasses, whereas swine effluent had higher forage yields than urea for the winter forage grasses. However, in both the winter and summer grasses the yield differences between urea and swine effluent were not statistically significant according to the ANOVA model. Unlike what was found in the corn experiments, there was no separation of mean yields between swine effluent and urea as the application rate of nitrogen was increased. The overall conclusion of the forage grass study was that no significant difference in dry matter yield was found between swine effluent and urea, which provides empirical evidence that swine effluent can be an equivalent substitute for the commercial fertilizer for forage production systems commonly used in the Panhandle region.

Economic comparisons among the alternative nitrogen sources tested in the Western Oklahoma field trials are presented in Figure 3. Both beef manure and swine effluent generated higher economic returns than anhydrous ammonia under corn production. The highest economic return was found with swine effluent, but its returns were not significantly different from those of beef manure. Less separation among mean economic returns was found at the lower and middle rates of nitrogen application, 56 and 168 Kg N per ha, but swine effluent generated the highest economic return at the highest rate of nitrogen application followed by beef manure and anhydrous ammonia.

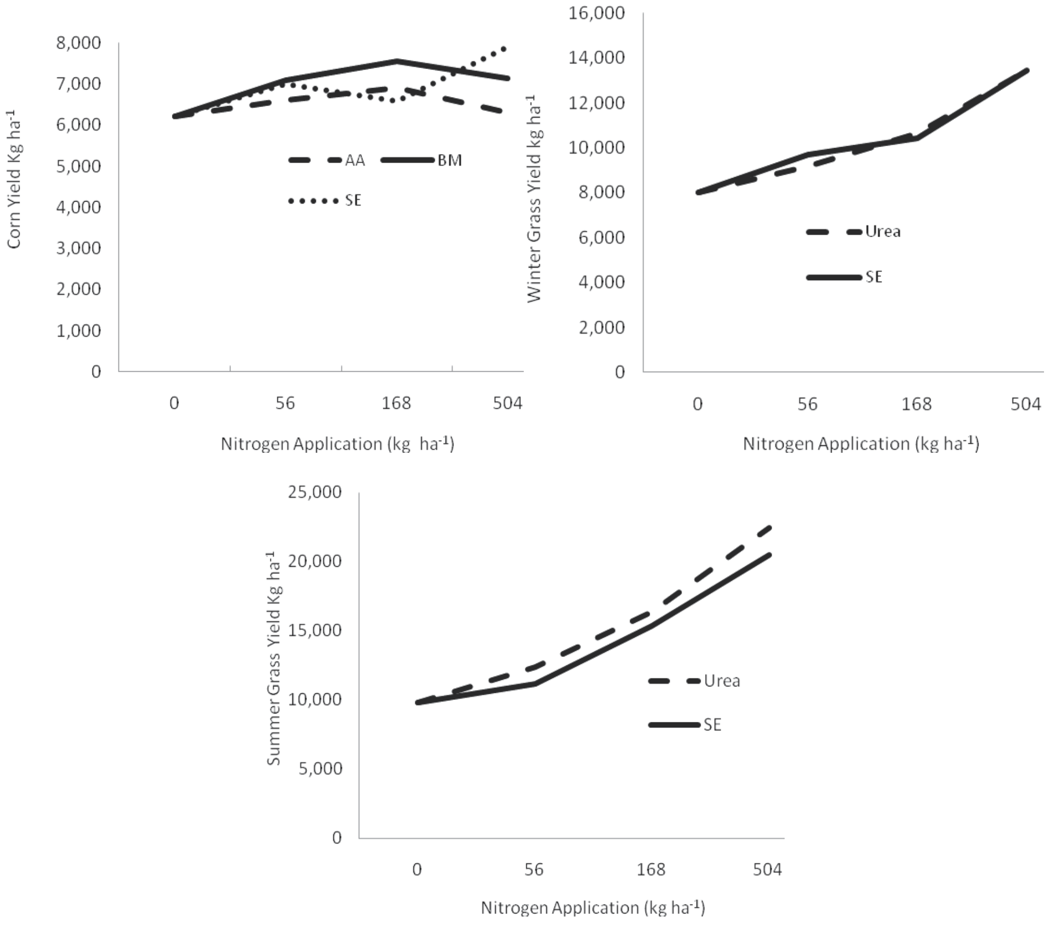


Fig. 2. Results of the swine effluent and beef manure field trials in Western Oklahoma on corn, winter grass forage (orchard grass and wheatgrass), and summer grass forage (bermudagrass and buffalo grass).

Under the forage production systems, swine effluent generated significantly higher economic returns than urea in the summer forage (Figure 3). Greater mean separations of economic returns between swine effluent and urea were found as the rates of nitrogen application increase. In the winter forage, higher economic return was found in swine effluent but was not significantly different from that in urea. Also, there was no mean separation of economic returns between swine effluent and urea as the N rates increase.

In summary, the field experiments in Western Oklahoma show that swine effluent and beef manure can be economically viable substitutes for commercial fertilizers when applied on corn, one of the major crops in the region. Both types of animal manure can also be applied economically on forage grasses, crops that commercial fertilizers are typically applied on less intensively since they are not as profitable. Hence, swine effluent and beef manure can benefit producers in the Oklahoma Panhandle by generating higher yields and economic benefits compared to commercial fertilizer.

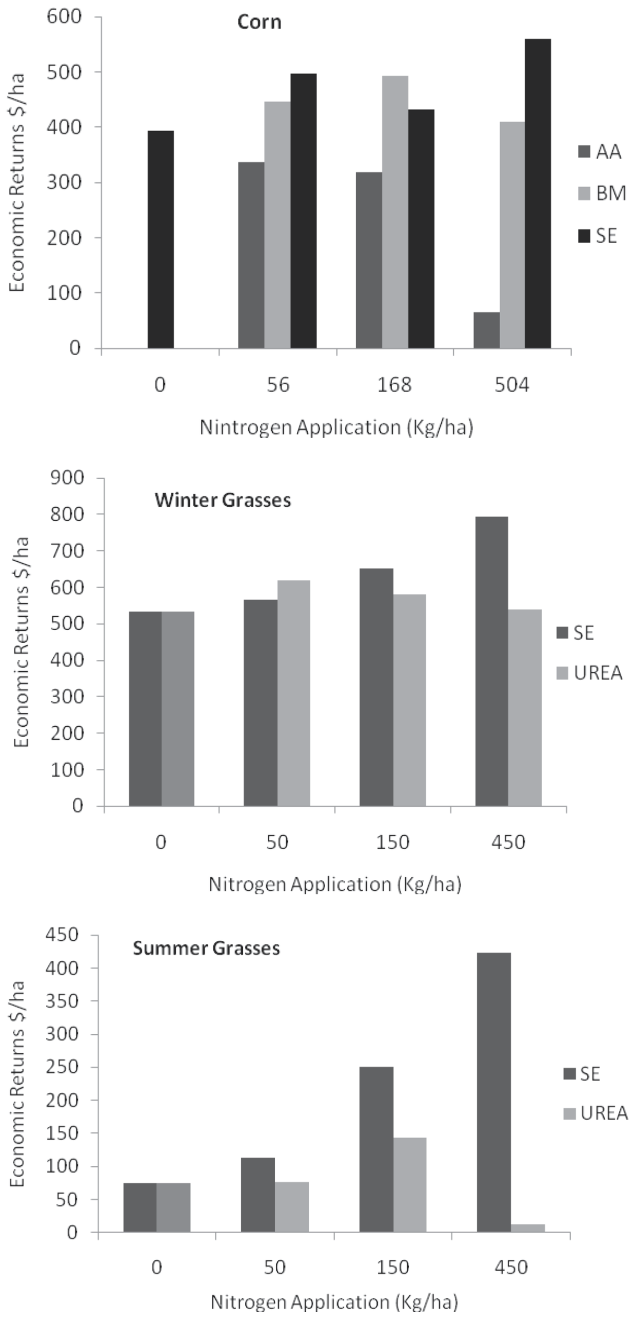


Fig. 3. Economic comparisons among swine effluent, beef manure, anhydrous ammonia, and urea when applied on corn, winter grass forage (orchard grass and wheatgrass), and summer grass forage (bermudagrass and buffalo grass). Each panel in the figure summarizes the findings of the field trials in Western Oklahoma from several years of field trials (1999-2007)

3.1 Poultry litter

Poultry litter was found to be similar to commercial fertilizer when applied at commensurate levels (Figure 4). Although sweet sorghum and Bermudagrass yields had slightly higher values on the commercially fertilized plots compared to the poultry litter plots, the difference was sometimes not statistically significant (P>0.05). Sweet sorghum yields reached 19.5 Mg ha^{-1} when 240 kg ha^{-1} of nitrogen was applied with commercial fertilizers, and with poultry litter sweet sorghum yields were 15.8 Mg ha^{-1}. Averaged over 3 years, bermudagrass biomass yields responded to fertilizer rate in a linear fashion for both fertilizer types ranging from 2.95-5.82 Mg biomass ha^{-1} (Figure 4). The 2008 and 2009 growing season was extremely dry with significantly higher yield for inorganic fertilizer sources than poultry litter. This was likely due to the lack of mineralization due to the dry conditions. On the other hand, 2007 bermudagrass biomass yield was not significantly different between the two nutrient sources; this was likely due to the fact that 2007 was a wet year and moisture was not limiting for mineralization to occur.

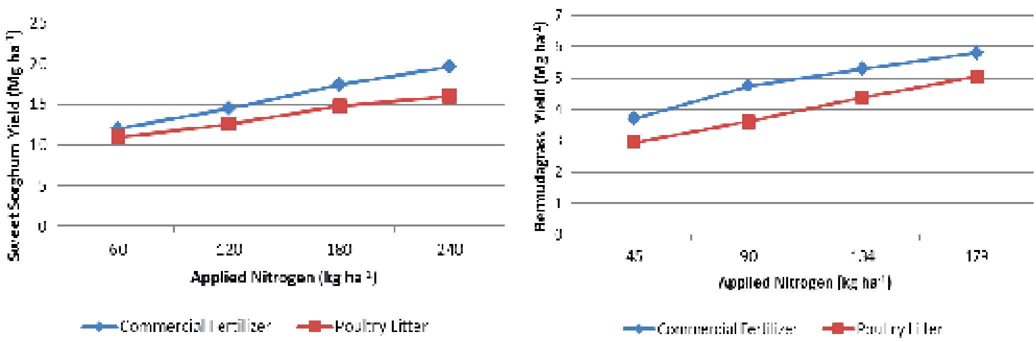


Fig. 4. Response of nitrogen application and source (poultry litter and commercial fertilizer) on sweet sorghum and Bermudagrass yield based on three years of field trials in Eastern Oklahoma. Note that for each nitrogen rate, an equivalent amount of phosphorus and potassium was also applied among the two nutrient sources. Results of the field trials showed no significant difference in yield among the nutrient sources.

Haskell sweet sorghum biomass yields in 2007 and 2009 were not substantially different due to non-ideal conditions, while 2008 produced significantly higher yields and was the only year with a linear response to N and significant differences between fertilizer type as the inorganic outperformed the litter. Yield over the 3 years ranged from 9.1-29.7 Mg ha^{-1}. Woods county sweet sorghum produced a linear result to N application with no difference between fertilizer types with yield ranging from 4.9-7.9 Mg ha^{-1}. No significant difference between fertilizer types was observed for nutrient uptake among both crops and sites. Nutrient removal appeared to be controlled by the rate of fertilizer applied and total biomass removed.

The field trials also tested degraded litter and found that it also provided equivalent agronomic performance to commercial fertilizer when applied on equivalent nitrogen and phosphorus basis (Penn et al., 2011). Use of poultry litter appears to be a good alternative to inorganic commercial fertilizer especially when P and K deficiencies are present and ideal mineralization conditions occur.

The results of the field trials in Eastern and Western Oklahoma are important since they indicated that animal manure, when applied at equivalent rates with commercial fertilizer, performs equally well as commercial fertilizer. Moreover, the field trials in Western Oklahoma suggest that animal manure can provide enhanced agronomic performance due to increased levels of micronutrients and the ability to maintain soil pH. Such agronomic benefits are also anticipated to be present in poultry litter. For example, bermudagrass and sweet sorghum plots treated with poultry litter result in a significantly higher soil pH after three years of annual applications compared to commercial fertilizer treatments. In addition, sweet sorghum litter treated plots possessed a significantly greater soil aggregate stability (indicator of soil quality) at high application rates compared to commercial fertilizer. Given the substitutability of animal manure with commercial fertilizer, economic benefits can be achieved if manure can be marketed, transported, and field applied at lower cost than commercial fertilizer. The next section presents the anticipated benefits from animal manure based on the findings of the Oklahoma field trials and the economic model described above.

3.2 Transportation model

Results of the transportation model project the optimal movement of Oklahoma's annual production of animal manure over a 50 year period (Figure 5). As illustrated in Figure 2, the movement of animal manure is greatly determined by the shipping costs and the location of the animal producers. Poultry litter is shipped the furthest and swine effluent the shortest. poultry litter is generally shipped westward from the eastern portion of Oklahoma in and near the Illinois River watershed, to locations that reach up to 200 miles away. By year 25, the model projects poultry litter movements that reach roughly one-half of the state (Figure 5). Swine effluent, due to its bulkiness, is primarily confined to the Oklahoma Panhandle region. Movements of beef manure are also primarily concentrated in the Panhandle, but there are a couple of other areas in the state with noteworthy movements of beef manure.

Poultry litter would provide the largest movements of nitrogen and phosphorus over the first 10 years, with 58,457 metric tons of nitrogen and 24,712 metric tons of phosphorus delivered to producers (Table 1). Swine effluent would deliver nearly the same quantity of nitrogen as poultry litter over the first ten years, 58,245 metric tons, however the swine effluent would deliver significantly less phosphorus, 6,055 metric tons (Table 1). This is simply due to the fact the swine effluent contains very little P compared to poultry and beef manure. With the largest quantity of macronutrients delivered, poultry litter would generate the greatest economic benefits, $37.4 million, which corresponds to an average benefit of $3.75 million per year. The model projects that swine effluent would result in $28.2 million in economic benefits and beef manure an additional $6.48 million (Table 1). The total economic benefits to Oklahoma producers from the movement of all three types of animal manure would be $72.0 million (Table 1).

Animal waste movements change noticeably over the next fifteen years. By year 25, swine effluent would account for the largest movement of nitrogen, while poultry litter would still be the largest deliverer of phosphorus (Figure 6; Figure 7). According to the model results, swine effluent would deliver 145,613 metric tons of nitrogen to Oklahoma producers, compared to the 103,715 metric tons of nitrogen that that poultry litter is projected to deliver (Table 1). Swine effluent would deliver 40.4% more nitrogen in year 25 than poultry litter, reversing the trend that occurred during the first 10 years.

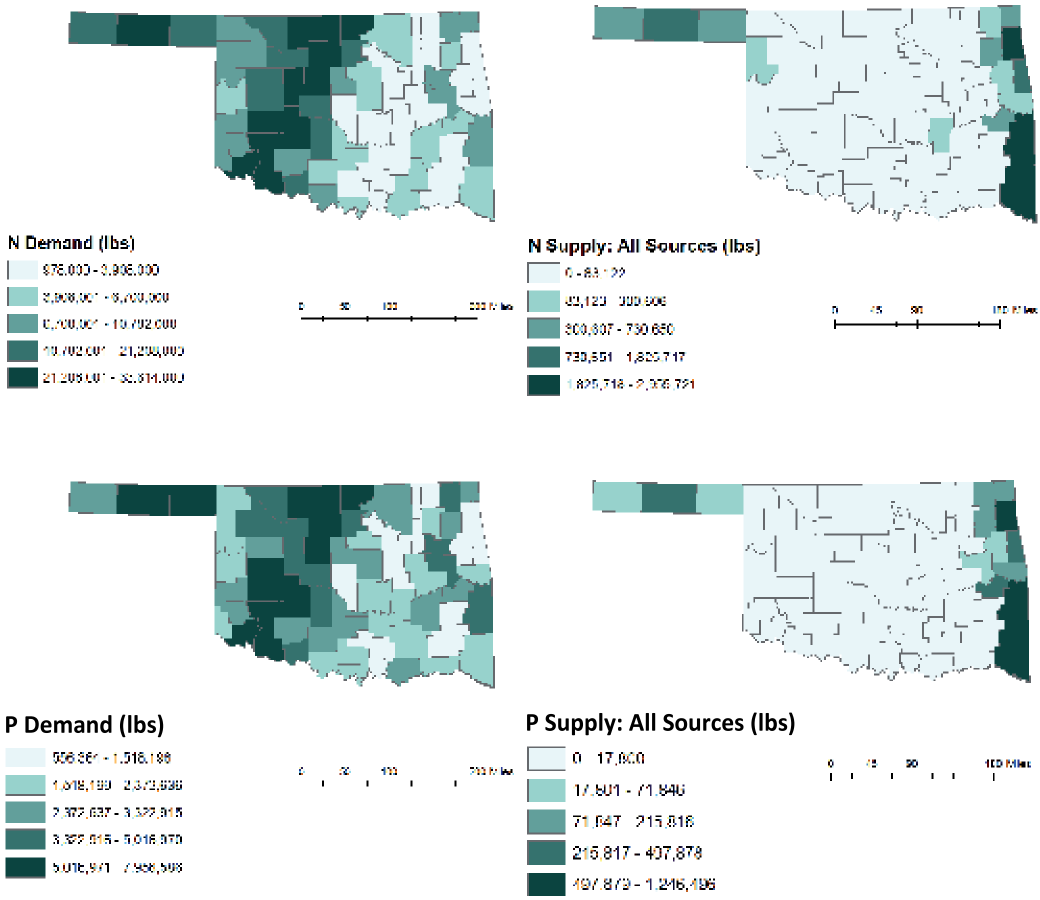


Fig. 5. Demand and potential supply of nitrogen and phosphorus in the state of Oklahoma from all three major animal waste sources, poultry litter, swine effluent, and beef manure.

While poultry litter would still be the largest supplier of phosphorus to producers, delivering 43,844 metric tons of phosphorus to producers by year 25, its' annual delivery has decreased compared to swine effluent and beef manure. For instance, in the first 10 years of the analysis poultry litter delivered an average of 2,417 metric tons of phosphorus per year. Between years 10 and 25, however, phosphorus deliveries in Eastern Oklahoma declined to an average of 1,275 metric tons per year. During that same 25 year period, both swine effluent and beef manure delivered the same quantity of phosphorus each year, 605 and 96 metric tons (Table 1). This decline in phosphorus movements from poultry litter is a result of the combination of two factors: poultry manure has a higher P concentration compared to swine and beef manure, and P is able to "build up" in soils (unlike N) to a point in which an agronomic optimum P level is achieved (32.5 mg kg^{-1} Mehlich-3). In other words, soils in Eastern Oklahoma will reach the agronomic P optimum level more quickly, preventing further delivery of manure. This requires poultry litter movements shift further west, increasing transportation costs.

	Conventional Litter Scenario				Degraded Litter Scenario			
Time (Yrs)	Convent Litter	Swine Effluent	Beef Manure	Total	Degraded Litter	Swine Effluent	Beef Manure	Total
	Animal Waste Delivered (metric tons)				Animal Waste Delivered (metric tons)			
5	927,887	57,898,488	105,315	58,931,689	927,887	57,898,488	105,315	58,931,689
10	1,855,774	115,796,975	210,629	117,863,379	1,855,774	115,796,975	210,629	117,863,379
25	3,292,537	289,492,439	526,573	293,311,548	3,912,093	289,492,439	526,573	293,931,104
50	3,658,328	117,626,890	1,053,146	122,338,364	3,891,780	491,092,266	1,053,146	496,037,192
	Nitrogen Delivered (metric tons)				Nitrogen Delivered (metric tons)			
5	29,228	29,123	1,264	59,615	29,228	29,123	1,264	59,615
10	58,457	58,245	2,528	119,230	58,457	58,245	2,528	119,230
25	103,715	145,613	6,319	255,647	123,231	145,613	6,319	275,163
50	115,237	247,016	12,638	374,892	122,591	247,016	12,638	382,245
	Phosphorus Delivered (metric tons)				Phosphorus Delivered (metric tons)			
5	12,356	3,027	483	15,866	12,356	3,027	483	15,866
10	24,712	6,055	966	31,732	24,712	6,055	966	31,732
25	43,844	15,137	2,414	61,395	52,095	15,137	2,414	69,645
50	48,715	25,678	4,828	79,221	51,824	25,678	4,828	82,330
	Economic Benefits ($ million)				Economic Benefits ($ million)			
5	24.2	14.1	3.2	41.5	29.9	14.1	4.03	48.1
10	37.4	28.2	6.48	72.0	46.9	28.2	8.06	83.1
25	46.2	70.5	14.8	131.4	62.8	70.5	18.7	151.9
50	48.5	120.0	21.3	189.8	65.0	119.9	29.2	214.3

Source: Authors' calculations.

Table 1. Potential animal waste transport and economic benefits of beef manure, swine effluent, and poultry litter relative to commercial fertilizer as determined by the transportation model.

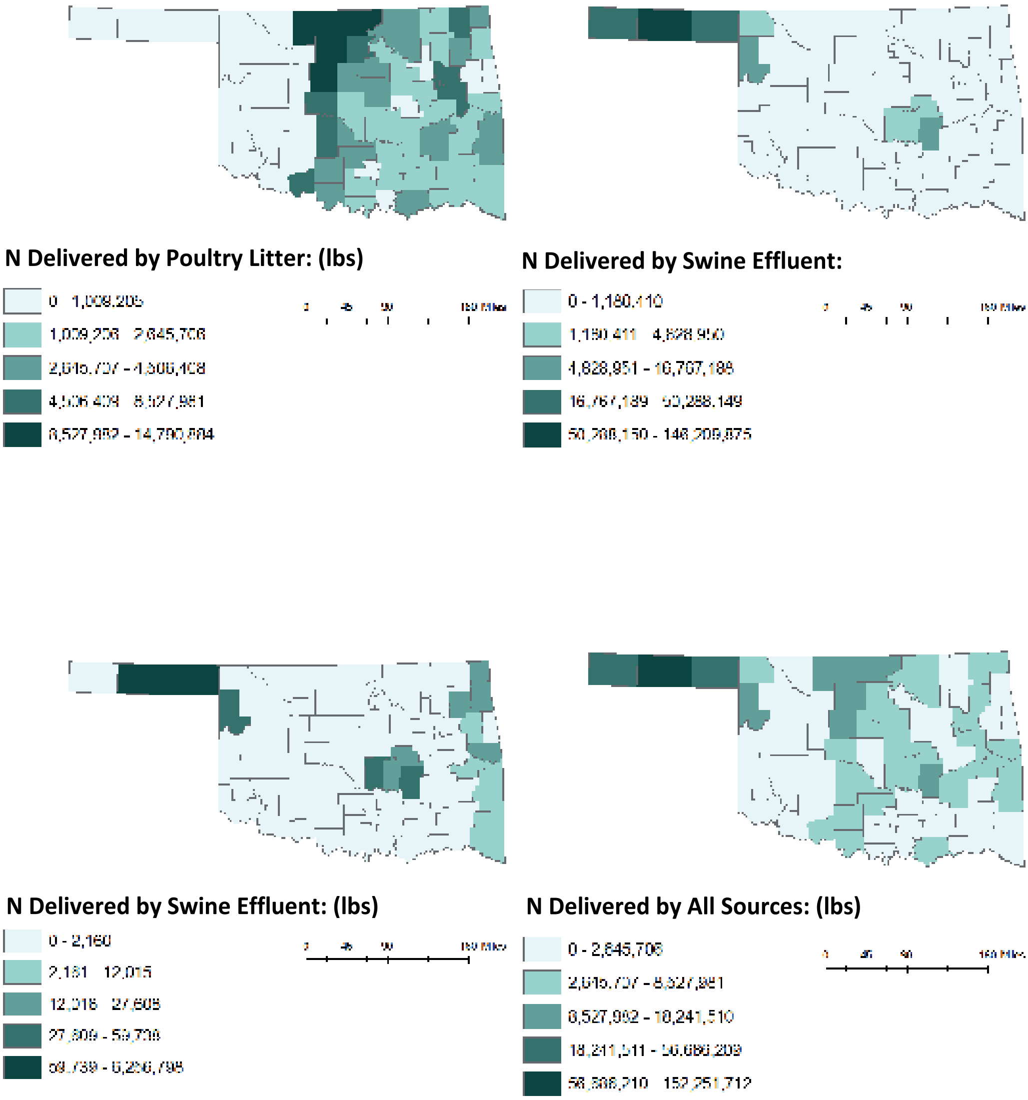


Fig. 6. Nitrogen delivered by the major animal waste types when field applied on crop and pasture lands over the span of 25 years.

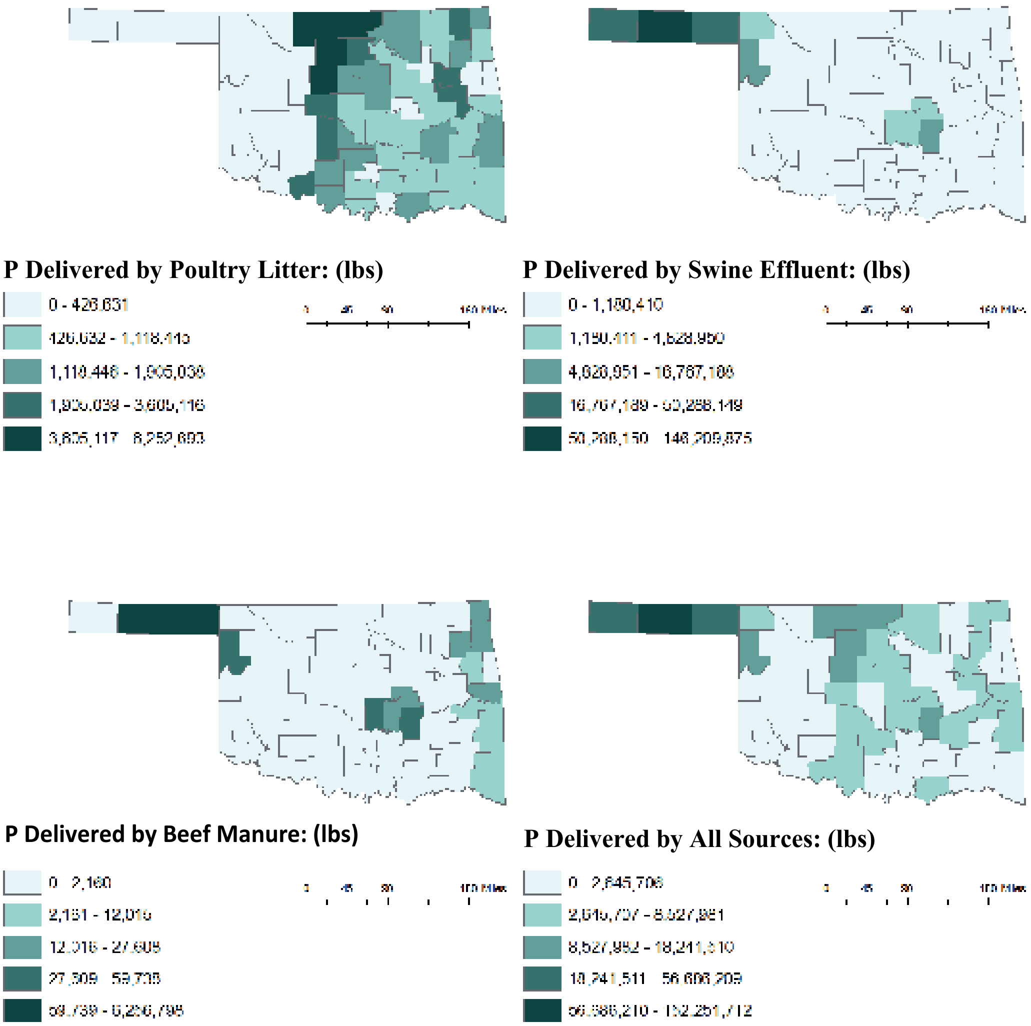


Fig. 7. Phosphorus delivered by the major animal waste types when field applied on crop and pasture lands over the span of 25 years.

The long-term buildup of soil P concentration in Eastern Oklahoma, and its effect on limiting the movement of poultry litter, is even more apparent by year 50. In the transportation model, as soil P levels reached the state mandated threshold level of 32.5 mg kg^{-1}, poultry litter had to be shipped (if economically feasible) further west at increased transportation costs. This reduced the quantity of poultry litter transported off-farm and delivered to producers. By year 50, only 195 metric tons of phosphorus would be shipped from poultry producers in Eastern Oklahoma, due to the build-up of soil phosphorus (Table 1). This is a substantial decline compared to the first few years, when poultry producers delivered an average of 2,417 metric tons of phosphorus per year (Table 1). However, in Western Oklahoma, many soils are deficient in phosphorus and better able to accept phosphorus applications. Typically, in Western Oklahoma swine effluent and beef manure applications are limited by nitrogen instead of phosphorus.

With the build-up of phosphorus in Eastern Oklahoma, the economic benefits of transporting poultry litter decline at a much faster rate than for swine effluent or beef manure (Table 1). This decline in benefits is evident by year 25, when swine effluent has for the first time, as reported in Table 1, the largest economic benefit. According to the transportation model, swine effluent would generate \$70.5 million in economic benefits by year 25, compared to \$46.2 million that poultry litter is projected to benefit. The difficulties in transporting poultry litter in an economical manner becomes even more apparent between years 25 and 50 in the model analysis, when the economic benefits from shipping poultry litter increases by only \$2.3 million. In comparison, swine effluent increased economic benefits generated by \$45.5 million during that same period of time, without any noticeable decline in benefits from one year to the next (Table 1).

3.3 Degraded litter

In the degraded litter scenario, degraded litter was placed in the transportation model to assess what impact it has on reducing transportation costs and increasing distances over which poultry litter can be shipped. According to the model results, degraded litter has a higher economic value than conventional litter. When equal quantities of degraded and fresh poultry litter are transported (70% dry matter), degraded litter has a greater value since its total phosphorus and potassium concentrations are larger than conventional litter, with only a small decrease in total nitrogen concentrations. According to the field trials on degraded litter (see above), both phosphorus and potassium can be transported at lower costs when shipped as degraded litter rather than conventional fresh litter. On a standard truck unit carrying 21.7 metric tons of litter (70% dry matter), degraded litter would be able to deliver 337 kg of phosphorus, significantly more than the 266 kg delivered by non-degraded litter, assuming P concentrations determined in the small scale study. Based on a typical hauling distance of 80 km, results of the field trials imply that degraded litter would increase economic benefits by \$180.96 per haul due to overall higher nutrient concentration. There would also be a substantial increase in the break-even distance over which litter could be profitably transported. Degraded litter could be transported as far as 416 km from the farm-gate, 82 km further than conventional litter's break-even distance of 334 km, when based on current market values of nitrogen, phosphorus, and potassium (Oklahoma State University NPK, 2010).

Degraded litter would also result in more efficient and effective movement of phosphorus out of nutrient limited watersheds that are concentrated primarily in Eastern Oklahoma (Table 1). Within the first five years, according to the transportation model, the seven major poultry producing counties in the Illinois River watershed would have produced and applied enough P on their soils to meet agronomic P requirements within their respective county. Once phosphorus thresholds are reached, poultry litter needs to be exported to non-poultry producing counties, generally located further west. Because of lower transportation costs and a greater break-even shipping distance, at some point in time degraded litter would have a larger market area and would be able to deliver larger quantities of P than conventional litter as indicated by the larger shipping quantities (Table 1). According to the transportation model, by year 25 degraded litter would be able to access producers in Western Oklahoma that would be out of the economic grasp of conventional litter due to the higher shipping costs (Table 1). The most noticeable effect of degraded litter appears after year 25. At this time, conventional litter would have nearly reached its break-even distance, meaning that the cost of hauling litter beyond the break-even distance would exceed the litter nutrient value, rendering the movement of conventional unprofitable. As discussed in the previous section, there was little movement of poultry litter after year 25 (Table 1). For instance, over the last 25 years of the modeling scenario only 365, 790 metric tons of conventional litter was transported. By comparison, in a single year an average of 185,577 metric tons of conventional litter was transported in the first 5 years.

Degraded poultry litter would remain an economically viable option for all 50 of the years included in the analysis (Table 1). As a result, a noticeably larger quantity of poultry litter would be shipped if producers used degraded litter rather than conventional litter. Over the 50 year modeling scenario time span, degraded litter would ship 3.89 million metric tons of poultry litter, 19.8% more than conventional litter's movement of 3.66 million metric tons. Degraded litter would also continue to generate economically important benefits from year 25 to year 50 (Table 1). Degraded poultry litter would provide the largest impact, reaching $65.0 million over a 50 yr period. Conventional litter would generate an economic impact of $48.5 million, $16.5 million less than degraded litter, and corresponding to a difference of 34.0% compared to degraded litter (Table 1). Hence, conventional litter offers less economic potential than degraded litter since it is more costly to ship, ultimately limiting its ability to reach wayward points to the west.

4. Conclusions

For policy makers, Oklahoma's experience with animal waste management suggests the need for developing a program to recycle animal waste. This includes agricultural research to test the efficacy of animal waste products when field applied, as well as economics to assess its viability as a substitute for chemical fertilizers. According to the analysis presented in this paper, animal manure treatments provided a beneficial soil amendment for Oklahoma crops and forages with effects comparable to commercial fertilizer treatments. Furthermore, Oklahoma's animal producers should be able to transport animal waste products off-farm for a period of at least 25 years. Poultry producers are anticipated to encounter limitations on animal waste movements before swine or beef cattle producers. In Eastern Oklahoma, where the poultry industry is located, agronomic conditions are less favorable for the long-term application of phosphorus, and will more quickly reach state mandated limits on soil phosphorus concentration, due to the fact that litter contains more P compared to swine effluent and beef manure. In Western Oklahoma, soils generally possess less P and the locally available manure sources (swine and beef) contain less P than poultry litter, enabling long-term application of manure or effluent phosphorus with less potential buildup of soil phosphorus. Hence, swine producers and feedlot operators are anticipated to be able to transport animal waste off-farm for the entire 50 year period considered in the analysis of this chapter.

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10

Emerging Issues on Urban Mining in Automobile Recycling: Outlook on Resource Recycling in East Asia

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1. Introduction

The continued depletion of resources in today's consumerist society has put countries in East Asia to rethink and embrace recycling as a sound approach to sustainable consumption of resources. Recycling is not an "outside-of-the-box" approach but rather an "inside-of-the-box" method that has long been considered but was overtaken by the notion that resources are infinite. With societies' hunger for raw materials to feed into their industrial demands, production of new materials has become challenging and economically inefficient. In the automobile industry, the demand for rare earth metals is exponentially growing with increasing demand for vehicles. China, which is a top source of rare earth metals, has imposed limits on the export of such product due to increasing internal demand. Japan, which sources metals from China, has had to make innovative steps to address the problem through "urban mining." It refers to the recovery of rare earth metals from end-of-life (ELV) vehicles and electronic waste or e-waste (Saito & Yu, 2011). It does not require new materials to be extracted and it helps close the loop of sustainable resource circulation.

The concept of urban mining came to fruition with the advent of recycling technologies. Japan, which is a leader in the automobile industry, has devised ways to recover and recycle used metal parts from ELVs. With the large volume of new vehicles produced each year, it was imperative to reduce the volume of waste from old vehicles thrown into incineration plants and landfills. Used car parts are deemed valuable to be discarded in the midst of competing demand for rare earth metals all over the world. The way to move forward with a booming automobile industry is to incorporate environmental responsibility through recycling. The benefits cannot be overemphasized from the point of a resource-deficient

Japan. It is also paving the way for economic and social opportunities in the recycling sector. However, challenges abound in terms of supply and how urban mining could be sustainably done in developing countries where technologies are lacking. Likewise, exposure to toxic metals is high due to the manual nature of recovering used car parts.

Recovery and recycling per se are good but accompanying issues need to be addressed in developing countries so that real societal benefits are achieved. It is necessary that countries like Japan, Korea and China identify emerging lessons from the implementation of their respective ELV recycling laws so that developing countries can learn from them and craft laws that are appropriate and tailored to local needs and existing resources. This paper discusses the experiences of Japan and China in the field of urban mining and concomitant issues and challenges and how they will shape ELV recycling policies in East Asia. Outlook for the future will also be tackled as a way to create a road map for East Asia in the field of automobile recycling.

2. Urban mining opportunities and markets in Asia

The term "urban mining" was coined by Professor Nanjyo of Tohoku University in the 1980s to encourage and promote the reuse of precious and rare-earth metals found in used and discarded electronics. Japan, a heavy user of rare earth metals for its electronic and automobile industries, depends largely from China which produces 90% of the world's rare earth metals. The table below shows the top producers of rare earth based on 2009 data:

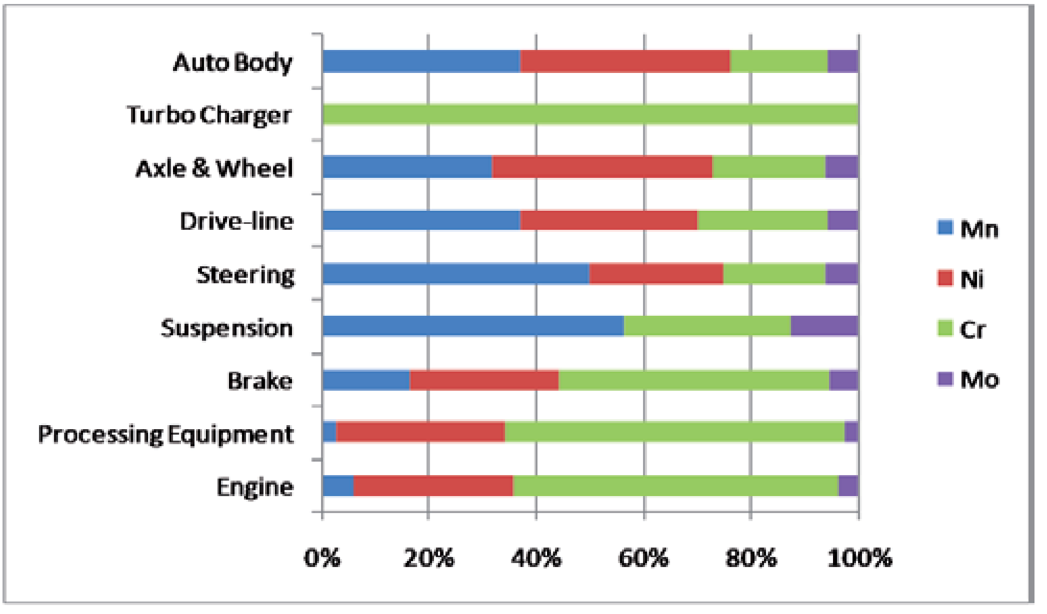
Country	Production (Metric Ton)	Reserves (Metric Ton)
United States	insignificant	13,000,000
Australia	insignificant	5,400,000
Brazil	650	48,000
China	120,000	36,000,000
Commonwealth of Independent States	Not available	19,000,000
India	2,700	3,100,000
Malaysia	380	30,000
Other countries	Not available	22,000,000
World total (rounded)	124,000	99,000,000

Source: http://geology.com/articles/rare-earth-elements/

Table 1. World Mine Production and Reserves (2009)

In July 2010, China announced a 72% reduction of exports due to increasing domestic consumption. This prompted the Japanese government to search for alternative sources and a research made by the National Institute of Material Science, a research organization affiliated with the Japanese government, announced that 6,800 tons of gold can be recovered from used electronics in Japan. This massive reserve is projected to be equivalent of 16% of the world's total reserves. Other reserves that can be generated are silver with 22%, tin with 11% and other materials at 5% (Kawakami, 2010). Clearly, the study showed the vast potential of internally sourcing rare earth metals in Japan rather than depending solely from foreign markets. It is sitting on mountains of used appliances and ELVs on its backyard where necessary resource inputs for new cars abound.

In ELVs, various metals can be found from different parts of a car. The following figure shows the distribution of rare earth metals from the exterior components of a car:



Source: Nagamura, 2010
*Mn – Manganese; Ni – Nickel; Cr – Chromium; Mo - Molybdenum

Fig. 1. Rare earth metals in auto parts

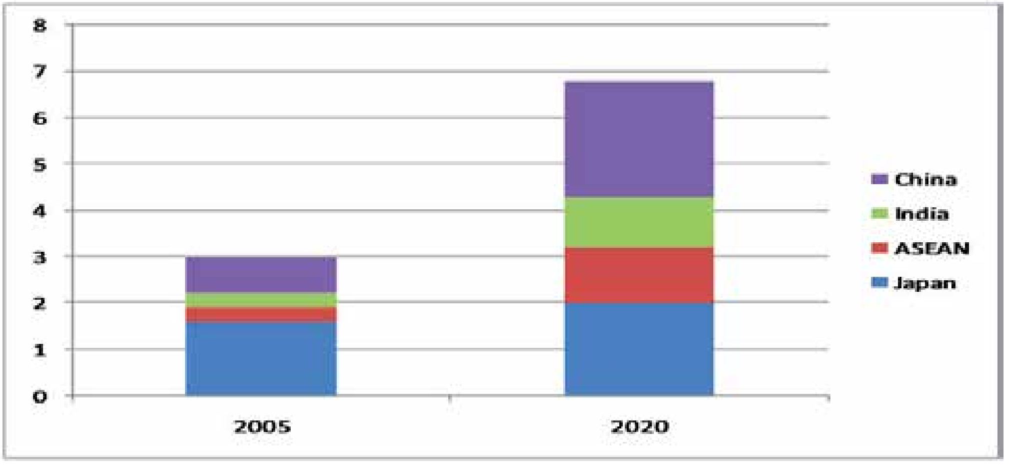
It will be noted from the above figure that Chromium has the highest concentration in engines and processing equipment while Manganese is abundant in suspension and steering parts. The rest of rare earth metals are spread in other auto parts. In terms of the price of earth metals, the following table shows the resource market fluctuation for iron ore, iron scrap, copper, silver and gold:

Metals ($/t)	Jan 2010	Jan 2011	Range of Elevation(%)
Iron ore	135	189.5	40.37
Iron scrap	313.5	462.5	47.53
Copper	7,065	9,585	35.67
Gold	1,084.80	1,340.70	23.59
Silver	1,621.20	2,791.90	72.21

Source: Asahi Newspaper, 2011

Table 2. Resource market fluctuation

The prices of valuable earth metals in the world market have significantly risen from 2010 to 2011 with silver achieving the highest increase. In terms of recycling market, the top three countries which have captured substantial markets for recycling are China, India and Japan as shown on the following:



Source: Ministry of Environment, Japan

Fig. 2. Recycling markets in Asia

3. Existing ELV legislations examined

The momentum towards ELV recycling was jumpstarted by the European Union (EU) with the passage of an ELV recycling law in 2000. Japan passed the "Law for the Recycling of End-of-Life Vehicles" in 2005. Korea legislated the "Act for Resource Recycling of Electrical and Electronic Equipment and Vehicles" in 2008. China, on the other hand, has "Statute 307" which was enforced in 2010. One of the salient features of this law is that vehicle producers of imported vehicles shall be responsible for the recovery and treatment of used vehicles (Serrona, Yu & Che, 2009). There are distinct variations in each of these laws but they all fall under the principle of "Extended Producer Responsibility" or EPR. Producers are largely responsible for recovery but consumers are also entrusted with certain responsibilities. It may be worth to comparatively revisit these laws as follows:

	European Union	**Japan**	**Korea**	**China**
Implementation year	2000	2005	2008	2010
Dismantlement method	Machine and Manual	Machine and Manual	Machine and Manual	Manual
Accountable entity and associated recycling costs	Manufacturer	End users	Manufacturer	Manufacturer
Operating principle	Market-based	Fund system (Air Bag, Freon gas & Automobile Shredder Residue or ASR)	Market-based	Market-based
Institutional mechanism	Member states	Japan Automobile Recycling Promotion Center	Korea Environment Corporation Eco Assurance System (ECOAS)	China National Resources Recycling Association

Table 3. Comparison of existing ELV laws

The above table reflects the uniqueness of Japan in terms of who is responsible in ELV recycling. The end users are the main actors as far as financial obligations are concerned such as payment of recycling fees. However, manufacturers are liable too like setting and publication of user fees and collection and disposal of shredder residues. Further, the above laws put both the manufacturers and users at the helm of recycling. This characteristic represents the necessary symbiosis that stakeholders play in resource recycling (Yu, Omura, & Yoshimura, 2008).

4. ELV Recycling in Japan and China

Japan has been implementing its ELV law since 2005. The vehicles covered by the law are four-wheeled passenger cars and commercial vehicles including mini-cars. The obligations of the car manufacturer comprise of collection and disposal of freon gas and airbags, collection and recycling of automobile shredder residue and setting and publication of user charges. Unlike other countries with ELV laws, financial responsibility is with the users where they are required to deposit a recycling fee at the time of sale. For old vehicles, deposit is required at the time of automobile inspection. The fees are managed by a fund management corporation (Kanari, Pineau, & Shallari, 2003).
Car recycling in Japan is not encompassing as it only covers three parts: airbag, freon and automobile shredder residue (ASR). As of the present, recycling rate in Japan is pegged at 95%. Table 1 shows the change in the generation of ELV and used car export of Japan for the period 2005 to 2008:

	2005	**2006**	**2007**	**2008**
ELV Generation	305	357	371	358
Cancelled registration of used car for export	107	144	161	130
Sales of used car	811	807	753	718

Source: Japan Ministry of Economy, Trade and Industry (METI)

Table 4. ELV generation and used car export figures (Unit: 10,000 cars)

ELV generation grew rapidly from 2005 up to 2007 and a decline was noted in 2008. This was due to the introduction of an ELV bounty system or subsidy. However, it is not only the recycling rate of used car parts that is important but also the reduction in the volume of ASR because of its harmful effects to human health and the environment, in general. Table 5 reflects the recycling rates for both ASR and airbag:

	Recycling Rate (%)	
	ASR	**Airbag**
Goal (%)	70% (until 2015)	85%
	50% (until 2010)	
	30% (until 2005)	
2008	72.4-80.5	94.1-94.9
2009	64.278.0	92.0-94.7

Source: Japan Ministry of Economy, Trade and Industry (METI)

Table 5. Recycling rates for ASR and airbag in Japan

ASR remains a challenge for Japan as recycling rate is still not catching up with say airbags. The goal in the next five years is to increase the rate from 50% to 70% in 2015. In summary, the flow of automobile recycling in Japan is shown in the following figure:

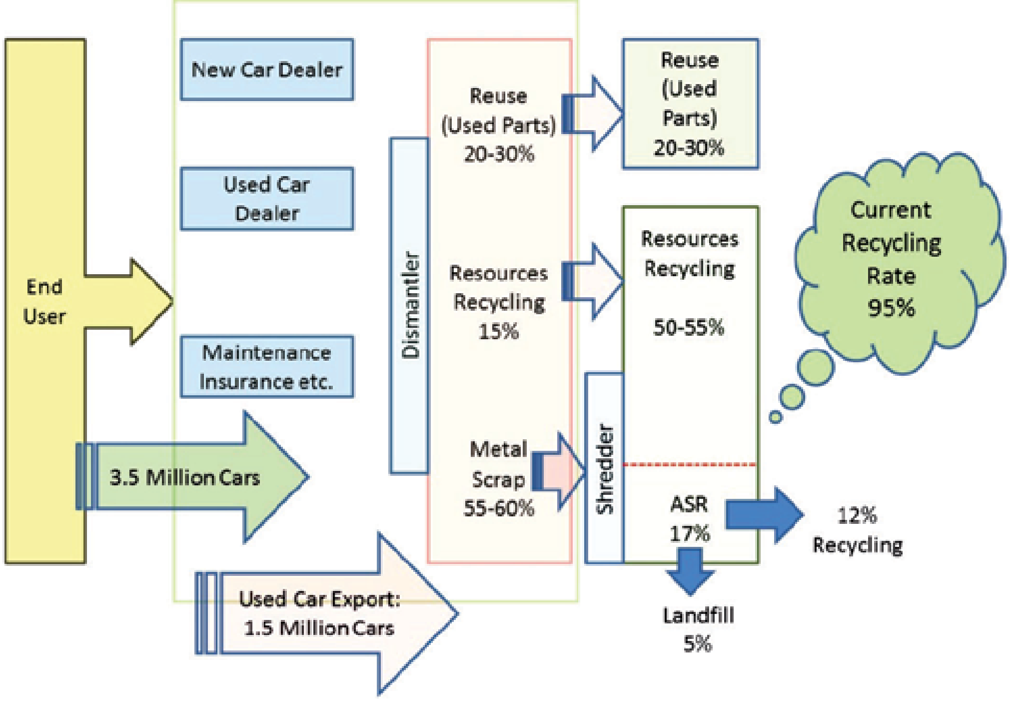


Fig. 3. Flow of automobile recycling in Japan

The above figure shows the efficiency of car recycling in Japan as used car parts are categorized. For example, reuse of used parts is about 20-30%, resource recycling is 50-55% while ASR recycling is 12%. Only five percent (5%) goes to the landfill (Yu, 2010).

4.1 Dismantling experiment in Japan

Rare earth metals are not the only valuable resource found in used cars. Plastic materials are also abundant. As part of the 3R Research and Development Project by Miyagi Prefecture in Japan, an experiment was conducted to demonstrate the time element involved in dismantling a used car with plastic as the main material recovered. Two types of used cars were dismantled: commercial car and luxury car. Methodologies were manual and machine dismantling (note that dismantling was done by non-experts). The following are the results:

Methodology	Time	Responsible
Manual dismantling	15 minutes	1 person (non-expert)
Machine dismantling	5 minutes	
Separation and collecting plastic	10 minutes	1 person (non-expert)

Table 6. Dismantling of a commercial vehicle

Overall, the recovered amount of plastic was 20 kilograms. The following picture shows an image of a commercial vehicle dismantled by a machine.

Fig. 4. Commercial vehicle to be dismantled for waste plastic recycling

Waste plastic recovered are shown below:

Fig. 5. Recovered plastics from a commercial vehicle

It was observed that it is very easy to retrieve plastic materials from a commercial vehicle because of the simplicity of its interior and a single type of plastic was used. In addition, commercial vehicles are designed where it is easy to dismantle and recover plastic materials. On the other hand, a luxury car was dismantled with the following results:

Methodology	Time	Responsible
Manual dismantling	40 minutes	2 persons (non-expert)
Separation and collection	10 minutes	2 persons (non-expert)

Table 7. Dismantling of a luxury vehicle

The recovered amount of waste plastic was only five (5) kilograms compared to the commercial vehicle which was 20 kilograms. The reason was that a luxury vehicle has complex interior and is made of composite plastic materials. Also, many adjoining materials are used which complicates the dismantling process and is time consuming as well.

Fig. 6. Luxury car dismantled for waste plastic recycling

Sample of recovered plastic materials is shown below:

Fig. 7. Waste plastic from a luxury car

A dismantling experiment was also made for a small sedan with amount of plastic materials recovered shown below:

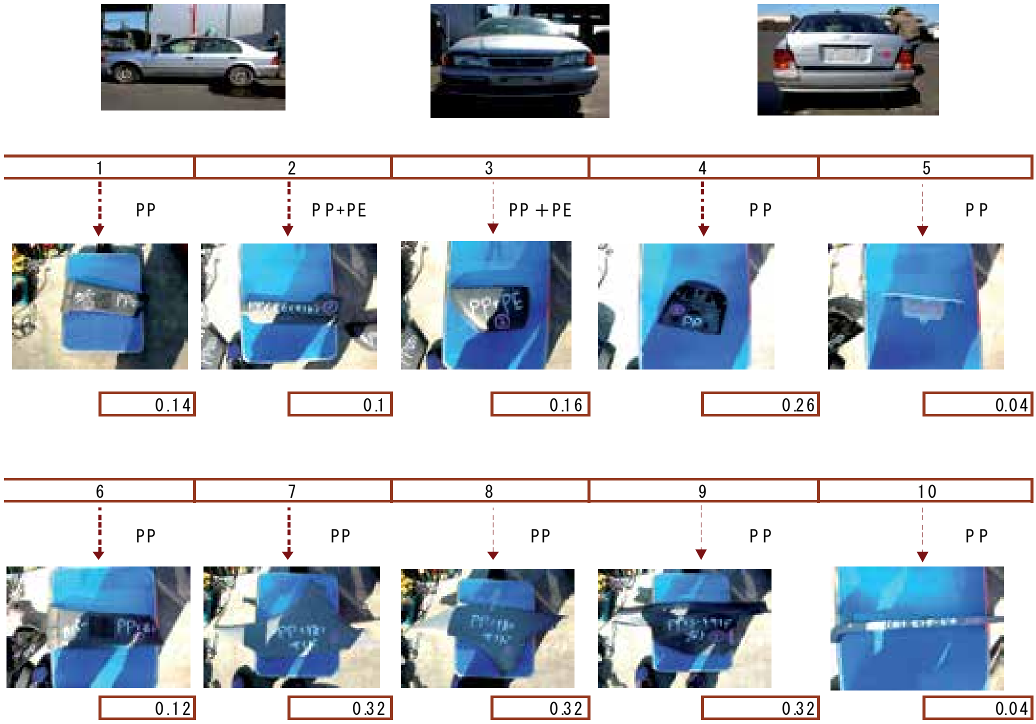


Fig. 8. Waste plastics recovered from a small sedan

In summary, the amount of plastics recovered are as follows:

	Polypropylene (PP)	PP+Polyethylene (PE)	PE	NA	Composition	Total (kg)
Weight (kg)	19.5	2.90	2.72	0.52	0.8	26.44

The downside of the experiment was the time it took to dismantle the vehicle. The total amount of time consumed was 2.5 hours by four (4) non-expert persons with a plastic recovery of only 19.5 kilograms. It was concluded that waste plastic recycling from a small sedan is not good considering the lengthy dismantling time and poor recovery efficiency. Sample of plastics recovered are:

On the left side are plastic pellets and on the right side is the internal part of a fender. In summary, the volume of plastics that can be recovered as well as the recovery efficiency depends on the type of vehicle. Figure 9 shows this type of variation.

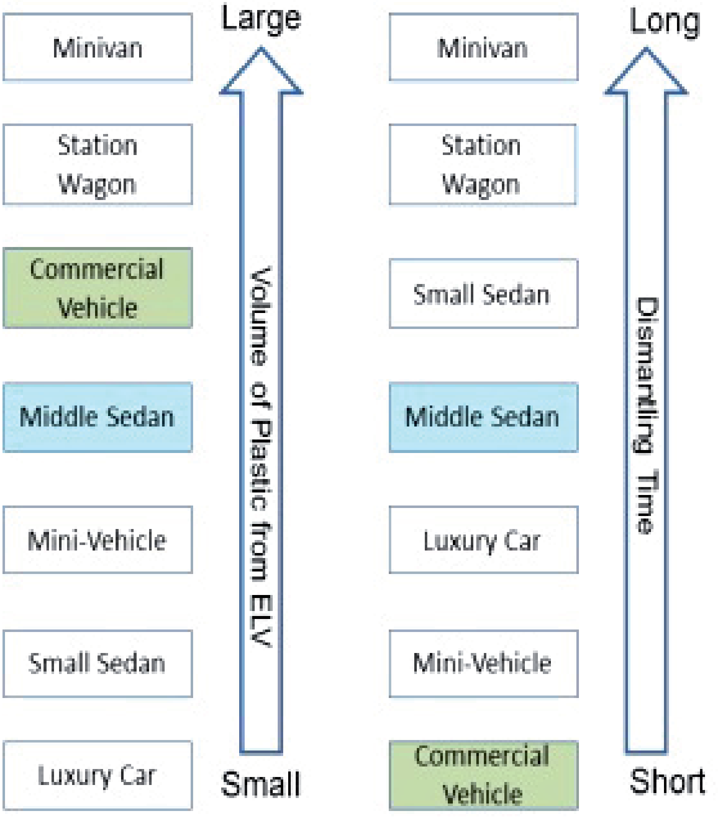


Fig. 9. Recovery efficiency for various types of vehicles

Based on the experiment, it can be concluded that for dismantling time, commercial vehicles are quick to be dismantled while minivans take time. In terms of volume of plastics recovered, relatively big cars like commercial vehicles, station wagons, and minivans have large amount of plastic materials.

4.2 Economic potential of ELV recycling in China

China's economy is booming at an unprecedented rate. Looking at car possession alone, the following table shows the rate for the period 2005-2008:

Year	Volume of car (million)
2008	49.75
2007	43.58
2006	36.97
2005	31.6

Table 8. Car possession rate in China (2005-08)

It is projected that the number of cars in China will increase a million per year in the near future due to increasing purchasing power and demand for personal transportation. From the data mentioned, it is also worth to examine the volume of cars that will become "used" in the near future. Using the following formulae:

$$E=A+B-C$$

Where:
E number of presumed used cars at current year
A number of car possessions in previous year
B number of sales at current year
C number of car possession at current year
Thus, the number of projected used car between 2005 and 2007 is as follows:

Year	**Projected used car (million)**	**Rate (%)**
2007	1.97	4.5
2006	1.79	4.8
2005	1.05	3.3

Table 9. Projected used cars in China (2005-2007)

It is, therefore, projected using the above data that China will have a large volume of used cars in the years to come.
There are about 10.33 million passenger cars in China in 2001 with an engine displacement of 1,600 cc or less. In several years, these will become used cars. A study made in Shanghai City and the City of Beijing showed that recovery percentage of ELVs in China is only 20%. To validate this, Tohoku University through the environmental research fund of Sumitomo Foundation conducted a dismantling experiment of a car commonly sold in China as shown below:

Fig. 10. Popular type of car sold in China

Considering the costs and difficulty involved in dismantling a new car, the research group selected a model closed to the picture as shown in Fig. 11.

Fig. 11. Replacement car used for dismantlement

The recovered materials are as follows:

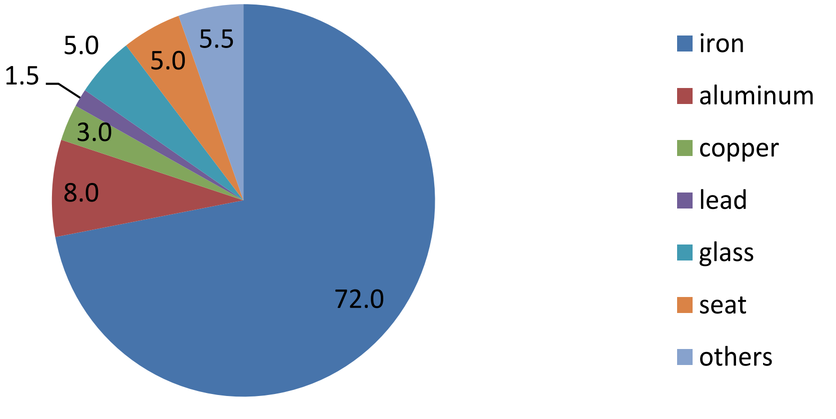


Fig. 12. Material composition of a passenger car

If only 20% are to be recovered from the 7.2 million passenger cars sold in 2009, the projected economic loss is about 200 billion Chinese Yuan based on the metallic market price in China. Thus, the need to find ways to increase recovery rate of ELVs is necessary in the country where car possession is increasing at a rapid rate (Che, 2011).

5. Monitoring system: A prerequisite in ELV recycling

The importance of establishing a monitoring system in ELV recycling is essential in so many aspects. ELV recycling is about collating information or data needed to make it more efficient and useful for key stakeholders like manufacturers, recyclers and even those who manually recover used car parts. In the case of EU, it ensures the inventory of ELVs while in Korea, every ELV is checked including weight, type and main parts like bumper, fuel cell, engine, exterior and interior parts and so on ("Yu, Omura, & Yoshimura, 2008). In the case of Japan, monitoring focuses on airbag, Freon gas and ASR as mentioned earlier. The actual benefits of having a monitoring system abound. It helps in increasing dismantling efficiency when each part is identified i.e. weight, type, time consumed during dismantling and the amount. Having these data will be useful for sharing purposes between the manufacturers and recyclers (Yu, 2010). The data could also be used in life cycle analysis (LCA) and cost-benefit analysis (CBA). Another benefit is the potential of determining who is responsible for what and what roles should relevant stakeholders play in the whole gamut of ELV recycling.

6. Emerging issues and challenges

Advance technology in used vehicle recovery and recycling does not necessarily apply in developing countries. They can recover resources using labor-intensive and manual strategies. Efficiency remains a question but in a research in China (Serrona, Yu & Che, 2010), a comparison between manual and machine-based dismantling was made. It was discovered that the former recovers more valuable parts which means more parts sold or recycled while the latter destroys more parts. In terms of value, manual dismantling provides more monetary compensation with more recovered parts. It also translates into more job opportunities and feeds people. The downside is that manual labor exposes people, engaged in the recovery of used car parts, to hazardous and toxic chemicals due to the absence of protective gears. It also consumes a lot of time. Manual recovery is good from the standpoint of local community participation. It allows them to be important stakeholders in the resource recycling ecosystem.

There is no blueprint for ELV recycling across countries. Each country has its distinct characteristics of stakeholders and geographical location. The formulation of recycling laws should be based on the unique features of communities rather than applying blanket legislations which may or may not be effective at all. To come up with sound laws is to have good documentation of vehicle database i.e. registration, type of vehicle and other ELV data. Policy makers should take these into consideration in order to arrive at legislations that incorporate locality-based conditions, needs and characteristics into national policies.

7. Conclusions and recommendations

Urban mining is driving ELV recycling into sustainable waste management. Just like solid waste management, various stakeholders or players are involved such as manufacturers, recyclers, users, waste reclaimers and the communities where waste recovery is done. There is a wide range of opportunities in this field and what is significant is that it has tremendous impact in terms of reducing toxic waste and providing local employment. In addition, it helps mitigate climate change by reducing greenhouse gases as recovering and recycling rare earth metals consume less energy than extracting raw metals. Closing the loop of rare earth metal utilization is a necessary attribute of sustainable consumption and production.

The experiences of Japan and Korea in this aspect are worthy of emulation by other countries in East Asia. Developing countries, being the destination of used cars, should formulate legislations that will address the collection, trading and disposal of ELVs which consider local conditions and capacities. There is a vibrant informal ELV waste sector in the Philippines, for example, but there is neither database nor monitoring system of what comes to the country as other ELVs are brought in illegally. The challenge lies in coming up with a legislation that strongly advocates safe and sustainable resource recovery. The role of Japan and Korea is to provide technical assistance based on their best practices. And being the leading manufacturers of vehicles in Asia, they should incorporate LCA in the production process so that the end users in developing countries are able to responsibly dispose used cars. Likewise, regulatory institutions will be able to promote the development of appropriate, labor-intensive and simple technologies to recover and recycle used metal parts. This kind of symbiosis will allow for a sound partnership between these countries through job promotion and people to people exchange of ideas. To strengthen ELV recycling regulations, the following are recommended:

1. Establishment of a resource recycling network that integrates economic and social dimension of ELV recycling. It should serve as a platform for exchanging innovative ideas, appropriate technologies and best practices in both developed and developing countries. There should also be component for capacity-building and community organizing to strengthen local waste reclaimers doing manual recovery. Organizing them into a legal entity will allow them to be accorded with rights, privileges and access to social and health services.
2. Build up a communication system between car manufacturers and recyclers. Currently, there is limited information exchange on ELV recycling processes. In the context of sustainable consumption and production, auto makers need to design cars that are easier to scrap and recover recyclable parts. Furthermore, they should provide the method of resource recovery including rare metals. And it is important to collect data on the content of materials as well as the cost and time involved in the recycling process.
3. In the formulation of ELV recycling legislations, lessons should be culled out from the best and bad practices of ELV recycling in Japan, Korea and even China. This includes an assessment of market principles used in the current system and how it will affect the future of resource recycling network worldwide. Policies should also be region-specific and localized to reflect in recognition of the distinct features of each locality or community.
4. The technological dimension of urban mining should be given attention with respect to local capacities and recovery efficiency. Based on dismantling experiments cited,

manual dismantling is found to be effective in terms of recovering more resource materials compared to machine-based recovery. It is not always necessary that machines are effective since manual recycling in developing countries generates local jobs and increases the value of recovered parts. However, such method should ensure adequate protection of workers from exposure to toxic materials.

5. Engage constructive policy dialogues with exporting countries like Japan and Korea to discuss ways to ensure that used car importation comes with the necessary recovery system in developing countries. This will ensure that developing economies are not inundated with pollutive used cars and ELV recycling is done.

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11

Phosphorus in Water Quality and Waste Management

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1. Introduction

Phosphorus (P) is a key element for all living systems. Phosphorus is a component of DNA and RNA and indispensable for the energetic metabolism (ADP/ATP) of living beings. Phosphorus cannot be substituted in these biological functions by any other element. The tremendous growth of global population is therefore linked to a proportional increase of phosphorus requirement for the production of food, which actually to a large extent is depending on the use of mineral phosphorus fertiliser.

Many natural (aquatic) ecosystems are controlled by restricted availability of phosphorus which represents one important factor for high biodiversity. The anthropogenic increase of phosphorus flows therefore has the potential to cause severe negative effects on natural (aquatic) ecosystems (see section 2).

Roughly 80 - 90% of the extracted phosphate rock is used for food production and nutrition. Given that P is a non-renewable resource and the global reserves are limited (contrary to nitrogen another essential nutrient) the aspects of scarcity and recycling/recovery have to be considered. Today's global mine production and reserves of phosphate rock (average P_2O_5 content is 31 % (P 13.5 %), ranges from 26 - 34 % (P 11 - 15 %) (Kratz et al, 2007; Steen, 1998) are reported ca. 160 Mio t/a and 16 billion tons, respectively (USGS, 2010). This gives a static lifetime for the reserves of some 120 years, a number which has been similarly reported by several authors before (Röhling, 2007; Wagner, 2005; Rosmarin, 2004; Pradt, 2003; Steen, 1998; Herring et al., 1993), others come to lifetimes up to hundreds of years (EFMA, 2000).

Phosphate ore is produced mostly from open pit mines, resulting in dust emissions and large quantities of tailings (mining wastes). Villabla and colleagues (2008) report material and energy consumption data for the production of 1 ton of P_2O_5 (Table 1). Another major waste is produced at a later stage when wet phosphoric acid (H_3PO_4) is produced from phosphate rock concentrate using sulphuric acid. This so-called phosphogypsum (ca. 5 tons per ton of wet H_3PO_4) is normally disposed of at sea or in large-scale settling ponds. It has very little use because it contains a considerable number of impurities such as Cadmium and radioactive elements (Villabla et al., 2008).

Attention: in historic literature phosphorus content of products or minerals is mostly expressed as P_2O_5 which corresponds to 0,44 P. Actually there is a trend to relate all data to P as element.

The above mentioned coverage times show that there is no urgent scarcity problem appearing at the horizon but the following aspects are worth to be considered: First, the relevant reserves of phosphate rock are highly geographically concentrated in China, Morocco & Western Sahara, South Africa and the U.S. The world's largest producers are China, followed by the U.S., Morocco and Russia (USGS, 2010). Large economies such as Western Europe or India have virtually no domestic supply and are dependent on imports. This is one major ingredient for a geopolitically instable situation. Second, the quality of extracted phosphate rock is continuously declining, meaning that the content of hazardous substances such as Cadmium and Uranium is rising (Kratz & Schnugg, 2006; Van Kauwenbergh, 1997; Steen, 1998). This will require, e.g., the employment of costly decadmiation processes in the future if agricultural soil shall be further on protected and not be used as a sink for heavy metals. Third, population growth, nutrient conditions of soils in developing countries and changing nutrition patterns (changeover to a meat- and dairy based diet) will entail increased demand of fertilisers. The Food and Agriculture Organisation predicts annual growth rates between 0.7 to 1.3 % until 2030 (FAO, 2000) which would mean an increase of some 25 % in phosphate rock consumption compared to now. The Population Division of the Department of Economic and Social Affairs of the United Nations Secretariat predict 9.2 billion people in 2050 (+37 %) (UN, 2008). Considering these facts similar market effects and price volatility as currently are the case for crude oil have to be anticipated for phosphorous fertilisers in the future (Fig. 1).

Production Stage	Input	Output
Mining	Electricity 697 MJ	Waste 21.8 tons
	Diesel 125 MJ	Mine water
	Explosives 3,3 MJ	Diesel exhaust gases
Mineral processing	Water	Waste water
	Electricity 1,128 MJ Flotation reagents	Tailings 6.5 tons
	Diesel 396 MJ	Diesel exhaust gases
Total primary, energy consumption		ca. 5,500 MJ
Total solid waste		ca. 28 tons

Table 1. Material and energy consumption for the production of 1 ton P_2O_5 (= 0.44 t P), adapted from Villabla et al. (2008).

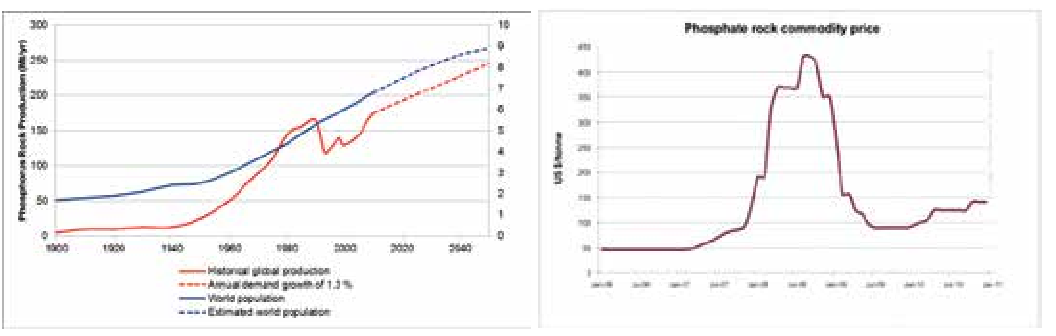


Fig. 1. a) Phosphate rock production and world population (historic situation and future trends); b) Phosphate rock commodity price 2006 – 2010

Mankind's usage of P is rather non-circular and dissipative. Bacinni & Brunner (1991) established a P-balance for the 1980ies where it is shown that the use of P is inefficient (only 10% of P employed in agriculture is contained in the food) and large losses occur to the soil and the hydrosphere. These findings still hold and today there is consensus that effective phosphorous management in agriculture and waste water is of upmost importance. Considering the above mentioned today's goals for P management in a region could be: Keeping soil fertility under minimized P fertilization, reducing surface emissions to the hydrosphere, reducing point emissions to the hydrosphere and increasing recycling.

In countries with centralised waste water infrastructure sewage sludge is a by-product a result of fulfilling a legal requirement for water protection by waste water treatment. Sewage sludge represents a relevant source for P-recycling, especially if P-removal from waste water is applied (UWWD, 1996) (see section 1.2). For example, Austria imported 31.000 t P/a in form of phosphate rock, fertilisers, food, and fodder in 2001 (Seyhan, 2006). The actual P content of sewage sludge in Austria is ca. 7,000 t P/a, which corresponds to a P-removal efficiency of ~ 85 % from waste water.

Sewage sludge is a waste according to actual legislation, as long as it cannot be marketed as a product (e.g. to phosphate industry substituting phosphate rock). The common practise to recycle the P-content of sewage sludge is its application in agriculture as fertiliser and soil conditioner. As sewage sludge also contains potentially hazardous substances (pathogens, heavy metals and organic micro-pollutants) this recycling path is subject to various legal constraints (US/EPA Biosolids Rule and NPDS, EU Sludge Directive 86/278, numerous national and regional sludge regulations, soil protection regulations) and is still a matter of controversial discussion even in the scientific community. (Engelhart et al., 2000; Brunner et al., 1988; Giger et al., 1984; Giam et al., 1984; Sjöström et al., 2008; Sandaa et al., 2001; Ghaudri et al., 2000; Giller et al., 1998)

In US and UK the main risk associated with sludge recycling is related to hygiene (pathogens) and there is a clear distinction between sludge and biosolids, the later being a market product meeting certain quality criteria. In parts of Europe the main risk is associated with soil and food protection (heavy metals and micro-pollutants). Beyond agriculture and waste water treatment plant operators and also food industry, food chain traders and retailers and numerous NGOs are stakeholders. Also the competition between sludge as P-fertiliser and manure application in countries/regions with extensive animal production has to be considered in this context.

Sludge application on land has strongly supported the reduction of emissions from point sources which was very successful for heavy metals where the concentrations in sewage sludge considerably decreased during past decades. Regarding the source control of micro-pollutants the discussion has started but as these substances to a large extent originate from market products the situation is much more complex than for heavy metals. Sewage sludge not meeting the actual quality criteria for land application or sludge produced in countries/regions where land application is banned (e.g. in Switzerland) have to be disposed of. Where landfilling of organic material, as sludge, is not allowed any more incineration has become a viable treatment option as ashes meet the criteria for landfilling.

At present sludges are incinerated in so-called mono-incinerators or co-incinerated in cement kilns, coal-fired power plants and waste incineration plants. Incineration destroys the organic sludge fraction including the micro-pollutants. Phosphorus and also most of the heavy metals are contained in the ashes. A favourable condition for P-recovery is only with mono-incineration (maybe co-incineration with P-rich waste fractions) because only then P-rich ashes are produced which can be put to special storage sites for future recycling of P.

2. Phosphorus in water quality management

Phosphorus is the limiting nutrient for algae (autotrophic) growth in most fresh water bodies (lakes, rivers and reservoirs) and some coastal waters influenced by river discharges. The anthropogenic discharge of phosphorus to these waters therefore increases the potential for algae growth, which is the starting point of eutrophication. Eutrophication is characterised by increased availability of phosphorus for primary production (algae) which represents the basic substrate for the aquatic ecosystems. Even moderate anthropogenic increase of P availability influences the competition between the species which results in changes of the aquatic ecosystem.. Beyond certain thresholds of phosphorus discharge the ecosystems shift to a completely different status with steadily increasing deterioration of water quality.

In natural environments phosphorus is mainly present in particulate form as minerals with low solubility. The availability of phosphorus for plants and algae is therefore quite restricted. Hence in many natural aquatic ecosystems life is limited by phosphorus deficiency. The decay of the organic material produced by photosynthesis under aerobic conditions again results mainly in mineral phosphorus compounds in the sediments with low availability. Under anaerobic conditions decomposition process results in the release of phosphorus in dissolved and therefore easily accessible form.

Natural environments normally are characterised by restricted dissolved phosphorus flows. Soil erosion results in relevant phosphorus loads depending on the P-content of the particulate material either coming from natural rocks and soils or from agricultural land. As long as this material remains under aerobic conditions in the waters and their sediments, availability of phosphorus for algae growth is low. The main anthropogenic sources of phosphorus, except erosion, for the aquatic environment are (Lee et al., 1978):

- municipal and industrial waste waters
- drainage from agricultural land
- excreta from livestock
- diffuse urban drainage

Humans need a daily phosphorus uptake of 1.8 - 2 g P via their nutrition and discharge it with their excreta to the waste water. Most of this phosphorus is easily accessible for plants, in waters for algae and macrophytes. Waste water therefore has a great potential for enhancing eutrophication if not properly handled before discharge to the aquatic environment. Agriculture uses either mineral phosphorus fertilisers or manure to meet the phosphorus requirement for crop production. If correctly applied phosphorus is either taken up by the crops or retained in the topsoil. Only soils with low quality tend to release phosphorus compounds to the ground water, as e.g. at reclaimed agricultural areas from wetlands. For the assessment of the effect of phosphorus discharges to waters it is always relevant to investigate the availability of the phosphorus loads (particulate versus dissolved) and that this strongly depends on the redox conditions in the sediments.

2.1 Source of P in waste water

The main sources of phosphorus in waste water are the human excreta, phosphorus containing household detergents and some industrial and trade effluents. Precipitation runoff only little contributes to P-loads in waste water if combined sewer systems are applied. Figure 2 shows the input and output loads of households in Austria, where P-free laundry detergents but phosphorus containing dish-wash detergents are used.

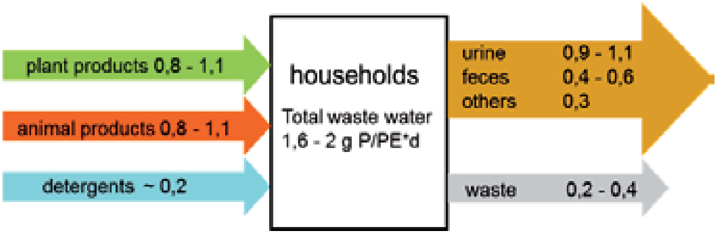


Fig. 2. Recent phosphorus loads from households (Zessner & Aichinger, 2003)

Industrial use of phosphorus is quite limited; relevant phosphorus loads in industrial effluents therefore are relatively low. P-discharges to waste water can origin from food and textile industry and from rendering plants. Industrial and trade contribution to P-loads in municipal waste water can vary in a broad range depending on local situations and is in a range of 10 to 40 %.

Historical development

In the 1970 and 80's the daily phosphorus load per inhabitant in municipal waste water was up to 5 g, the main source being the phosphorus containing laundry detergents. Due to the resulting eutrophication problems the removal of P-containing detergents from the market was achieved which resulted in the actual P-loads in municipal waste water of 1.4 - 1.6 g P/PE/d (1PE = 60 g BOD_5/d, or 120 g COD/d). Dish wash detergents still contain phosphorus and contribute to about 10% of the P-loads (ATV, 1997; ATV DVWK, 2004). This development makes P-elimination from waste water at treatment plants more economically and ecologically advantageous. Application of P containing dishwashing detergents is still increasing and therefore the development and application of P-free dish wash detergents are aimed at.

While phosphorus can be or has to be removed from municipal waste water, the biological treatment of industrial waste water often requires the addition of phosphorus As bacteria are able to store phosphorus the control of P-dosing is not easy. Low phosphorus availability favours the growth of filamentous bacteria (at activated sludge treatment plants), which results in bad settling and thickening properties of the sludge. Impairment of thickening properties of the sludge can be used as a sensible indicator of phosphorus deficiency. Severe P-deficiency strongly affects treatment efficiency. An excess dosing of phosphorus can result first in filling the P-storage capacity of the microorganisms and therefore will not immediately be detected by rising P-concentrations in the effluent. (Prendl et at., 2000; Nowak et al., 2000)

2.2 Phosphorus elimination

In most of the surface waters, availability of phosphorus limits the growth of algae and macrophytes. The discharge of P-loads contained in waste waters therefore normally results in an increased growth of algae and macrophytes which may cause reduced water quality by eutrophication. This is especially relevant for lakes and estuaries ("Sensitive Areas") but also can affect the quality of rivers. According to EU Urban Waste Water Directive (91/271/EEC) phosphorus needs to be eliminated from the waste water at treatment plants in sensitive areas.

The EU effluent standards for sensitive areas for total phosphorus (TP) are 2 mg/L for WWTP 10.000 - 100.000 PE and 1 mg/L for more than 100.000 PE or a minimum TP load reduction of 80 %. National standards can be even more restrictive especially for catchments of lakes or very sensitive coastal regions. Where environmental standards cannot be met by applying only the minimum requirements (effluent standards according to precautionary principle) much higher P-removal requirements can be prescribed for specific WWTPs (e.g. MURTHY, Chesapeake Bay, Blue Plains WWTP). Figure 3 shows a single step waste water treatment plant with its typical mechanical, biological and sludge treatment processes.

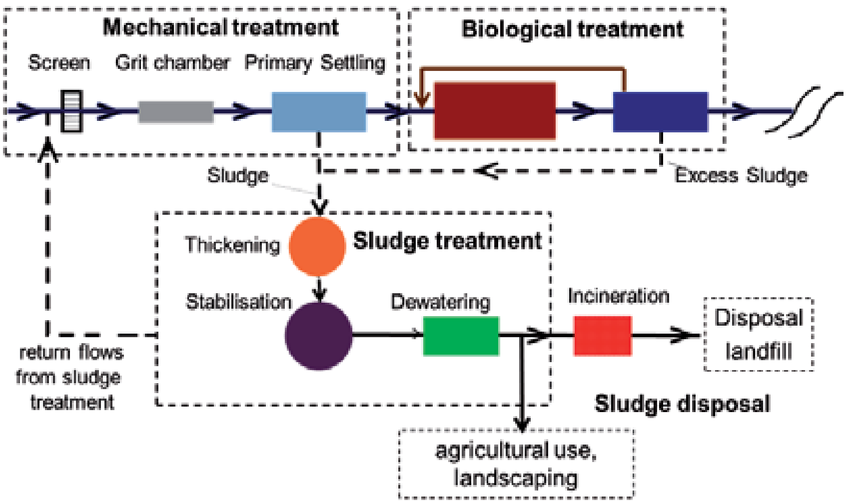


Fig. 3. Schematic of a conventional activated sludge treatment plant

2.2.1 Mechanical treatment

During mechanical treatment phosphorus contained in the particulate material is removed from the waste water together with primary sludge which results in a TP removal of 10 - 15 %. Biological incorporation, enhanced biological P-removal and chemical precipitation are state-of-the-art processes to reliably reduce P-load from waste water. The total phosphorus loads removed from the waste water in most of the processes applied in practice end up in the sludge. In principle the P-content of the sludge can be recovered and reused which is of increasing relevance for the long term availability of this limited resource.

2.2.2 Chemical-physical P-elimination

The most reliable and most frequently applied removal process is chemical phosphorus precipitation by addition of metal salts. Dissolved phosphorus is converted to solids which are removed from the waste water together with the sludge. If very low effluent concentrations < 0.5 mg TP/l) have to be achieved secondary effluents can be treated by flocculation filtration.

P-removal by precipitation is based on five processes (ATV DVWK, 2004):

1. Dosing: complete mixing of precipitants (metal cations: Fe^{3+}, Al^{3+}, Ca^{2+}) to waste water stream
2. Precipitant reaction: Formation of particular compounds as precipitant cations, phosphate anions and other anions.

3. Coagulation: destabilization of colloids in waste water and coagulation to micro-flocs
4. Co-precipitation and flocculation: Formation of separable macro-flocs. Inclusion of particulate matter, colloids as well as organic bounded phosphorus in these flocs
5. Separation: By sedimentation, filtration, flotation and a combination of these processes, macro-flocs will be separated.

The elimination of phosphorus is based on the precipitation of the negative charged dissolved phosphate (PO_4^{3-}) by a trivalent metal ion. Sources of these metals are ferric and ferrous chloride, ferrous chloride sulphate, ferrous sulphate, aluminium sulphate sodium aluminate. Except the last one which is alkaline all the other precipitants are acid. Phosphate compounds as $FePO_4$ and $AlPO_4$ with very low solubility product will be formed ($pK_L \sim 22$). Iron and aluminium have nearly the same effect, but the optimum pH for iron is about one unit lower than for aluminium. Iron-flocs have a higher density are more compact and more shear resistant than aluminium-flocs. These properties influence the separation process especially if pre- and post-precipitation processes are applied. With both metal ions PO_4-P effluent concentrations ≤ 0.1 mg/l can be achieved if dosage is sufficiently high.

$$Me^{3+} + PO_4^{3-} \rightarrow MePO_4 \tag{1}$$

Theoretically one mole of Fe or Al is needed to precipitate one mole of P. Due to the different atomic weight of the atoms, the appropriate mass dosage needs to be calculated based on the molar weights (1 Mol P: 31 g; 1 Mol Fe: 56 g; 1 Mol Al: 27 g). The specific precipitant dosage (β-value) is the molar ratio between precipitant and phosphorus to be precipitated as e.g. described above for simultaneous precipitation.

$$\beta = \frac{mol\ phosphorus}{mol\ metal} \tag{2}$$

But iron- and aluminium-ions also react with other compounds therefore more precipitants have to be added than theoretically necessary. With a chemical addition corresponding to a β-value of 1.5 an effluent PO4-P concentration below 0.5 mg/l (TP < 1 mg/l) can be achieved with simultaneous precipitation at activated sludge plants. Figure 4 is based on TP effluent data of full scale municipal treatment plants. It shows the relation between TP effluent concentrations and the β-factor applied for different P-removal techniques. As mentioned above satisfying P-effluent concentrations need a higher dosing of metal salts than derived from stoichiometry only. To precipitate 1 kg P theoretically 1.8 kg of iron (56/31) and 0.9 kg (27/31) aluminium are necessary (β-value of 1). For simultaneous precipitation it has to be considered that part of the phosphorus will be incorporated into the sludge and therefore needs not to be precipitated. For rough calculations it can be assumed that this incorporated phosphorus at least corresponds to ~ 1 % of the BOD_5- load in the influent to the biological treatment (e.g. 0.6 g P/PE_{60}). If enhanced biological P-removal occurs the precipitant requirement can only be derived from the operational experience on site based on effluent monitoring.

Crystallisation processes

By precipitation with calcium cations manifold reactions are known, which are hard to predict. High P-removal efficiency can be achieved at pH controlled crystallisation of calcium hydroxyapatite which has a very low solubility product ($pK_L \sim 59$). PO_4-P concentrations below 0.1 mg/l can be achieved depending on the dosage.

$$5Ca^{2+} + 3PO_4^{3-} + OH^- \rightarrow Ca_5(PO_4)_3OH \tag{3}$$

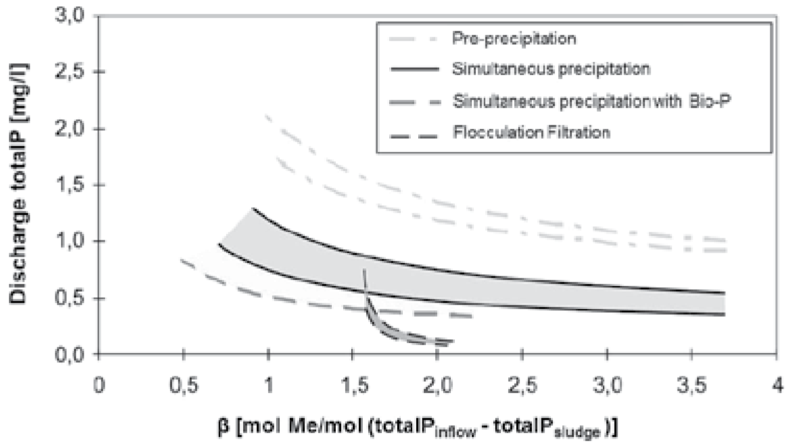


Fig. 4. P-discharge in relation to the β-factor (Nikolavcic et al., 1998)

Phosphate reacts also with magnesium and ammonium forming magnesium-ammonium-phosphate (MAP, struvite), a precipitation product with low solubility (Schulze-Rettmer, 1991). The precipitation (crystallisation) process is strongly dependent on pH. All the 3 components are present in waste water in low concentrations while in the sludge liquors after anaerobic digestion even in high concentrations. Uncontrolled MAP-precipitation can cause operational problems by incrustations of pipes and machine parts at sludge treatment plants. Normally the precipitation process is limited by the (low) Mg concentrations. Efficient MAP precipitation can be achieved in a controlled process by dosing Mg salts (see side stream and crystallisation processes). Depending on the location of the addition of the precipitants at waste water treatment plants the following techniques can be distinguished (ATV-DVWK 2004):

- Main stream processes:
  - Pre- precipitation (1),
  - Simultaneous precipitation (2),
  - Post precipitation with flocculation and sedimentation (3) and
  - Post precipitation with flocculation filtration (4)
- Side stream processes
  - Sludge liquors

Pre- precipitation (1)

The metal salts are added to the influent of grit chambers or primary clarifiers. The precipitation product can be separated together with the primary sludge. P-precipitation results also in additional removal of organic suspended solids which has to be considered for design and operation of nitrifying and denitrifying treatment plants. Pre-precipitation has to be continuously controlled according to the influent P-load and for a following biological treatment P deficiency must be avoid. Especially nitrifying bacteria are sensible to P-deficiency.

Simultaneous precipitation (2)

This is by far the most frequently applied process in full scale. Precipitants are added to the influent of the aeration tank, directly into the aeration tank, to the inflow of the secondary clarifier or to the return sludge. If enhanced biological P-removal is aimed at, the

precipitants are preferably added to the effluent of the aeration tank in order to avoid competition between biological and chemical P-removal. In such a case divalent iron salts (e.g. ferrous sulphate) cannot be used because they have to be first transformed to trivalent iron by oxidation. Divalent salts should only be added to the aerated volume of the tanks. The precipitation products are removed together with the excess sludge. The increase of the inorganic fraction of the excess sludge has to be considered for the design but this has only a low relevance if P-free detergents are applied in the catchment. With simultaneous precipitation P-effluent concentrations of ≤ 0.5 mg TP/l can be achieved if the dosage of precipitants is adequate (β-value see capita 2.2.2). At the dosing points good mixing conditions have to be ensured (high turbulence).
The access of iron salt is precipitated as $Fe(OH)_3$ and stored in the activated sludge. At higher phosphorus loads this iron storage is able to precipitate surplus phosphorus. Thus control of the dosage is much simpler than at pre or post precipitation.

Post precipitation with sedimentation (3)

This P-removal process normally is only applied as a polishing step after biological treatment with P-removal in order to achieve TP concentrations below 0.5 mg/l. All these processes have to concentrate on the retention of particulate material as bacterial flocs and fine precipitation products containing phosphorus. The dissolved P fraction in the treated effluent depends on the chemicals used, the dosage related to the P-load and the environmental conditions (pH, T). For particle separation flocculants are added which allows removing the precipitation products by sedimentation. In order to improve the removal efficiency sludge can be recycled to the flocculator where gentle mixing is applied.

Post precipitation with flocculation filtration (4)

The only important differences to the process described above consist in particle separation after flocculation by rapid sand filtration and in the future probably by membrane filtration.

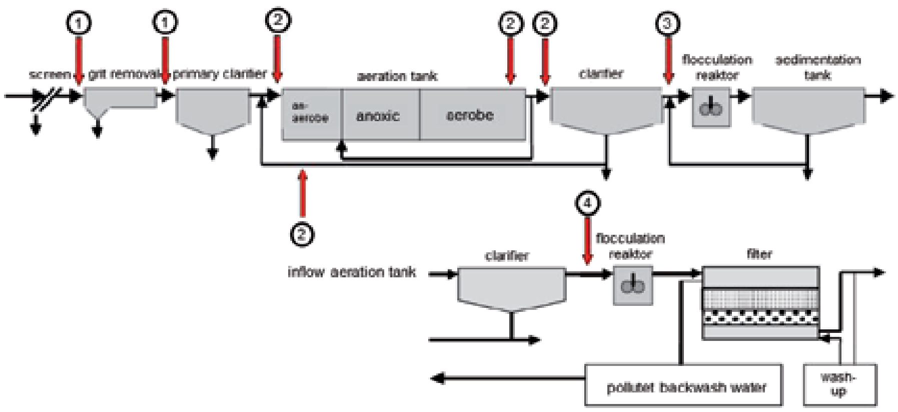


Fig. 5. Physical- chemical P-elimination

Effluent TP concentrations ≤ 0.1 mg/l can be achieved. It has to be stressed that the specific cost for the post precipitation processes (€/kg P removed) are much higher than for the in-stream processes described earlier.
For all the processes the effluent concentrations of dissolved PO_4-P are depending on type of chemicals used, β-value applied, pH and temperature. The percental removal efficiency strongly depends on the P-influent concentration. Applying the same β-value the removal efficiency will decrease with decreasing influent concentrations. For the TP effluent

concentration particle removal efficiency and the fraction of complex phosphorus compounds which cannot be precipitated (e.g. phosphonic acids) in the influent is decisive.

2.2.3 Biological phosphorus removal processes

P-removal by normal-uptake of bacteria

Microorganisms need phosphorus for their growth i.e. the excess sludge production. P-removal from the waste water therefore is a necessary side effect of conventional biological waste water treatment, depending on the specific sludge production per unit of organic pollution (e.g. population equivalent of 120g COD/(PE*d). An average P-removal of 0.6 to 1 g P/(PE*d) can be achieved with bacterial growth only. Microorganisms are able to store phosphorus in order to survive periods with phosphorus deficiency. This is not relevant in municipal waste water treatment plants, where there is a continuous excess of P in the waste water. At plants treating P-deficient industrial waste waters this ability of bacteria has to be considered for an adequate control strategy for the P-dosage (Svardal, 1998).

P-removal by luxury-uptake of bacteria

Luxury uptake is performed by phosphorus accumulating organisms (PAO) able to store phosphorus up to ≥ 5 % of their dry weight if process configurations are applied which increase the competitiveness and survival probability of PAOs in biological treatment plants. The main characteristic of this process is to subject the microorganisms to alternating anaerobic and aerobic (anoxic) conditions. With this process phosphate storing microorganisms are enriched in the system (Barnard, 1975; Ludzak, 1972; Nicolls, 1972; Levin, 1965).

Process description:

Anaerobic conditions are characterized by the absence of dissolved oxygen (DO = 0 mg/l) and oxidized forms of nitrogen (nitrate and nitrite) and the presence of biodegradable material causing an oxygen demand. Under anaerobic conditions PAOs are not able to grow but can accumulate and store organic substrate by converting organic acids to poly-hydroxy butyrate (PHB) and similar energy rich organic compounds. For this process the bacteria need energy which they gain under anaerobic conditions from the conversion of stored energy rich poly-phosphate to dissolved phosphate which is released to the water under these conditions.

The substrate storage process under anaerobic conditions is controlled by the energy stored in the bacteria in the form of polyphosphates. The "poly-phosphate energy battery" is recharged under anoxic or/and aerobic conditions where PAOs use part of the energy gained from the aerobic degradation of the organic carbonaceous pollution.

The excess sludge withdrawn from or after the aerobic (anoxic) zone contains the stored polyphosphate load which increases the P-removal from the waste water via the excess sludge. Dry solids of conventional activated sludge have a TP contents of 1 - 1.5 %, while those of enhanced biological P removal plants can achieve up to > 4 % TP.

As substrate availability in the aeration tank is low the growth rate of the bacteria is low due to substrate limitation (low effluent concentration). Under such conditions PAOs gain an increased competitiveness as they can additionally grow using the stored accumulated substrate. The best indicator for P luxury uptake in an activated sludge plant is the increase of the dissolved phosphate load in the anaerobic tank (figure 3).

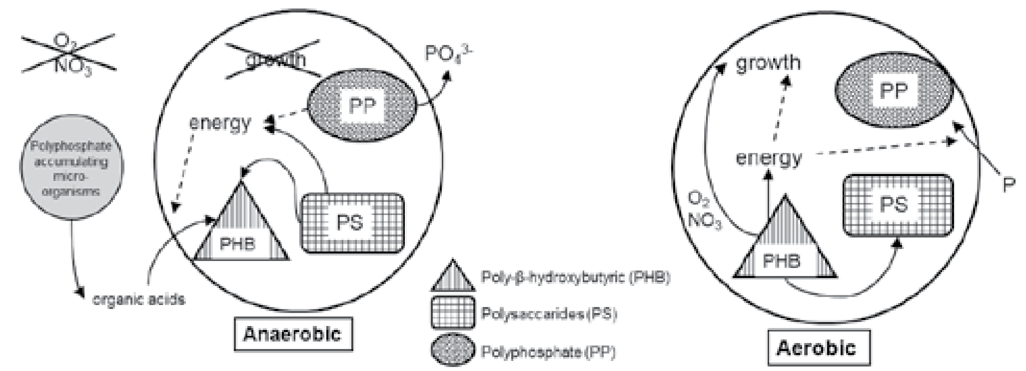


Fig. 6. Metabolism processes at luxury uptake (Henze, 2008)

Conclusion:

Enhanced bio-P removal (luxury uptake) from waste water is advantageous as it does not require chemical addition. The reliability of this process strongly depends on specific local conditions which can partly be compensated by favourable process configurations and the detention time in the anaerobic tank volume. Enhanced bio-P removal can easily be combined with the very reliable P removal by chemical precipitation. Optimization of design and operation can therefore be achieved on the basis of economic and operational considerations.
As the temperatures of the waste water strongly vary over the year it is reasonable to make optimal use of the total aeration tank volume over the whole year. E.g.: at high temperatures the anaerobic tank is used for Bio-P removal while during the lowest temperatures this tank can also be used for denitrification, while P-removal is maintained by chemical precipitation.
An important consequence of bio P removal is that the excess sludge must not be subject to anaerobic conditions during thickening. If primary and secondary sludge are thickened together, this would l result in a release of P-luxury uptake to the supernatant which returns it to the influent. Separate mechanical thickening of the excess sludge is the most common solution to avoid this effect.
During one step anaerobic sludge digestion P-release normally remains neglectable as long as the organic acid concentration is kept low, which is the case at sludge retention times > 20 days and quasi steady state conditions. Phosphorus remains bound to the solids which is favourable for sludge application in agriculture or incineration with P-recovery from the ashes.
The application of a two-step digestion process (hydrolysis + methanisation) at Bio-P removal plants would result in the release of the luxury P uptake to the supernatant (sludge liquor). In this case P crystallisation processes can be applied to the sludge liquor in order to convert the released P to a market fertiliser (Triple-phosphate) with low contamination by heavy metals and micro-pollutants. The normal P content of bacteria and the chemical precipitated phosphorus remain bound to the sludge solids. Whether such a process configuration is economically viable has to be proved in practice (Pinnekamp, 2007).

Enhanced bio-P removal technologies

There are numerous processes for enhanced biological phosphorus removal. All of them contain an anaerobic tank (cascade) and try to minimize the negative impact of oxygen and oxidized nitrogen compounds to the process. It is possible to distinguish in stream and side stream processes. All in-stream processes remove phosphorus together with the excess sludge. Side-stream processes force the release of phosphate to the liquid phase of the return sludge or to the sludge liquors from where phosphate is precipitated (crystallised) by chemical addition. The following simplified process configurations are used in practice:

In-Stream Processes

The following in-stream processes are described by Pinnekamp (2007), Kunst (1991) and Matsché (1989).

Bardenpho (Barnard, 1974)

The Bardenpho-process consist of the anaerobic tank in front, a pre-denitrification/ nitrification step with internal recirculation followed by an anoxic tank and a final aeration step in order to avoid nitrate transfer to the anaerobic tank by the return sludge. This process achieves high P-elimination and is operated with very low loadings. Numerous WWTP of this type operate in South Africa and North America.

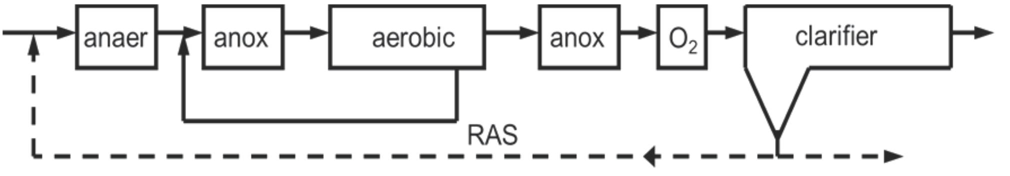


Fig. 7. Bardenpho-Process

UCT (University of Cape Town (Rabinowitz and Marais, 1980))

Because of remaining nitrate in the return sludge at the Bardenpho- and Phoredox-process release of phosphorus in the anaerobic tank can be affected. The UCT-Process was developed at the University of Capetown in order to avoid this. The main difference to the Bardenpho process is that the return sludge is fed to the anoxic pre-denitrification tank from where the denitrified activated sludge is returned to the anaerobic tank.

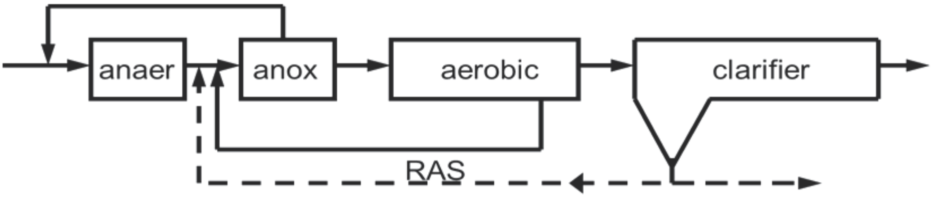


Fig. 8. UCT-Process

Phoredox (Barnard, 1976)

The Phoredox-process is a simplification of the Bardenpho-process. Because of the low reduction rate in the second anoxic tank and the aeration tank these steps were omitted.

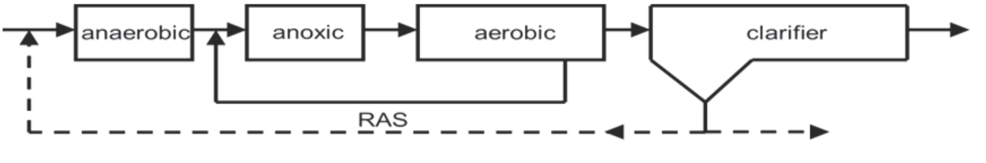


Fig. 9. Phoredox-Process

Johannesburg (JHB)-Process (Burke et al., 1986)

This process is a modification of the Phoredox-process. Return sludge will be denitrified in an anoxic tank before reaching the anaerobic step. Therefore the input of the anaerobic tank is the inflow and nitrate free return sludge. The name of this process is based on its location in Johannesburg.

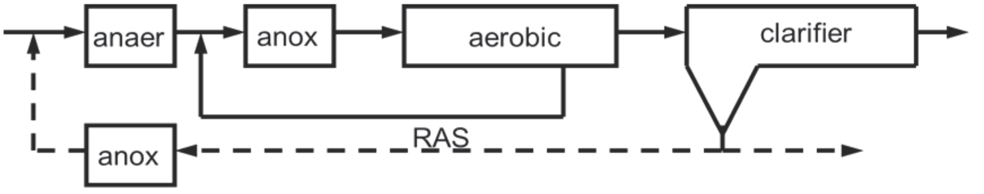


Fig. 10. Johannesburg-Process

A²/O (Barnard, 1974)

The A²/O is a modification of the Phoredox-process with nitrification and denitrification. Compared to the Phoredox-process the tanks are constructed as cascades, while the order of the tanks is identical (Figure 11). This process is operated in the US and Brazil in some plants.

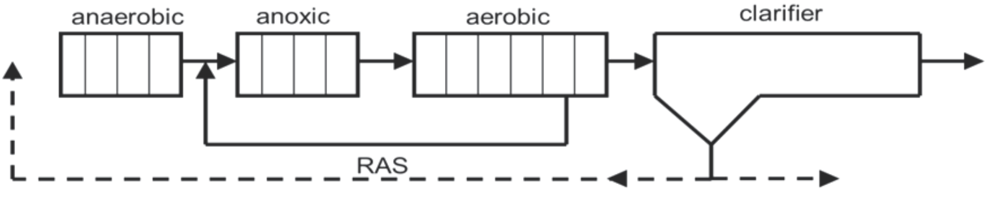


Fig. 11. A²/O

Biodenipho (Krüger, n.d.)

Two circular aeration tanks are operated with alternating nitrification and denitrification. Waste water flow is added during the periods of denitrification where aeration is stopped. Thus waste water of the first activated sludge tank will be denitrified while the second activated sludge tank will be aerated and therefore nitrification will occur. As soon as all nitrate in the not aerated tank will be denitrified, the inflow and the aeration changes. Thus the created nitrate of the aerated tank will be denitrified and the available ammonia in the aerobic tank will be oxidized. An upstream situated anaerobic tank enables a biological P-elimination.

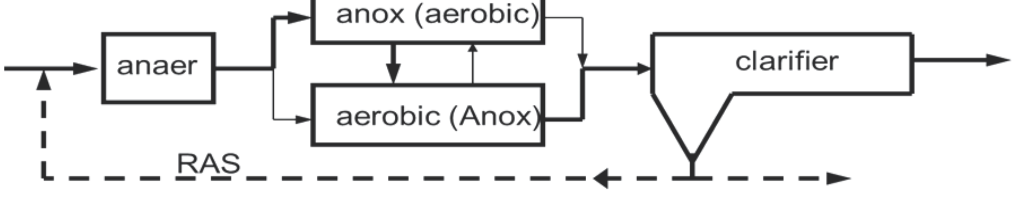


Fig. 12. Biodenipho-Process

Extended Anaerobic Sludge Contact (EASC) (Schoenberger, 1989)

This process was developed to implement biological P-elimination to existing WWTP. The primary sedimentation tank is used as an anaerobic reactor. Return sludge is fed to the sedimentation tank where anaerobic conditions enable sedimentation of primary sludge and RAS (Return activated sludge). Therefore the residence time of the sludge extends and leads to an acidification of the raw water. This acidification leads to improvement of the substrate quality for P-storing microorganisms. The runoff of the sedimentation tank and settled sludge will be fed to the anoxic tank together with the sludge recycled from the aerated nitrification tank.

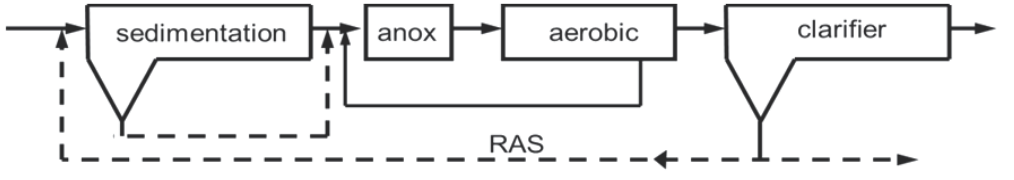


Fig. 13. Extended Anaerobic Sludge Contact (EASC) Process

ISAH (Austermann-Haun, 1998)

The ISAH is an approved process under unfavourable conditions (low temperature, dilution by external water or low substrate concentrations). RAS will be denitrified in a separate anoxic tank, which inhibits a possible disturbance of phosphate re-dissolution. Thus the easy degradable waste water inflow is fully available to phosphate-rich microorganisms.

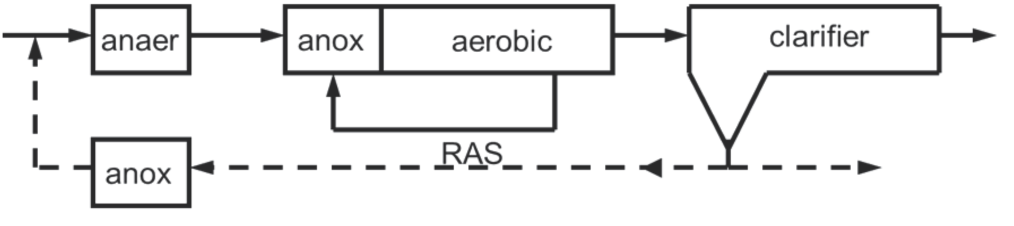


Fig. 14. ISAH-Process

Sequencing Batch Reactor (SBR) (ATV, 1997)

Compared to the continual flow processes, SBR works with only one tank in time sequences. One cycle passes the steps of filling (anoxic), filling and mixing (anaerobic), aeration (aerobic), sedimentation and removal of the treated effluent (Fig. 15).

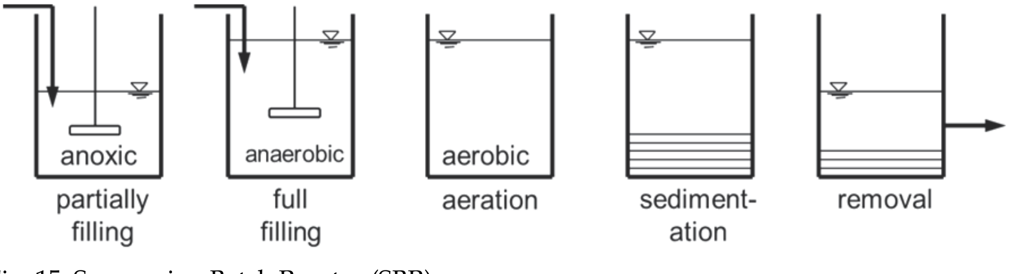


Fig. 15. Sequencing Batch Reactor (SBR)

Full scale experience shows that Bio P-removal can be achieved with many aeration tank configurations where anaerobic zones or cascades occur and irrespective whether the treatment efficiency is high i.e. with nitrification or only BOD-removal is achieved. Typical reports on these full scale investigations can be found in (Spatzierer et al., 1985) and for Berlin (Peter et al., 1991). Especially for treatment plants with low treatment efficiency (low sludge age) it has to be considered that in secondary clarifiers anaerobic conditions can occur which will cause a release of stored phosphate to the effluent.

Side stream processes

Phostrip (Levin, 1965)

In contrast to the in stream processes the anaerobic phase is situated in the return sludge cycle. Core of this process is the stripping tank which works similar to a common sludge thickener. The steps in this tank are:

- Separation of sludge and waste water by sedimentation
- Sludge thickening up to 4 % dry matter
- Anaerobic storage of the sludge
- Hydrolysis of particular COD or activated sludge to easy degradable organic substrate
- *P-release*

In this sludge-water phase concentrations up to 50 mg L^{-1} PO_4-P are possible. The actual practical relevance of this process is low.

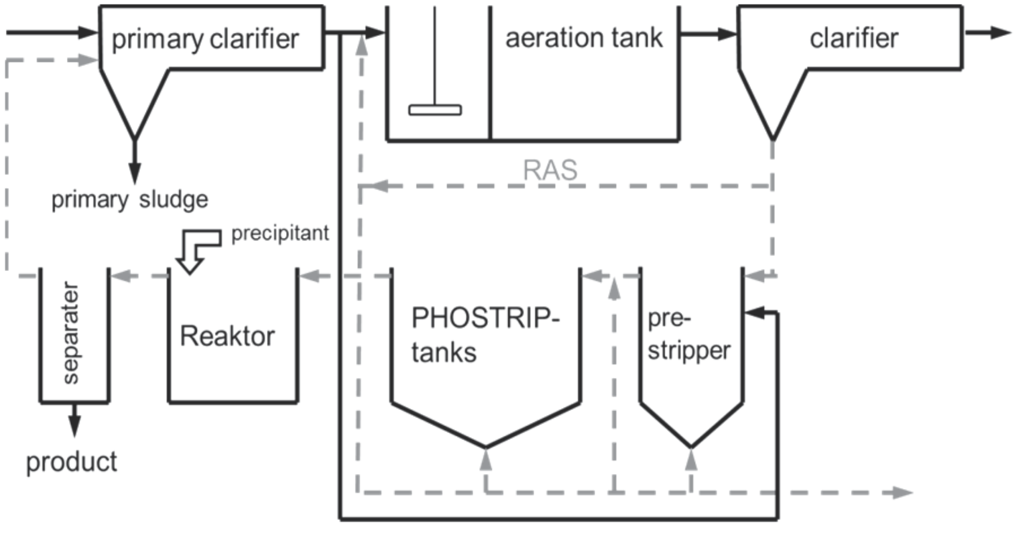


Fig. 16. PHOSTRIP-Process

2.2.4 Crystallisation processes

Crystallisation precipitation) processes are mainly applied to remove phosphorus from sludge liquors and from urine if separately collected. These processes are easy to integrate in to consisting WWTP. By adding different chemicals as calcium hydroxide ($Ca(OH)_2$), calcium chloride ($CaCl_2$), magnesium hydroxide ($Mg(OH)_2$) or magnesium chloride ($MgCl_2$) to crystallisation reactors, calcium phosphate ($Ca_3(PO_4)_2$) and magnesium-ammonium-phosphate (MAP -> NH_4MgPO_4 x H_2O) will be formed. This product can be used directly in agriculture. Following to the P-recovery, the pH of the purified water needs to be lowered because of the alkaline chemicals in the reactor. With this process only ≤ 45 % of the P load of the raw waste water can be recovered. Worldwide there are few different large scale realisations but also numerous attempts to implement this technology successfully at waste water treatment plants especially to sludge liquors with high phosphate and ammonia concentrations. These are namely:

DHV Crystalactor ®	Netherlands	Giesen (2002)
WASSTRIP	US	Ostara (2008)
Unitaka Phosnix	Japan	Ueno et al. (2001)
Nishihara Reactor	Japan	Petruzzelli et al. (2003)
WWTP Trevisio	Italy	Cecchi et al. (2003)
PRISA	Germany	Montag (2008)

Table 2. Crystallisation processes worldwide

2.2.5 Urine separation

The daily amount of urine per person of 1.5 to 2 litres offers a highly concentrated dissolved phosphate mass flow. The following Table 3shows the distribution of nitrogen and phosphorus in urine and faeces (Vinneras, 2004).

	Unit	**Urine**	Faeces	**Urine (%)**	Faeces (%)
N	g/(PE*a)	**4000**	550	**88**	12
P	g/(PE*a)	**365**	183	**67**	33

Table 3. Distribution of nitrogen and phosphorus in human excretions

Dry toilets, vacuum toilets, separation toilets or waterless urinals are necessary to separate urine. Afterwards the phosphorus and part of the ammonia can be recovered by MAP-precipitation, resulting in a market fertiliser (Bischof, n.d.).

MAP-precipitation

MAP (Struvite) is produced when ammonium, phosphate and magnesium ions react in a stoichiometric molar ration of 1:1:1. By adding MgO or MgCl to the urine with a minimum pH of 9, MAP can be precipitated. If a complete ammonium recovery is aimed, phosphate and additional magnesium have to be added (Sreeramachandran, 2006). The final product MAP is low in heavy metals and micro-pollutants and represents a valuable market fertiliser. Figure 17 shows the multi-stage process developed by Hans Huber AG with ammonia recovery by stripping.

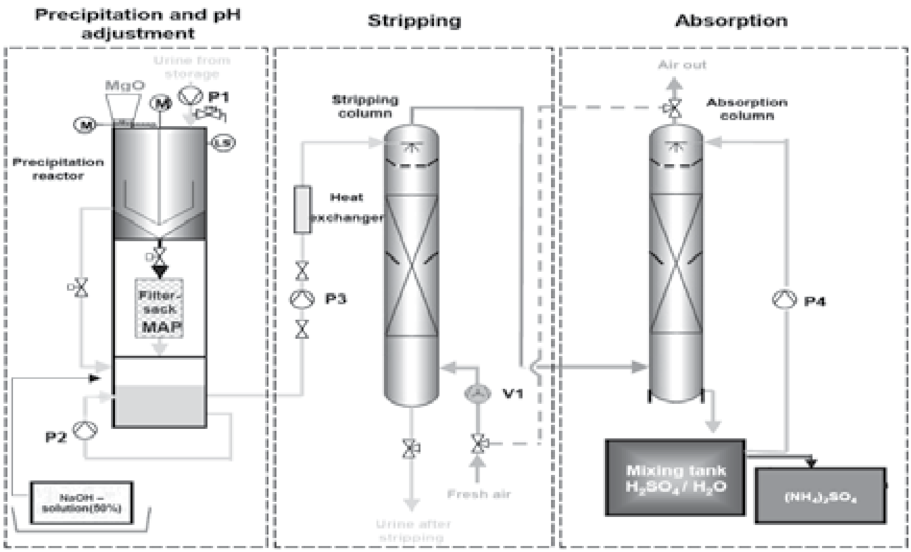


Fig. 17. Multi-stage MAP-Precipitation with ammonia recovery by stripping (Bischof, n.d.)

2.2.6 Cost estimation

The costs of P-elimination are subjected to regional and temporal variations. They include WWTP operational costs for the P-removal (chemical cost) and have to reflect the influence on sludge treatment and disposal. Therefore reported costs strongly depend on specific local situation, assumptions made and cost calculation method applied. Table 4 shows costs for P-elimination derived from literature (EAWAG, 2008, Baumann, 2003).

Treatment	Costs
Biological P-elimination	3,5 €/(PE*a) 2,5 - 9 €/kg $P_{eliminated}$
Chemical P-elimination	4 - 9 €/(PE*a) 6 - 14 €/(kg $P_{eliminated}$)

Table 4. Specific costs for P-elimination

The specific costs of new recycling technologies from sewage sludge and sewage sludge ash are 1.5 - 5 €/(inhabitant*year) or 5 - 20 €/kg $P_{recycled}$. Recycling directly from waste water is most expensive due the low recycling rates and high investment cost. The actual market price for 1 kg $P_{recycled}$ is about 0.5 - 1 €. Therefore these technologies are presently not cost efficient (Pinnekamp, 2007).

2.3 Sewage sludge

Sewage sludge is a necessary by- product of waste water treatment and is a mixture of water and solids. It consists of primary sludge from primarily setting tank, excess sludge form the biological treatment step and precipitants from chemical P-precipitation. Because sludge production cannot be avoided the operator of a waste water treatment plant needs a technologically and legally reliable sludge disposal method at any time. This has to be considered especially for all marketing strategies for sludge or products from sludge.

2.3.1 Quantities

Due to increasing capacity of waste water treatment in EU the quantity of sewage sludge is increasing, too. In Europe (EU 15) the actual sludge production is about 10 Mio tons dry matter per year which corresponds to about 22 kg per capita (EUROSTAT, 2010). North America (USA and Canada) produces about 7.7 Mio tons of municipal wastewater sewage sludge every year. Calculated with a P concentration of 2.5 to 3 %/t DM the theoretical P-recovery potential in Europe and North America is up to 0.5 Mio tons every year. This corresponds to ~ 3 % of the annual phosphorus fertilisers sourced from mining phosphate rock (15 Mio tons P (Jasinksi, 2010)).

2.3.2 Composition of sewage sludge

Nutrients

Depending on the origin of waste water the composition of sewage sludge can vary in a wide range. Nutrients in sewage sludge originate mainly from human excreta but also from detergents and different industrial effluents (food, pulp and paper, chemical industry). Typical concentrations of valuable sludge compounds are shown in Table 5.

oDM	N	NH_4-N	P_2O_5*	K_2O	CaO	MgO	S	Na
% DM	g/kg DM	g/kg DM	g/kg DM	g/kg DM	g/kg DM	g/kg DM	g/kg DM	g/kg DM
35 - 60	**13 – 65**	0,6 - 13	**20 - 45**	1 - 8	60 - 130	5 - 16	5 - 10	1 - 3

Table 5. Nutrient concentrations in sewage sludge (Zessner & Aichberger, 2003)

Sludge application has to be integrated into agricultural fertiliser management. For this goal availability of the P content of the sludge has to be considered. Enhanced biological P-elimination without or little use of precipitants during the waste water treatment process is favourable for agricultural use of sewage sludge, as the availability for the plants is high. Phosphorus availability is lower with alum precipitation and even more restricted with iron precipitation. P-precipitation as MAP results in full availability of the phosphorus and contains also ammonia as a fertiliser but is normally not applied for pre- and simultaneous P-removal.

Literature reports regarding plant availability of phosphorus compounds in sewage sludge in agriculture, widely differ depending on the investigation methods, the sludge applied, and whether lab-scale tests or full scale experience have been used. The availability not only depends on the P removal process applied (biological/chemical, chemicals used) but also on the soil properties (pH, redox potential, type of soil, content of humus, organic substance and nutritional status) as well as on sewage sludge stabilization process applied

(anaerobic/aerobic). Also the test procedure for the determination of plant availability can be of substantial influence. There is consent that short term P-availability of iron or aluminium precipitated phosphorus in sewage sludge is restricted (Krogstad et al., 2005; Henke, 2000; Jokinen, 1990). Biological treatment with lime addition for P-removal resulted in sewage sludge with a P fertilisation effect comparable to inorganic P fertilisers as investigated in pot trials. The application of sewage sludge to pot experiments also resulted in low concentrations of water extractable P, which is positive. They caused a considerable accumulation of P with low plant availability in soil (50 - 95 % increase) which represents a potential environmental risk due to the transport of erosion products by surface runoff to receiving waters (Krogstad et al., 2005). This also stressed the importance of inclusion of sludge application into fertiliser management as reported by Krogstad et al. (2005).
Plant roots absorb exclusively dissolved inorganic phosphate, but its concentration in soil solution is low (Blume et al., 2010). Especially during the growing phase the need of dissolved P is markedly increasing. By desorption of adsorbed P, dissolution of Ca-phosphate and mineralization of organic P the need of dissolved phosphate will be satisfied (Blume et al., 2010) Most market fertilisers contain phosphorus compounds which are immediately and at least easily available to plants. Nutrients included in organic fertilisers mineralise slowly to plant available substances. It therefore can be recommended to combine mineral fertiliser application with organic fertilisers as sewage sludge. The characteristic of P compounds in sewage sludge is shown in Table 6.

Inorganic phosphorus	60 - 90 %, depending on P-precipitation (chemical or biological) and sludge treatment (anaerobic/aerobic)
Water soluble Citric acid soluble	<1 - 38 % 60 - 90 % only biological treatment (Gutser, 1996) Clearly lower after chemical precipitation
Inorganic compounds	Octacalcium phosphate ($Ca_8H_2(PO_4)_6$ x $5H_2O$) Dicalcium phosphate (-dihydrate) Fe- and al-phosphate (vavianite and wavellite)
Organic compounds	Monoester Diester Phosphorus lipids

Table 6. Characteristic of P in sewage sludge (Frossard, 1996)

Where P removal from waste water is required, normally the area specific application of sewage sludge is limited by the phosphorus addition, which should be adapted to the crop uptake in order to avoid unwanted P-accumulation in top-soils (eutrophication abatement from erosion). Guidelines on application of sewage sludge in agriculture in regard to phosphorus fertiliser and heavy metal management are available (e.g. ÖWAV Regelblatt 14, 2004). Sludge from treatment plants with enhanced P-removal requirements can be classified as a phosphorus fertiliser. Normally phosphorus content limits the mean area specific application of sludge to1 - 2t DM/ha/year.

Metals and heavy metals

Heavy metal loads in waste water and hence in sewage sludge varies in a broad range depending on industrial and trade discharges, surface runoff, sewer system, household infrastructure, waste water treatment system and geogenic background. Typical metal concentrations (mg/kg DM) in Central Europe sludges are shown in Table 7. The heavy metals used for sludge quality characterisation in sludge regulations are marked in bolt.

As	Be	Br	**Cd**	**Cr**	Co	**Cu**	Fe	**Pb**
4 - 10	0,2	37	**0,6 - 3**	**3 - 54**	5 - 11	**120 - 300**	11800 - 17000	**37 - 145**

Mn	**Hg**	Mo	**Ni**	Se	V	Sn	**Zn**
220 - 320	**0,5 - 2,3**	3,9 - 14	**17 - 37**	1,8	15	32	**700 - 1320**

Table 7. Heavy metal concentrations in sewage sludge (Zessner & Aichberger, 2003)

Metals as boron, iron, copper, zinc, molybdenum, manganese or selenium are essential trace elements for plants and living organisms, but too high concentration can be harmful and toxic. Heavy metal as cadmium, lead and mercury do not have proved functions in living organisms and can be toxic and harmful beyond threshold concentration or doses. Therefore the area specific application of sewage sludge to agricultural area is strictly regulated in order to avoid an enrichment of heavy metals in soil or plants (Phatak et al., 2009; Gaskin et al., 2003; LFU, 2003). The standards for the heavy metal concentrations can be either expressed as mg/kg DM or as mg/g P, the latter is especially relevant for sludge with high P-content up to 35 g P/kg DM from P-removal treatment plants.

Pathogens

Sewage sludge contains pathogens as bacteria's, viruses, protozoas and worm eggs which stem mainly from human excreta but also from animals. Anaerobic mesophilic stabilisation and low temperature drying do not achieve the required reduction and inactivation of those pathogens. Especially for salmonella, enteroviruses, roundworm eggs, cryptosporidium, multi-resistant enterococcus and staphylococcus (Böhm, 2006). This means that sewage sludge has a potential to transfer infectious pathogens to animals and humans. Depending on the application of the sludge and the infection potential different hygienic quality criteria can be derived. Appropriate processes to minimize the hygienic risk of sludge application are sludge pasteurisation, thermophilic treatment, quicklime addition, composting and long term storage. The highest hygienic risk is at the farm level, where a close contact of animals and humans with sludge can occur. Furthermore sludge application is restricted to specific production areas.

Micro-pollutants

Micro-pollutants are ubiquitous in the aquatic environment even the concentrations of only a very limited number is monitored. Their effects on organisms differ widely and are often insufficiently investigated. Depending on their properties the substances can have genotoxic/immunotoxic/neurotoxic, carcinogenic and endocrine impact on living organisms (Gangl, 2001). Table 8 shows different micro-pollutants of concern:

Organic pollutants	AOX, LAS, PAH, PCB, PCDD/F, DEHP, HC, NPE
Pharmaceutical substances	Antibiotics, endocrine hormonal drugs, psychotropic drugs, cytostatic

Table 8. Micro-pollutants in waste water and sewage sludge

Up to now there are no scientific reports on negative effects on agriculture and food if controlled sludge application on land is used even for decades in several regions. Whether they represent a long term risk for humans and the environment it is still a matter of scientific research and discussion.

2.4 Recovery, treatment and disposal of sewage sludge

The following figure shows the current situation of sewage sludge recovery, treatment and disposal in Europe and North America (Emscher Lippe, 2006; WEF, 2011; CCME, 2011).

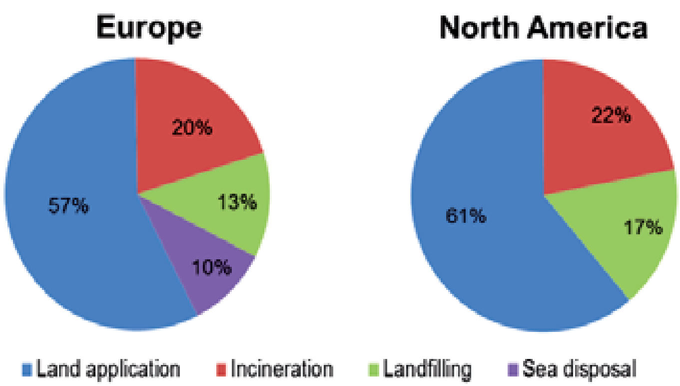


Fig. 18. Sewage sludge recovery, treatment and disposal in the EU and North America

2.4.1 Direct land application

Direct application of sludge in agriculture is closing the nutrient cycle especially for phosphorus. Sewage sludge contains also valuable other nutrients (nitrogen), organic matter and many macro- and micronutrients which are essential for plant growth.

Use of stabilised sewage sludge on land

The use of stabilised sewage sludge on agricultural land has a long tradition and is subject to strict legal requirements for quality control, monitoring and documentation (see section 1). Especially for small treatment plants in rural areas this method represents an easy and economically favourable solution for P-recycling. For national agriculture use of sludge in agriculture is of of low economic relevance (Kroiss et al., 2007). For farmers able to substitute mineral phosphorus fertiliser free of charge by sewage sludge this is economically interesting

(Kroiss, 2005). For the treatment plants it has the important consequence that a sludge storage capacity for up to 6 months has to be provided, as during wintertime fertilising is not allowed. For large treatment plants the main problem of this disposal route is the restricted reliability (legislation, public perception) as well as increasing costs for storage and transport.
Sludge can be composted if additional carbonaceous material (e.g. wood chips) is added. The compost can be applied in agriculture and for landscaping as it is possible for sludge depending on national regulations. If sludge or compost made of sludge is used for landscaping in most of the cases the area specific phosphorus dosing is much too high as compared to the uptake if relevant area specific mass of organic material is applied. This is detrimental for P-resource protection and may contribute to eutrophication by erosion products. In EU15 currently 40 % of sewage sludge produced (4 Mio t/a) are directly applied in agriculture. 17 % are used for recultivation. In North America 61 % (4.6 Mio t/a) were applied on land.

Problematic of harmful substances

In principle the application of sewage sludge can cause an increase of heavy metals in soils if removal by harvesting and washout is lower than supply. There is continuous loss of HM via surface runoff, intermediate runoff and the ground water which is very difficult to quantify due to the limited analytical and sampling accuracy. Numerous studies show, that the accumulation of heavy metals is very low as the dilution factor of sludge in the top soil is in the order of 1:5000 up to 1:10.000 if sludge is applied according to modern legal requirements. Only monitoring with sophisticated sampling procedures over several decennia can prove an accumulation. Heavy metal loading of soils has therefore to be monitored in order to avoid potential risks which are different for several metals (VDLUFA, 2001).

heavy metal	**soil protection**		**plant nutrition, quality of food plants**	**risk**
	increase of soil content	mobility		
Cd	possible	high	endangered	high
Pb, Cr, Ni, Hg	possible	minimal	not endangered	medium
Cu, Zn	possible, welcome by fertiliser need	Cu low, Zn high	encouraged by fertiliser needs, otherwise no risk	low

Table 9. Assessing heavy metals concerning their possible risk

Plants have "root barriers" which inhibit or even stop the uptake of certain heavy metals (Pb, Cr, Ni, Cu, Hg) and many organic micro-pollutants. With the exception to Cd and Zn, plants are protected concerning the uptake of high concentration of these substances. Zn is also an important trace element for plant growth and human nutrition, Cd concentration in much sludge from Central European and also US treatment plants has dropped below the soil standards.
Soils contain the most versatile natural microbial communities with high performance potential in mineralizing organic substances, even so called persistent harmful substances as PCB and PCDD as could be verified by research Also the adsorption potential as very high

due to the extremely large surface area. As a consequence the controlled application of sewage sludge on land does not result in acute risks, long term risks by accumulation can be avoided by adequate monitoring. Sludge is not the only pathway for micro-pollutants to the soils (air pollution, precipitation).

2.4.2 Incineration and P-recovery

A process enabling P recovery of phosphorus is the incineration of sewage sludge in mono-incineration plants. All organic compounds will be destroyed, while phosphorus and the heavy metals are transferred to the ash. The direct application of this ash to agricultural fields is still a matter of discussion. The availability of P in the ash is restricted. The main goals of new P-recovery technologies are on the one hand the elimination of pollutants and on the other hand making phosphorus available to plants. Currently there are only few technologies available which meet both requirements, but they are still not ready for market introduction. The following technologies for P-recovery from ash are reported in literature: ASH DEC, PASCH, Mephrec and ATZ Eisenbadreaktor (Mocker and Faulstich, 2005). An immediately applicable option could be to store the P-rich ash in a monofill for future recovery The use of mono-incineration ash for construction material or its dumping in landfills together with other waste should be avoided as phosphorus recovery will be disabled.

2.4.3 Incineration without P-recovery

Because of the relatively high calorific value (11 - 17 MJ/kg) of dried sewage sludge, comparable to brown coal and therefore used in the cement industry, in coal power plants but also in ordinary municipal waste incineration plants. Dried sewage sludges are used in the cement industry, in coal power plants but also in ordinary municipal waste incineration plants. In these processes all organic compounds will be destroyed completely, but the valuable nutrients as P cannot be recovered. End products ash bottom as and fly ash with low content of pollutants can be used as a construction material or get landfilled. Pollutant rich filter cake need to be disposed of in underground disposal facilities.

2.4.4 Landfilling

In Europe and North America about 2.5 Mio tons of sewage sludge are currently dumped in landfills. This causes gaseous emissions as CH_4 and CO_2 from these landfills, which are climate relevant. Phosphorus in this dumped sewage sludge is lost irretrievable. European landfill legislation therefore requests a continuous reduction of organic material to be put to landfill, with the goal to completely stop it in the near future. Several central European countries have already banned landfill disposal of organic matter in the past (Germany, Austria).

2.4.5 Possibilities of P-recovery from sewage sludge

Due to the pollutants contained in sewage sludge a great number of research and development projects have been started to recover phosphorus fertiliser with low pollution from the sludge, in order to meet the same quality standards as for market fertilisers. Most of the processes described below have not proved economic viability up to now, some of them are still lacking full scale experience.

Processes with precipitation

There are three main processes to recover phosphorus fertiliser with low pollution levels and high plant availability from sewage sludge. Enhanced biological P-elimination without or low use of precipitants during the waste water treatment process is advantageous for working-recovery by precipitation from the sludge. In sewage sludge phosphorus is bound to several organic and inorganic solids. By changing the pH using acids, phosphorus can be brought into solution. Particulate matter will be separated and the pH is increased to about 8.5 by adding alkalinity. If e.g. MgCl is used as precipitant for MAP a fertiliser rich in phosphorus with high plant-availability and low heavy metal content will be produced. (Airprex, Seaborne, Stuttgarter Verfahren)

Wet oxidation process

During the wet oxidation process the organic fraction of sewage sludge is oxidized with pure oxygen at super-critical conditions (pressure > 221 bar, T > 374 °C). Phosphorus concentrates in a highly reactive form and will be extracted by precipitation with calcium hydroxide. (Aqua Reci)

Thermal hydrolysis with following precipitation

Sewage sludge will be heated under pressure up to 140 C and treated with sulphuric acid to reach a pH of 1 - 3. Part of the inorganic material dissolves and is separated from the particulate matter. By increasing the pH in the liquid phase phosphorus is precipitated by adding iron salts. The plant availability of P is comparable to simultaneous precipitation. (KREPRO)

2.4.6 Discussion

The direct application of sewage sludge on land is a well-established method of nutrient and organic substance recovery. The sludge treatment processes applied (storage, dewatering, drying) have to be adapted to the specific local situation including the legal requirements for, monitoring and reporting and the whole logistics. Sludge composting is also a well-established sludge disposal method. If sludge compost is used according to the requirements for organic material (land reclamation or soil conditioning in agriculture normally the P-addition is much higher than plant uptake which is detrimental for P-recovery and eutrophication abatement. The relevance of the potentially harmful substances in the sewage sludge applied on land for long term soil protection and related health effects are still a matter of research and discussion. It finally can only be solved by a political agreement on an acceptable risk at acceptable costs. The processes to recover phosphorus from sewage sludge with a quality as market fertilisers with new technologies, as described in section 1.4.1, use large quantities of chemicals (acids, bases) and energy. The remaining waste fraction after phosphorus extraction still contains potentially harmful compounds and will have to be disposed or reused. Currently these technologies are not competitive economically. Incineration is applied to recover the energy contained in the organic fraction of the sludge. During incineration micro-pollutants are destroyed and phosphorus is concentrated in the ash if mono-incineration of sludge is applied. Co-incineration of sludge with coal (power plants) or solid waste therefore should not be used in the future, the same is with sludge incineration in cement factories. Whether the ash of mono-incineration plants can directly be applied on land (P-contents similar to market fertiliser) is still a matter of discussion because of the heavy metal content and the reduced P-availability.

Sludge from nutrient removal plants with bio P and/or aluminium P-precipitation can be used as raw material for phosphate fertiliser industry (Schipper et al., 2004)

3. Phosphorus in waste management

Vegetable and animals wastes contain significant quantities of phosphorus. Major sources for such wastes are agriculture, the food processing industry and private households.

3.1 Private households

The average P-content in mixed household waste is reported with 0.9 g P/kg fresh mass (FM) in Schachermayer et al. (1995) and 1.4 g P/kg FM in Skutan & Brunner (2006). This translates into a P-load of 190.000 to 300.000 tons/a for the EU15. The proportion of organic waste at the whole municipal solid waste generation is up to 35 %. In EU15 this corresponds to 75 Mio tons every year and a P-load of about 150.000 (Figure 19). Thereof only about 30 % or 22 Mio tons are collected separately. This separately collected organic waste fraction consists of kitchen- and garden waste from households and park- and garden waste from public area. The current waste treatment options are shown in Figure 19.

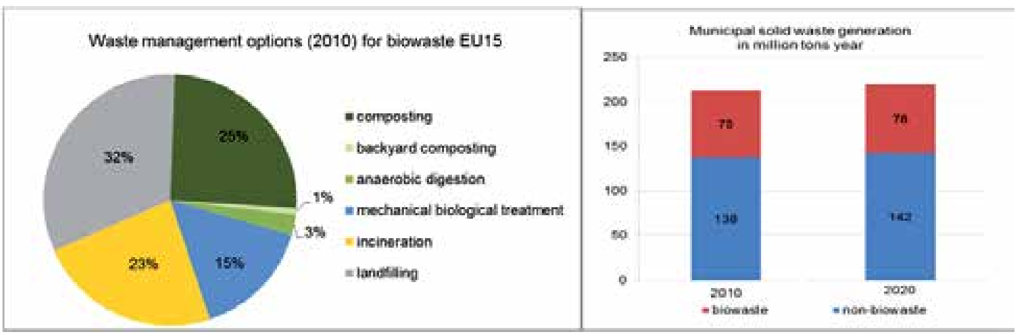


Fig. 19. a) MSW generation in households; b) Waste treatment of biowaste (Arcadis Eunomia, 2010)

Taking the loss of composting into account, 11 Mio tons of compost can be generated and therefore 50.000 tons of phosphorus can be recovered every year at current collection rates. The potential amount is ca. three times higher under real conditions. If this potential can be exploited, up to 150.000 tons P could be recovered from biowaste annually. In Europe approximately 50 % of the produced composts are applied on agricultural fields. The remaining quantities are used in landscaping, gardens or in humification processes. Another appropriate treatment for organic waste, especially pasty wastes is anaerobic fermentation. The resulting biogas slurry can be used as an organic fertiliser.

3.2 Food industry (vegetable and animal waste)

The amount of organic waste generated by manufactures of food products, beverages and tobacco products is about 150 kg per habitant and year in Europe (EU15) (Oreopoulou, 2007; EU STAT, 2011). This corresponds to a total of 59 Mio tons. Because of the heterogeneity of these wastes the P-recovery potential is difficult to determine. Under the assumption of an average phosphorus concentration of 0.5 %, the recovery potential of vegetable and animal waste is about 290.000 t/a. Due to the high P-concentration, especially in bones and teeth, animal wastes contain most of the phosphorus load from the food industry. Waste from slaughtering and meat processing are treated in animal cadaver utilization plans. Therefore annually approximately 9 kg (Nottrodt, 2001; ASH DEC, 2008) of carcass meal emerge per inhabitant in Europe. Related to all inhabitants in the EU15 3.5 Mio tons of carcass meal arise every year. Calculated with a P-concentration of about 5 to 6 % the recovery potential is approx. 200.000 tons of phosphorus. This P-load corresponds to about 70 %of the total wastes from food industry.

3.3 Ash from energy wood

According to the statistics of EU STAT, 60 Mio tons (dry matter) of energy woods like firewood, wood chips and wood residues (including pellets) are used as alternative energy source. With an assumed ash content of 1.5 % and a P-concentration in ash of 1.2 % a potential P-load of 10.000 tons/a can be calculated.

3.4 Steal production

In steel production P is viewed as harmful to the production of high-quality steel. P occurs in coal, iron ore, and limestone, which are the main raw materials for iron making. During the steelmaking process P is transferred from the molten pig iron to the slag. Yoon and Shim (2004) report P concentrations in dephosphorization slag of 1 - 3 % (P_2O_5). Jeong et al. (2009) demonstrate the potential of such slag for P recovery by a P balance for South Korea where they show that steelmaking slag contains about 10 % of the domestic P consumption. They argue that technologies to recover this waste flow could substantially reduce the dependence on imports of phosphate rock.

3.5 Recovery processes for organic waste

3.5.1 Composting

The main treatment option of separately collect organic waste in households is composting. During this aerobic treatment process, the organic fraction gets stabilized through microbial decay and volume and mass are reduced while the concentration of nutrients increases. Composting requires three key activities: aeration (by regularly turning the compost pile), moisture, and a proper carbon to nitrogen (C:N) ratio. A ratio between 25:1 and 35:1 is generally considered as optimal.

3.5.2 Biogas plants

Biogas plants are a well-known technology to transform organic wastes into a useful fertiliser, to gain electricity and thermal energy from them and to increase their nutritive characteristics. Through biologic decomposition under anaerobic conditions methane bacteria produce biogas. The methane is used for combustion either in a gas motor or combined heat and power plant to produce electricity and heat (e.g. for district heating). The resulting biogas slurry can be used as an organic fertiliser.

3.5.3 Thermal treatment

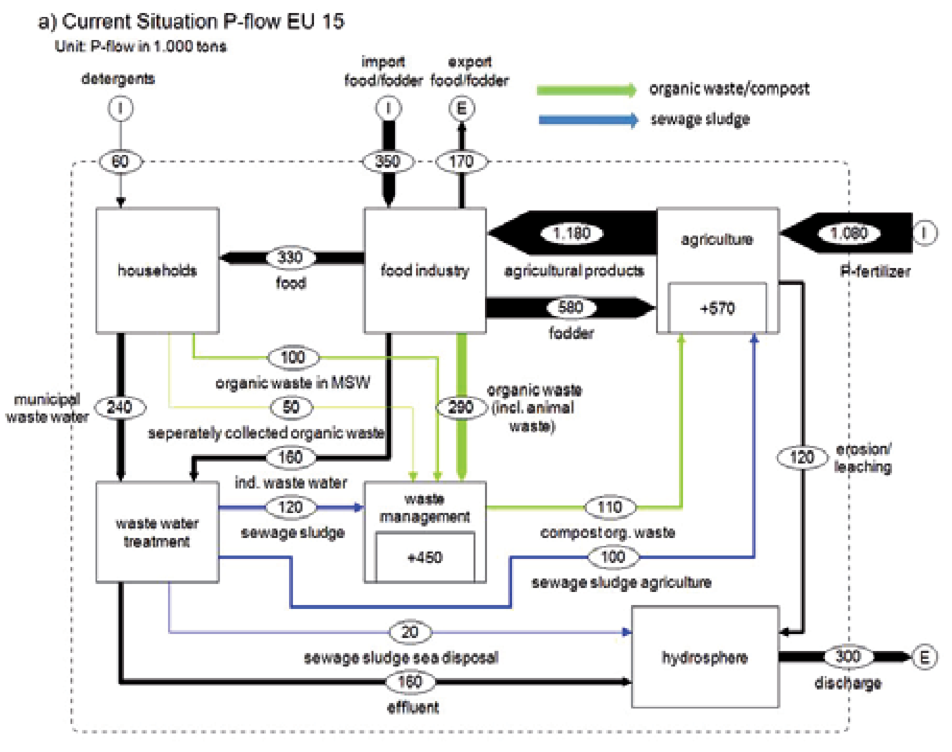
Utilization of carcass meal as animal feed has been banned as a consequence to the BSE crisis and therefore most of the carcass meal is utilized as a substitute fuel in the industry (mainly in cement kilns and coal-fired power plants). This treatment does not allow a recovery of phosphorus since it is either diluted in the product (cement) or in the coal ash. A possibility could be the co-incineration with sewage sludge in mono-incineration plants and recovering phosphorus from ash (Driver, 1998).

3.5.4 Conclusion

The present amount of organic waste from households and food production waste will not change significantly. But there is additional P-recovery potential concerning the separately collected organic waste. By tapping these potential the amount of P could theoretically rise from 50.000 to approx. 150.000 tons of P. In the sector of food production the recovery (anaerobic and aerobic treatment, fodder) is nearly 100 % and therefore there is no additional potential. As demonstrated in section 3.2, phosphorus is highly concentrated in animal wastes, but the present treatment (mainly incineration without P-recovery) does not allow using the possible P-quantities of over 200.000 t. Mono-incineration would allow the future recovery of the containing phosphorus if the ashes are stored in monofills. The potential phosphorus in ashes from energy wood is not practical for the production of a secondary P fertiliser because of the low phosphorus amount and the decentralized occurrence of these ashes. However, these ashes can be applied directly to the soil if the contents of heavy metals are moderate.

4. Scenario evaluation for European P-management

Figure 20a shows a simplified P-balance for the EU15. The dominating process is "agriculture" consuming 1.9 Mio t of P per year. Less than 0.4 Mio t/a of it reach the consumer ("Household"), showing that the P-chain is characterized by low efficiency and large losses such as accumulation of P in soils and landfills, losses to the hydrosphere by erosion, leaching, and waste water discharges. Figure 20b shows a partly optimized system, where the following adjustments or assumptions are made:



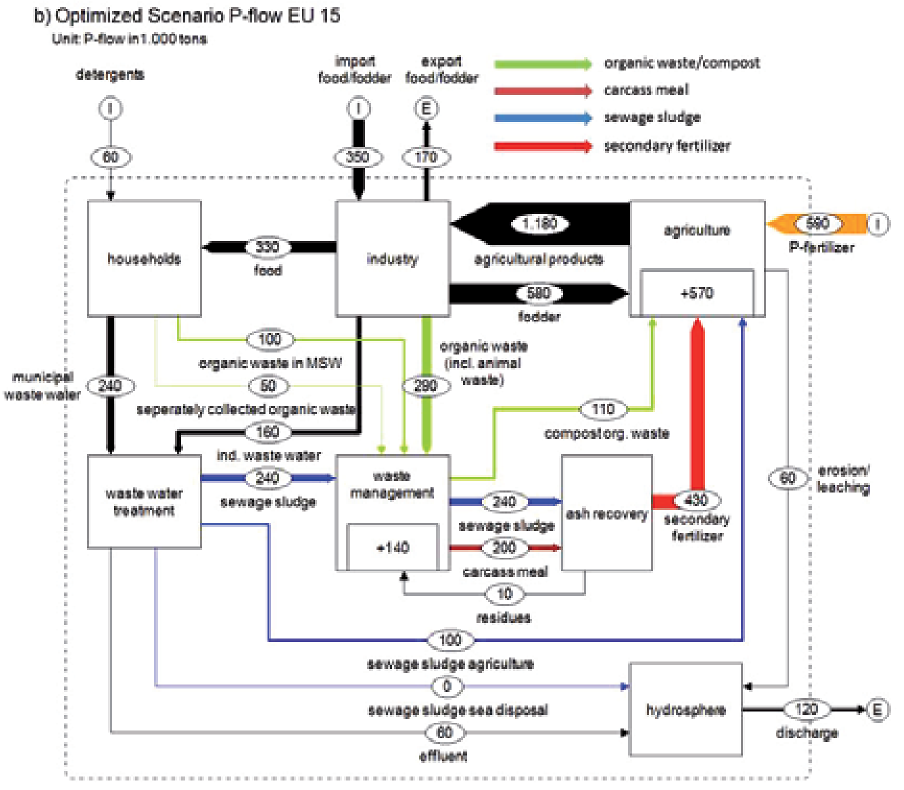


Fig. 20. Simplified phosphorus balance for the EU15: a) current situation (average year in the period 2005-2008); b) optimized scenario

- 50 % erosion reduction by implementing an efficient erosion abatement strategy for Europe
- mono-incineration of contaminated sewage sludge combined with carcass meal and production of a P-fertiliser from the ash
- no ocean dumping of sludge (already forbidden)
- 85 % P removal at all waste water treatment plants
- the amount for sewage sludge recycled in agriculture is maintained

The result as shown in Figure 20b is that losses to landfills and the hydrosphere are reduced significantly (-69 % and -60 %, respectively) and the import of P to the EU15 decreases by 45 %. Such scenarios show that there is considerable potential to optimize P management whereby optimization is a mixture of the implementation of new technologies and management practices in agriculture and waste management.

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12

Anaerobic Processes for Waste Treatment and Energy Generation

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1. Introduction

As global population increases and developing countries industrialize, energy demand around the world is increasing markedly. World energy consumption is expected to increase by 50% to 180,000 GWh/year by 2020 (Fernando et al., 2006), due primarily to increases in demand from rapidly growing Asian countries such as China and India (Khanal, 2008). According to the Intergovernmental Panel on Climate Change (IPCC, 2007), fossil fuel combustion already contributes 57% of emissions that cause global warming. Thus, to address future energy needs sustainably, renewable sources of energy must be developed as alternatives to fossil fuels.

To aid in developing such renewable energy alternatives, environmental scientists and engineers should consider anaerobic processes for waste treatment as alternatives to aerobic processes. When aerobic processes are used for waste treatment, the low energy compounds carbon dioxide and water are formed; much energy is lost to air – about 20 times as much as with an anaerobic process (Deublein and Steinhauser, 2008). Anaerobic processes produce products of high energy like methane. Methane can be captured and burned as an energy source, and used to power gas-burning appliances or internal combustion engines, or to generate electricity.

Anaerobic processes have been applied for decades in developed countries for wastewater treatment plant sludge stabilization. In recent years, considerable interest has developed in use of anaerobic treatment for a variety of other applications, due to the potential to generate renewable energy. Methane from anaerobic processes is being increasingly utilized as an alternative energy source in developed countries, via large projects that extract methane from landfills or wastewater treatment plants. Smaller plants, on the scale of an individual household or village, can also be a particularly important energy source in rural sectors of developing countries; transportation costs in these locations may limit use of fossil fuels, and lack of cheap and adequate energy hampers rural development. When generated from biomass, especially at a small scale, methane is often called biogas (FAO, 1984; Deublein and Steinhauser, 2008).

In addition to providing a renewable source of energy, anaerobic processes provide some of the simplest and most practical methods for minimizing public health hazards from human and animal wastes – pathogens are destroyed or greatly reduced. Anaerobic processes have been proven for treatment of a variety of organic wastes: solid wastes at landfills, industrial wastewater, human excrement and sludges at wastewater treatment plants, human excrement in rural areas, animal manure, agricultural wastes, and forestry wastes. The

residue is a valuable fertilizer, which is stabilized and almost odorless. This fertilizer is especially a benefit in developing countries, due to its potential to boost crop yields.
This chapter will discuss:

- basics of the anaerobic degradation process,
- methane production: quantities and rates,
- gas production system design, and
- benefits and limitations of anaerobic waste treatment processes.

2. Anaerobic process basics

Anaerobic degradation of organic material (biomass) involves decomposition by bacteria under humid conditions where contact with molecular oxygen is eliminated. The overall process of anaerobic degradation can be represented as (Deublein and Steinhauser, 2008):

$$C_cH_hO_oN_nS_s + y\ H_2O \rightarrow x\ CH_4 + (c\text{-}x)CO_2 + n\ NH_3 + s\ H_2S \quad (1)$$

where x = 1/8 * (4c + h -20 – 3n – 2s) and y = ¼ * (4c – h – 20 + 3n + 3s).
The above equation can be used to estimate the theoretical methane (CH_4) yield, if the chemical composition of the substrate is known. Primary sludge substrate can be approximated as $C_{10}H_{19}O_3N$, and waste activated sludge (biomass) can be approximated as $C_5H_7O_2N$. The overall process in Eq. 1 can be broken down into stages:

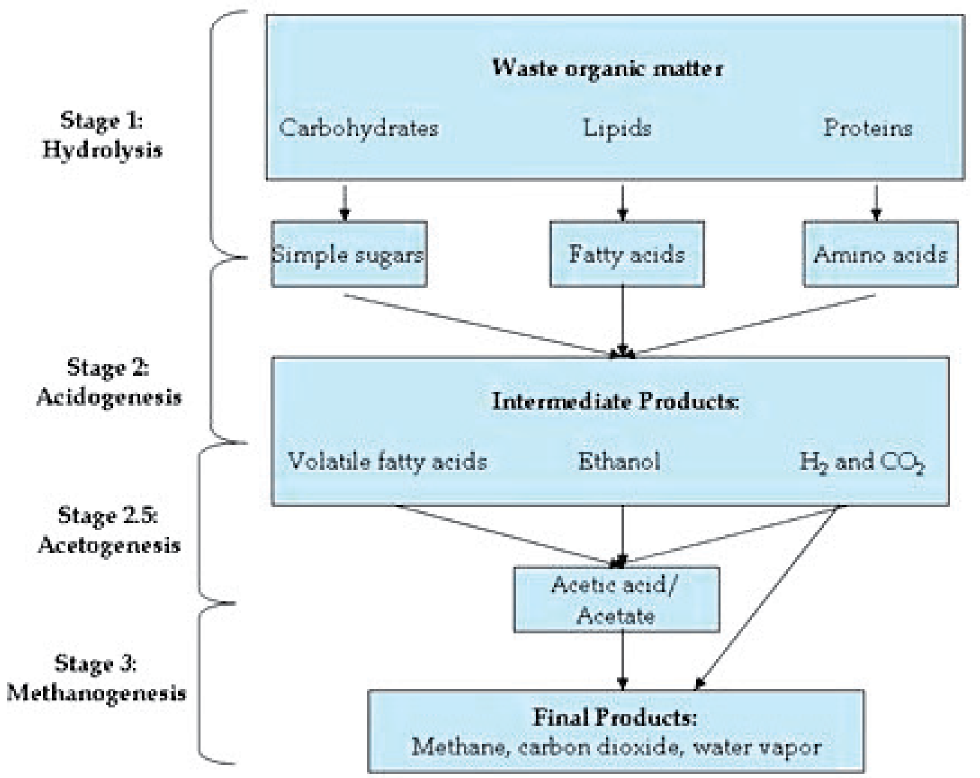

Fig. 1. Anaerobic digestion of organic matter (adapted from Khanal, 2008)

- Stage 1: Polymer Breakdown (Hydrolysis) – carbohydrates, lipids, and proteins are broken down into soluble monomers;
- Stage 2: Acid Production (Acidogenesis) – soluble monomers are converted to volatile fatty acids (lactic, propionic, and butyric acids);
- Stage 2.5: Acetic Acid Production (Acetogenesis) - Volatile fatty acids are converted to acetic acid;
- Stage 3: Methane Production (Methanogenesis) – Acetic acid is converted to methane; carbon dioxide and hydrogen are also converted to methane.

Figure 1 shows a schematic of the overall process of anaerobic digestion of organic matter. The stages are now discussed in more detail.

2.1 Stage 1: Polymer breakdown (hydrolysis)

The primary components of waste organic matter are carbohydrates, lipids, and proteins, as shown in Figure 1. In Stage 1, these components are broken down by cellulolytic, lipolytic, and proteolytic bacteria, respectively, into soluble monomers via hydrolysis (NAS, 1977). In hydrolysis, covalent bonds are split in a chemical reaction with water, as shown in Fig. 2 below.

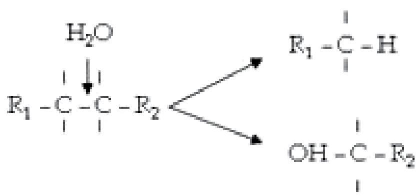

Fig. 2. Hydrolysis (adapted from Deublein and Steinhauser, 2008)

Bacteria of genera *Bacteroides, Lactobacillus, Propioni-bacterium, Sphingomonas, Sporobacterium, Megasphaera, Bifidobacterium* are most common in hydrolysis, including both facultative and obligatory anaerobes. More details concerning bacteria involved in hydrolysis, as well as the subsequent stages of anaerobic digestion, are given by Deublein and Steinhauser (2008).

The rate of hydrolysis is typically described using first-order kinetics according to (Sharma, 2008):

$$r_h = dC_x/dt = -k_h * C_x \qquad (2)$$

where

r_h = rate of hydrolysis, mass/(unit volume * time)

C_x = concentration of hydrolysable substrate x in the reactor, mass/volume

k_h = hydrolysis rate constant, time^{-1}

k_h depends on the specific substrate and temperature. This stage can be rate-limiting for difficult-to-degrade wastes (containing lipids and/or a significant amount of particulate matter, such as sewage sludge, animal manure, and food waste) (Henze and Harremos, 1983; van Haandel and Lettinga, 1994).

2.2 Stage 2: Acid production (acidogenesis)

In Stage 2, acid-forming bacteria (acidogens) convert the products of Stage 1, the soluble monomers, into short-chain organic acids (volatile fatty acids with C>2, such as lactic,

propionic, and butyric acids) (Khanal, 2008). Alcohols such as ethanol, hydrogen (H_2), and carbon dioxide (CO_2) are also produced.
The acid formers include both facultative and obligate anaerobic fermentative bacteria, including Clostridium spp., Peptococcus anaerobus, Bifidobacterium spp., Desulphovibrio spp., Corynebacterium spp., Lactobacillus, Actinomyces, Staphylococcus, and Esherichia coli (Metcalf & Eddy, 2004). Deublein and Steinhauser (2008) provide examples of degradation pathways.
The rate of growth of the acidogens can be described according to (Metcalf & Eddy, 2004; Sharma, 2008):

$$r_g = \mu X \tag{3}$$

where
r_g = rate of bacterial growth, mass/(unit volume * time)
μ = specific growth rate, time^{-1}
X = concentration of microorganisms, mass/unit volume
The microbial specific growth rate μ can be described via Monod kinetics (Metcalf & Eddy, 2004; Sharma, 2008):

$$\mu = \mu_{max} * S/(K_S + S) \tag{4}$$

where
μ = specific growth rate, time^{-1}
μ_{max} = maximum specific growth rate, time^{-1}
S = concentration of growth-limiting substrate in solution, mass/unit volume
K_S = substrate affinity constant, or half velocity constant, which represents the substrate concentration at which the growth rate becomes one half of the maximum growth rate, mass/unit volume
Substituting Eq. 4 into Eq. 3 gives:

$$r_g = \mu_{max} * X * S/(K_S + S) \tag{5}$$

The relationship between the rate of soluble monomer (substrate) utilization and rate of growth of the acidogens is given by (Metcalf & Eddy, 2004; Sharma, 2008):

$$r_g = -Yr_{su} \tag{6}$$

where
Y = maximum yield coefficient, mg/mg (defined as the ratio of the mass of cells formed to the mass of substrate consumed)
r_{su} = substrate utilization rate, mass/(unit volume * time)
The substrate utilization rate r_{su} can then be written as:

$$r_{su} = -r_g/Y = -\mu_{max} * X * S/[Y * (K_S + S)] \tag{7}$$

μ_{max}/Y is often replaced by k_m, defined as the maximum rate of substrate utilization per unit mass of microbes. r_{su} is then:

$$r_{su} = -k_m * X * S/(K_S + S) \tag{8}$$

A system of mass balance equations for each substrate and type of microorganism can be solved to obtain substrate and biomass concentrations as functions of time. More detail is provided by Sharma (2008). Monod kinetics can also be used to describe microbe growth and utilization of substrates in Stages 2.5 and 3.

2.3 Stage 2.5: Acetic acid production (acetogenesis)

In Stage 2.5, acetogenic microbes convert the volatile fatty acids and ethanol formed in Stage 2 into acetic acid (CH_3COOH)/acetate (CH_3COO^-), H_2, and CO_2. Examples include (Dolfing, 1988):

$$\text{Propionate} \rightarrow \text{acetate} \quad (\Delta G° = +76.1 \text{ kJ})$$

$$CH_3CH_2COO^- + 3H_2O \rightarrow CH_3COO^- + H^+ + HCO_3^- + 3H_2 \tag{9}$$

$$\text{Butyrate} \rightarrow \text{acetate} \quad (\Delta G° = +48.1 \text{ kJ})$$

$$CH_3CH_2CH_2COO^- + 2H_2O \rightarrow 2CH_3COO^- + H^+ + 2H_2 \tag{10}$$

$$\text{Ethanol} \rightarrow \text{acetate} \qquad (\Delta G° = +9.6 \text{ kJ})$$

$$CH_3CH_2OH + H_2O \rightarrow CH_3COO^- + H^+ + 2H_2 \tag{11}$$

The Gibb's free energy changes for the above reactions are positive, so the reactions are not thermodynamically favorable. However, in a co-culture of H_2-producing acetogenic bacteria and H_2-consuming methanogenic bacteria, a symbiotic relationship exists. Methanogenic bacteria keep the H_2 partial pressure low, which provides a thermodynamically favorable condition for formation of acetic acid/acetate (Khanal, 2008).

Acetic acid is also generated by homoacetogenic microbes, according to (Khanal, 2008):

$$2CO_2 + 4H_2 \rightarrow CH_3COOH + 2H_2O \tag{12}$$

Other homoacetogenic microbes can convert organic substrates such as formate and methanol into acetic acid/acetate according to (Khanal, 2008):

$$4COOH \rightarrow CH_3COOH + 2CO_2 + 2H_2O \tag{13}$$

$$4CH_3OH + 2CO_2 \rightarrow 3CH_3COOH + 2CO_2 \tag{14}$$

The homoacetogenic mesophilic bacteria *Clostridium aceticum* and *Acetobacterium woodii* have been isolated from sewage sludge (Novaes, 1986).

2.4 Stage 3: Methane production (methanogenesis)

Methanogenic bacteria, strictly anaerobic, can use the acetic acid/acetate from Stage 2.5 to form methane, according to:

$$CH_3COOH \rightarrow CH_4 + CO_2 \tag{15}$$

The above reaction accounts for up to about 70% of methane produced from anaerobic processes (NAS, 1977; Gujer and Zehnder, 1983). Acetotrophic (also called acetogenic or aceticlastic) methanogens, including bacteria from the genera *Methanosarcina* and

Methanosaeta, perform the conversion (Khanal, 2008). At high substrate concentrations, *Methanosarcina* will dominate. By converting acetic acid to the gaseous products CH_4 and CO_2, Stage 3 reduces the oxygen demand (BOD, COD) of the remaining waste.
Some species, called hydrogenotrophic methanogens, can produce methane from the CO_2 and H_2 formed as products in previous stages, according to:

$$4CO_2 + H_2 \rightarrow CH_4 + 2H_2O \tag{16}$$

The above reaction accounts for up to 30% of methane produced from anaerobic processes (NAS, 1977; Khanal, 2008).
Methanogens can also use formic acid (methanoic acid, HCOOH), methanol (CH_3OH), and methylamines ($(CH_3)_3N$) as substrates, according to (Metcalf and Eddy, 2004):

$$4HCOOH \rightarrow CH_4 + 3CO_2 + 2H_2O \tag{17}$$

$$4CH_3OH \rightarrow 3CH_4 + CO_2 + 2H_2O \tag{18}$$

$$4(CH_3)_3N + H_2O \rightarrow 9CH_4 + 3CO_2 + 6H_2O + 4NH_3 \tag{19}$$

Besides *Methanosarcina* and *Methanosaeta*, other genera of methanogens include *Methanobacterium, Methanobacillus, Methanococcus*. Methanogens have very low growth rates; their metabolism is usually rate-limiting in anaerobic treatment processes (Metcalf and Eddy, 2004). Long detention times are thus required, which has historically been a drawback of anaerobic processes compared with aerobic ones, although methods of overcoming this drawback have been developed in the last decade (Metcalf and Eddy, 2004). However, since only a small portion of the organic waste is synthesized into new cells, the amount of cells that must be wasted is small, which is an advantage (Metcalf and Eddy, 2004).
Not all of the carbon dioxide produced during Stage 3 (or Stage 2) is released as gas, since it is water soluble. CO_2 in solution reacts with hydroxyl ion (OH^-) to form bicarbonate:

$$CO_2 + OH^- \rightarrow HCO_3^- \tag{20}$$

The bicarbonate concentration depends on alkalinity, temperature, and the presence of other materials in the liquid phase. Conditions favoring bicarbonate production will lower the percent CO_2 in the gas phase, and increase the percent methane. Bicarbonate acts to buffer the solution pH (NAS, 1977).

3. Methane production: Quantities and rates

Factors associated with the waste impact the ultimate quantity of methane that can be produced. Factors associated with the waste, as well as the environment and reactor design, impact how fast methane is produced. This section discusses the impact of waste and environmental factors on methane production. The impact of reactor design will be discussed in the “Gas Production System Design” section. This section also discusses models for estimating methane production, as well as experimentally measured values of methane production.

3.1 Waste factors impacting methane production

Various factors associated with the waste impact both the quantity and rate of methane production:

- Waste composition/degradable organic content
- Particle size
- Organic loading rate ($kg/(m^3*d)$)

The maximum quantity of methane that can be produced depends on waste composition, and in particular the degradable organic content. The theoretical maximum methane yield can be estimated from Eq. 1 above; however, all of the organic content may not actually be able to be degraded by the bacteria. Degradability of the substrate decreases as lignin content increases. The practical amount of methane that can be generated from various wastes is given in Table 2 in Section 3.4.

Waste composition and particle size (along with the environmental factors moisture content, ambient temperature, and pH) have been observed to impact methane production rates. The smaller the waste grain size, the faster methane will be produced, since increased surface area is exposed to bacterial attack. Shredding the waste can increase the rate of methane formation, particularly for wastes with a high content of structural materials (e.g. cellulose, lignin), which make it difficult for microbes to access and degrade the substrate. The yield for substrates like hay and foliage can be increased by up to 20% by shredding (Deublein and Steinhauser, 2008). Shredding should be considered for large lumps of excrement, green cuttings, straw, and other agricultural residuals. Slow-running multiple screw mills, used also in composting technology, are often used.

The methane quantity generated also depends on organic loading rate, as will be discussed in the "Gas Production System Design" section.

3.2 Environmental factors impacting methane production

Environmental factors impacting the rate of methane generation include:

- Temperature
- pH
- Moisture content
- Nutrient content
- Concentration of toxic substances

Each of these factors will be discussed in turn.

3.2.1 Temperature

Anaerobic systems can be designed for temperatures appropriate for mesophilic bacteria (30-40°C) or thermophilic bacteria (50-60°C). Higher temperatures increase microbial activity, with activity roughly doubling for every 10°C increase within the optimal range (Khanal, 2008). Thermophilic systems thus produce methane 25-50% faster, depending on the substrate (Henze and Harremoes, 1983). Below 15°C, almost no methane will be generated (FAO, 1984). The digestion rate temperature dependence can be expressed using the Arrhenius equation (Khanal, 2008):

$$r_t = r_{30}\,(1.11)^{(t-30)} \qquad (21)$$

where

t = temperature in °C

r_t, r_{30} are digestion rates at temperature t and 30°C, respectively.

Operating systems in the thermophilic range improves pathogen destruction. However, start-up is slower, and systems are more susceptible to changes in loading variations, substrate, or toxicity (Khanal, 2008).

3.2.2 pH

Acidogens prefer pH 5.5-6.5; methanogens prefer 7.8-8.2. When both cultures coexist, the optimal pH range is 6.8-7.5 (Khanal, 2008). If the pH drops below 6.6, methanogens are significantly inhibited, and pH below 6.2 is toxic (Metcalf and Eddy, 2004). When acid-forming bacteria of Stage 2 and methanogenic bacteria of Stage 3 have reached equilibrium, the pH will naturally stabilize around 7, since organic acids will be removed as they are produced, unless a problem develops. Normally, alkalinity in anaerobic systems ranges from 1000 to 5000 mg/L, which provides sufficient buffering to avoid large drops in pH (Metcalf and Eddy, 2004).

3.2.3 Moisture content

Many landfill studies have confirmed that methane generation rate increases as waste moisture content increases (Barlaz et al., 1990; Chan et al., 2002; Chugh et al., 1998; Faour et al., 2007; Filipkowska and Agopsowicz, 2004; Gawande et al., 2003; Gurijala and Suflita, 1993; Mehta et al., 2002; Tolaymat et al., 2010; Vavilin et al., 2004; Wreford et al., 2000) . In many anaerobic systems, the digester is fed a water/waste mixture called slurry, as discussed in "Gas Production System Design". As long as typical rules of thubfor water addition are followed, moisture content does not limit methane production.

3.2.4 Nutrient content

Methanogens require macronutrients P and N, as well as micronutrients. The amount of P and N required can be calculated by assuming the empirical formula for a bacterial cell to be $C_5H_7O_2N$ (Speece and McCarty, 1964). P and N requirements can also be estimated using COD/N/P ratios, with a minimum ratio of 350:7:1 COD/N/P needed for highly loaded systems (0.8-1.2 kg COD/(kg VSS*day), and a minimum ratio of 1000:7:1 COD/N/P needed for lightly loaded systems (<0.5 kg COD/(kg VSS*day) (Henze and Harremoes, 1983). Phosphoric acid or phosphate salts are commonly used to supply needed additional phosphorous, and urea, aqueous ammonia, or ammonium chloride are used to supply nitrogen (Khanal, 2008).

Trace metals that have been found to enhance methane production include iron, cobalt, molybdenum, selenium, calcium, magnesium, sulfide zinc, copper, manganese, tungsten, and boron in the mg/L level and vitamin B_{12} in μg/L (Speece, 1988).

3.2.5 Toxic substance concentration

High levels of ammonia, soluble sulfides, soluble salts of metals, and alkali and alkaline-earth metal salts in solution (e.g. those of sodium, potassium, calcium, or magnesium) can be toxic to methanogens (NAS, 1977). Maximum allowable concentrations of various substances are given in Table 1 below. In addition, the methanogens are strict anaerobes; thus, their growth is inhibited by even small amounts of oxygen, or highly oxidized material (like nitrates).

3.3 Models for estimating methane production

Anaerobic Digestion Model No. 1 (ADM1), published by the International Water Association, provides a generic model and common platform for dynamic simulations of a variety of anaerobic processes. The model can be used as a tool for research, design, operation and optimization of anaerobic processes. It can be used for a variety of

Constituent	Maximum Recommended Concentration
Ammonia (NH_3)	1500-3000 mg/L
Calcium (Ca)	2500-4500 mg/L
Chromium (Cr)	200 mg/L
Copper (Cu)	100 mg/L
Cyanide (CN^{-})	<25 mg/L
Magnesium (Mg)	1000-1500 mg/L
Nickel (Ni)	200-500 mg/L
Potassium (K)	2500-4500 mg/L
Sodium (Na)	3500-5500 mg/L
Sodium chloride (NaCl)	40,000 ppm
Sulfate ($SO_4{}^{2-}$)	5000 ppm

Table 1. Maximum recommended concentrations of toxic substances in anaerobic slurries (adapted from OLGPB, 1976)

applications, from domestic (wastewater and sludge) treatment systems to specialized industrial applications. Outputs from the model include gas flow and composition, pH, separate organic acids, and ammonium.
Other specialized models for estimating methane production are available. For example, the U.S. Environmental Protection Agency's (EPA's) LandGEM "Landfill Gas Emission Model" and the IPCC CH_4 generation model are two of the most widely used models for estimating methane generation from landfills.

3.4 Experimental measurements of methane production

Maximum biogas yields for a variety of common materials are given in Table 2. The yields are maximum specific yields of biogas for a given waste, or q_{waste} (maximum biogas produced per total organic solids, volume/mass). Although wood is organic, it is not listed because lignin, the main component of wood, degrades slowly. When values in the table are not given, they could be estimated from similar type wastes. Deublein and Steinhauser (2008) provide biogas yields for additional categories of substrates.
Typically, biogas is 60%-70% methane and 30-40% CO_2 (NAS, 1977; Biogas). The fraction of methane in the biogas increases as the number of C-atoms in the substrate increases (Deublein and Steinhauser, 2008).
For economic reasons, biogas reactors are designed so that 75% of the maximum degradable organic matter is actually decomposed (Deublein and Steinhauser, 2008). This means that the maximum yield values from Table 2 should be multiplied by 0.75 to give an estimate of the practical biogas yield. For large scale plants, laboratory tests using reactors of 4-8 L, and then a **pilot plant** with reactors of size >50L, **should be used to determine the practically attainable methane yield** and rate of gas production. More details are given in Deublein and Steinhauser (2008).

4. Gas production system design

This section focuses on design of larger-scale centralized biogas plants designed for energy generation in developed countries, such as Germany, and smaller-scale units, that may be used in rural areas of developing countries. In either case, the microbiology and design elements are the same. Metcalf and Eddy (2004) and Deublein and Steinhauser (2008) provide a thorough discussion of design of suspended-growth anaerobic digesters for treatment of high-strength industrial organic wastes and sludges from wastewater treatment plants, so these systems will not be discussed in detail here. Design of anaerobic systems specifically at landfills is also discussed elsewhere (e.g. Bagchi, 2004; Rushbrook and Pugh, 1999).

A complete anaerobic system for waste treatment and energy generation includes 3 major components:

- Gas production system
- Gas use system
- Sludge/liquid product use system

This section focuses on design of the gas production system, for which an example schematic is shown in Fig. 3. More information about design of the gas use system can be found in Deublein and Steinhauser (2008) and Khanal (2008). Steps in design of the gas production system include:

1. Determine biogas production requirements,
2. Select waste materials and determine feed rates; size waste storage; determine rate of water addition and size the preparation tank,
3. Design the digester/reactor,
4. Design the gas storage system,
5. Determine system location.

Each of these steps will now be discussed in detail.

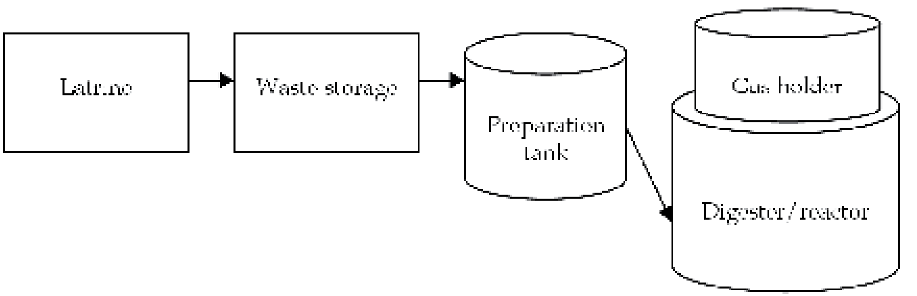

Fig. 3. Schematic of a small-scale biogas production system (adapted from NAS, 1977)

Substrate	Dry Matter (DM, %)	% of Dry Matter that is Organic (oDM)	Biogas Yield (m^3/kg oDM)	Hydraulic Retention time (days)	C/N
Wastes from households and gastronomy					
Bio waste	40-75	30-70	0.3-1.0	27	
Leftovers (overstored food)	14-18	81-97	0.2-0.5	10-40	
Sewage sludge (households)	5 (night soil)	3.4* (night soil)	0.20-0.75	35-45	2.9-6
Sewage sludge (industry)	--	--	0.30	20	
Flotation sludge	5-24	90-98	0.7-1.2	12	

Animal waste					
General manure from livestock	--	--	0.26-0.28	--	14 (farmyard avg)
Manure from cows	7-20	85-90	0.20-0.50 (per DM)	28-38	18-25
Manure from pigs	5-27.5	90	0.56	22-28	13
Manure from horses	--	--	0.2 – 0.3	--	24-25
Manure from poultry	15-75	75	0.31-0.54	17-22	
Manure from sheep	--	--	0.37-0.61 (per DM)	20	29
Cow dung	--	--	0.33 (per DM)	--	
Slaughterhouse waste	--	--	0.3-0.7	--	2
Animal fat	--	--	1.00	33	
Stomach content of pigs	12-15	80-84	0.3-0.4	62	
Greens, grass, vegetable wastes					
Vegetable wastes	5-20	76-90	0.4	8-20	
Leaves	--	82	0.6	8-20	41
Leaves from trees	--	--	0.210-0.294	--	
Grass cuttings from lawns	37	93	0.7-0.8	10	19
Market wastes	8-20	75-90	0.4-0.6	30	
Straw from cereals	86	89-94	0.2-0.5	--	128 (wheat straw)
Maize straw	86	72	0.4-1.0	--	53
Rice straw	25-50	70-95	0.55-0.62	--	67
Wastes from the food and fodder industry					
Potato pulp, potato peelings	6-18	85-96	0.3-0.9	3-10	25 (potato tops)
Mash from distillations	2-8	65-85	0.42	14	
Wheat flour	88	96	0.7	--	
Oilseed residuals (pressed)	92	97	0.9-1.0	--	
Cereal mash	6-8	83-90	0.9	3-10	
Wastes from other industries					
Egg waste	25	92	0.97-0.98	40-45	173
Waste from paper and carton production	--	--	0.2-0.3	--	
Pulp	13	90	0.65-0.75	--	

* % of total that is organic
Dry matter is equivalent to total solids.

Table 2. Maximum biogas yields and C/N of various substrates (adapted from Deublein and Steinhauser, 2008; OLGPB, 1976; NAS, 1977; Metcalf & Eddy, 2004)

4.1 Determine biogas production requirements

A biogas plant can be designed:

1. to process a given amount of waste material per time, or
2. to produce a given quantity of gas for specific uses.

The second objective is preferable for small-scale units, designed to serve an individual household or small village. In these cases, the biogas unit is the only source of power for the household or village; it is thus important to have sufficient gas available, and the waste can be stored. (NAS, 1977) The first objective could be considered for large-scale units, which are supplementing fossil fuel or other types of power plants for a region.

In the case of design for the second objective, which will be considered here, the engineer must first determine volume of gas needed per day (Q_{gas} = gas production rate, volume/day).

- The biogas may be used directly to power gas-burning appliances (hot water heating, building heating, room lighting, home cooking, refrigeration). In this situation, the engineer must itemize the biogas needed for intended applications and sum up the quantities of gas (see Table 3).
- If the biogas is to be used for electricity production, the thermal efficiency of the turbine must be known. The heating value of biogas is 500-700 Btu/ft^3 (for a gas that is 60%-70% methane and balance CO_2).
- If biogas is to be used to power an internal combustion engine (for water pumping, etc.), the engine efficiency must be known.

Use	**Specification**	**Quantity of Gas Required, m^3/hr**
Cooking	2″ burner	0.33
	4″ burner	0.47
	6″ burner	0.64
	2″-4″ burner	0.23-0.45
	Per person/day	0.34-0.42+
	Per person/day	0.34+
Gas lighting	Per lamp of 100 candle power	0.13
	Per mantle	0.07
	Per mantle	0.07-0.08
	2 mantle lamp	0.14
	3 mantle lamp	0.17
Gasoline or diesel engine	Converted to biogas, per hp (based on 25% efficiency)	0.45-0.51
Refrigerator	Per ft^3 capacity	0.028
	Per ft^3 capacity	0.034
Incubator	Per ft^3 capacity	0.013-0.017
	Per ft^3 capacity	0.014-0.020
Gasoline	1 liter	1.33-1.87
Diesel fuel	1 liter	1.50-2.07
Boiling water	1 liter	0.11

Table 3. Quantities of biogas required for specific applications (NAS, 1977)

Example 1

An anaerobic system is to be designed to provide energy for gas-burning appliances for a village with a population of 150, which live in 28 dwellings. Enough power should be provided for daily cooking, as well as for 2 lamps of 100 candle power per household burning 3 hours per day, and a 2-ft^3 capacity refrigerator for half of the households. Estimate the volume of digester gas that must be produced per day to provide power for the village.

Solution

From Table 3, an average of 0.38 m^3 gas/person/day is needed for cooking. Lamps of 100 candle power require 0.13 m^3 gas/hr, and refrigerators require on average 0.031 m^3 gas/hr/ft^3 capacity. The total biogas required for the village would thus be:
150 persons * 0.38 m^3 gas/person/day
+ 2 lamps/household * 28 households * 0.13 m^3 gas/lamp/hr * 3 hrs/day
+ 1 refrigerator/household * 14 households * 0.031 m^3 gas/hr/ft^3 capacity * 2 ft^3 average capacity/refrigerator * 24 hours/day
= 99.7 m^3 gas/day

4.2 Select waste materials and determine feed rates; size waste storage and preparation facilities

4.2.1 Select waste materials and determine feed rates

The engineer must determine the amount of waste ($\dot{M}_{waste}$ = waste feed rate, mass/day) needed to produce the biogas estimated in 4.1. The amount of available materials should first be inventoried. Materials to be used as reactor feed should be chosen based on:

- Local availability
- Biogas yield
- Required processing/pre-treatment
- C/N ratio
- Cost

Potential feedstocks for larger-scale biogas plants include municipal sewage water, sewage sludge, industrial wastewater, grass from lawns and other material from landscaping, organic wastes from households, waste from dairies or slaughterhouses, liquid manure, agricultural wastes, organic waste from industry, byproducts from food production. Feedstocks for rural biogas plants include excrement, liquid manure, straw, organic wastes from households, byproducts from agriculture and food production. Although crops can be grown expressly for biogas energy generation, these crops should be chosen carefully to avoid competition with potential food crops.
Quantities of manure provided by various animals are given in Table 4. Maximum biogas yields for a variety of common materials were given in Table 2. The potential biogas generated from various feedstocks can be estimated according to:

$$Q_{waste\ i} = q_{waste\ i} * \dot{M}_{waste\ i} * f_{TS} * f_{oTS} * 0.75 \quad (22)$$

where
$Q_{waste\ i}$ = gas production rate (volume/day) from waste i
$q_{waste\ i}$ = maximum specific yield of biogas for waste i (maximum biogas produced per organic total solids, volume/mass)
$\dot{M}_{waste\ i}$ = waste feed rate (mass/day) for waste i

f_{TS} = fraction of waste by weight that is solids
f_{oTS} = fraction of total solids by weight that are organic (volatile)
0.75 = factor to account for practical biogas yield
Note that the "Volatile solids" column of Table 4 gives $f_{TS} * f_{oTS}$.

Animal	**Daily manure per 500 kg live animal**		**Volatile solids, % of wet weight**	**Average weight per animal, kg**	**Daily manure per animal, wet weight, kg**
	Volume , m^3	**Wet weight, kg**			
Dairy cattle	0.038	38.5	7.98	450-650	34.7-50.1 (42.4 avg)
Beef cattle	0.038	41.7	9.33	485-554	40.4-46.2 (43.3 avg)
Swine	0.028	28.4	7.02	125-270	7.1-15.3 (11.2 avg)
Sheep	0.020	20.0	21.5	41-136 (female); 68-205 (male)	Female: 1.6-5.4 (3.5 avg) Male: 2.7-8.2 (5.5 avg)
Poultry	0.028	31.3	16.8	2-3 (chickens)	0.12 – 0.19 (0.16 avg)
Horses	0.025	28.0	14.3	380-1000	21.3-56 (38.6 avg)

Table 4. Daily manure production for various animals (NAS, 1977)

Required processing/pre-treatment for wastes may include adjusting water content (discussed in 4.2.3), shredding (discussed in 3.1), and/or removal of metals, plastics, glass, and sand. Feed materials should be chosen in combination so that their weighted average C/N ratio (by mass) is around 30, which has been shown through experience to be optimal (NAS, 1977). Since there are few common materials with a suitable C/N ratio, use of more than one source material is typically required (OLGPB, 1978). Although the amount of nitrogen needed is not large, the C/N ratio is important for efficient methane production. If the C/N ratio is too high, nitrogen availability limits the process; on the other hand, if the C/N ratio is too low, ammonia concentrations may become high enough to be toxic to the microorganisms. Supplementing substrates with a high C content with those containing N, and vice versa, can help maintain this ratio. C/N ratios of various wastes were shown in Table 2. The overall C/N ratio can be calculated from:

$$(C/N)_{overall} = \sum_{i=1}^{n} (C/N)_{waste\ i} * f_{waste\ i} \qquad (23)$$

where
$f_{waste\ i}$ = fraction of total waste feed that is waste i, by mass
n = total number of wastes
If $(C/N)_{overall}$ is not close to 30, the combination of waste feedstocks must be adjusted.

Example 2

For the scenario described in Example 1, locally available low-cost feedstocks include household sewage sludge/septage, dairy cow manure (28 cows), poultry manure (56 chickens), and rice straw. Determine waste feed rates for the various feedstocks that will generate the gas estimated in Example 1 with an acceptable overall C/N.

Solution

From Table 2, $q_{waste\ i}$, f_{TS}, f_{oTS}, and C/N for the 4 feedstocks are as follows:

Feedstock	$q_{waste\ i}$, m^3/kg oDM	f_{TS}	f_{oTS}	C/N
household sewage sludge/septage	0.20-0.75 (0.475 avg)	5%	47-83% (65% avg)	2.9-6 (4.5 average)
cow manure	0.2-0.5 (per DM) (0.375 avg)	7-20% (13.5% avg)	85-90% (87.5% avg)	18-25 (21.5 average)
poultry manure	0.31-0.54 (0.425 avg)	15-75% (45% avg)	75%	14 (farmyard avg)
rice straw	0.55-0.62 (0.585 avg)	25-50% (37.5 avg)	70-95% (82.5 avg)	67

We will assume initially that all of the septage, cow manure, and poultry manure are used to generate methane, in order to reduce pathogenic organisms. Rice straw will then be added, in a quantity to provide an overall C/N ratio for the waste mixture of 30.

Average septage (night soil, urine and feces) generation is 1.5 kg/person/day (wet weight) (Metcalf & Eddy, 2004). The mass of septage that would be available per day can thus be estimated as:

$$\dot{M}_{septage} = 1.5\ \text{kg/person/day} * 150\ \text{persons} = 225\ \text{kg/day}$$

The mass of cow and poultry manure that would be available per day can be estimated from Table 4, last column, as:

$$\dot{M}_{cow\ manure} = 42.4\ \text{kg/day} * 28\ \text{cows} = 1187\ \text{kg/day (wet weight)}$$

$$\dot{M}_{poultry\ manure} = 0.16\ \text{kg/day} * 56\ \text{chickens} = 9\ \text{kg/day (wet weight)}$$

Gas production for each waste can be calculated according to (22):

$$Q_{waste\ i} = q_{waste\ i} * \dot{M}_{waste\ i} * f_{TS} * f_{oTS} * 0.75$$

$$Q_{septage} = 0.475\ m^3\ \text{biogas/kg oDM} * 225\ \text{kg/day} * 0.05 * 0.65 * 0.75 = 2.6\ m^3\ \text{biogas/day}$$

$$Q_{cow\ manure} = 0.375\ m^3\text{/kg DM} * 1187\ \text{kg/day} * 0.135 * 0.75 = 45.1\ m^3\ \text{biogas/day}$$

$$Q_{poultry\ manure} = 0.425\ m^3\text{/kg} * 9\ \text{kg/day} * 0.45 * 0.75 * 0.75 = 1.0\ m^3\ \text{biogas/day}$$

Note that since $q_{waste\ i}$ for cow manure was given per DM instead of per oDM, f_{oTS} was not used in calculating $Q_{waste\ i}$.

The daily gas production from the septage, cow manure, and poultry manure would be 2.6 + 45.1 + 1.0 = 48.7 m³/day. Since the C/N for each of these wastes is less than 30, the overall C/N would be less than 30. To raise the overall C/N, a significant amount of rice straw, with C/N of 67, needs to be added. The mass of rice straw to be added can be estimated by first calculating the mass fraction of each waste $f_{waste\ i}$ as follows:

$$\dot{M}_{TOT\ waste} = \dot{M}_{septage} + \dot{M}_{cow\ manure} + \dot{M}_{poultry\ manure} + \dot{M}_{rice\ straw} = (225+1187+9)\ kg/day + \dot{M}_{rice\ straw} = 1421\ kg/day + \dot{M}_{rice\ straw}$$

$$f_{rice\ straw} = \dot{M}_{rice\ straw} / \dot{M}_{TOTAL\ waste} = \dot{M}_{rice\ straw} / (1421\ kg/day + \dot{M}_{rice\ straw})$$

$$f_{rseptage} = \dot{M}_{septage} / (1421\ kg/day + \dot{M}_{rice\ straw}) = 225\ kg/day / (1421\ kg/day + \dot{M}_{rice\ straw})$$

$$f_{cow\ manure} = \dot{M}_{cow\ manure} / (1421\ kg/day + \dot{M}_{rice\ straw}) = 1187\ kg/day / (1421\ kg/day + \dot{M}_{rice\ straw})$$

$$f_{polutry\ manure} = \dot{M}_{poultry\ manure} / (1421\ kg/day + \dot{M}\ rice_{straw}) = \\ = 9\ kg/day / (1421\ kg/day + \dot{M}_{rice\ straw})$$

Now, Eq. 23 can be solved to yield the mass of rice straw to be added per day, as follows:

$$(C/N)_{overall} = (C/N)_{rice\ straw} * f_{rice\ straw} + (C/N)_{septage} * f_{septage} + (C/N)_{cow\ manure} * f_{cow\ manure} + (C/N)_{poultry\ manure} * f_{poultry\ manure}$$

$$30 = 67 * \dot{M}_{rice\ straw} / (1421\ kg/day + M_{rice\ straw}) + 4.5 * 225\ kg/day / (1421\ kg/day + \dot{M}_{rice\ straw}) + 21.5 * 1187\ kg/day / (1421\ kg/day + \dot{M}_{rice\ straw}) + 14 * 9\ kg/day / (1421\ kg/day + \dot{M}_{rice\ straw})$$

$$30 * (1421\ kg/day + \dot{M}_{rice\ straw}) = 67 * \dot{M}_{rice\ straw} + 4.5 * 225\ kg/day + 21.5 * 1187\ kg/day + 14 * 9\ kg/day$$

$$42{,}630\ kg/day + 30\ \dot{M}_{rice\ straw} = 67 * \dot{M}_{rice\ straw} + 26{,}659\ kg/day$$

$$37\ \dot{M}_{rice\ straw} = 15{,}971\ kg/day$$

$$\dot{M}_{rice\ straw} = 432\ kg/day$$

If 432 kg/day of rice straw is not available, maize straw or wheat straw, also with high C/N ratios, could be added. The 432 kg/day of rice straw would provide additional methane generation as follows:

$$Q_{rice\ straw} = 0.585\ m^3/kg * 432\ kg/day * 0.375 * 0.825 * 0.75 = 58.6\ m^3\ biogas/day$$

The total gas production from all 4 wastes would be 48.7 + 58.6 = 107.3 m^3 biogas/day, which is greater than the amount needed to power the village (99.7 m^3 biogas/day), as calculated in Example 1. If the actual gas generated had been less than the amount required, additional feedstocks would need to have been found, or the demand for gas reduced.

4.2.2 Size waste storage

Storage is often required before feedstock wastes are added to the main digester reactor. Plant materials in particular should be left outside to rot for 10 days before being put in the digester (FAO, 1984). Prolonged storage, however, should be avoided if manure is present so that flies do not breed.

The storage shed can be constructed of local materials such as bamboo, tree limbs, and palm thatch, or bricks, tiles, and mortar, depending on available materials. Rain will leach a significant portion of soluble material from manure and excrement, so storage should be covered (NAS, 1977). The volume of storage required can be estimated from:

$$V_s = \sum_{i=1}^{n} \dot{M}_{waste\ i} / \rho_{waste\ i} * t_i \qquad (24)$$

where

V_s = total volume of storage

$\dot{M}_{waste\ i}$ = waste feed rate (mass/day) for waste i

$\rho_{waste\ i}$ = density of waste i (mass/volume)

t_i = maximum number of days for which storage is desired for waste i

4.2.3 Determine rate of water addition, and size the preparation tank

In many cases, water must be added to the waste to make a slurry. Pure or slightly contaminated water may be added. An overall water content of 75-90%, with 10-25% dry matter, is recommended (Deublein and Steinhauser, 2008). If too little water is added, acetic acid will accumulate, and scum will form on the liquid surface in the digester. The higher the water content, the more CO_2 dissolves in the liquid phase, reducing the CO_2 content of the gas phase, and thus increasing the methane content (Deublein and Steinhauser, 2008). However, if too much water is added, the rate of production per unit volume in the digester will fall.

The volume of water to be added per day (Q_{water}, mass/day) can be estimated given the water content of the fermentation materials, as follows:

$$0.90 = \dot{M}_{water} / \dot{M}_{TOTAL} \qquad (25)$$

where 0.90 = desired water content (can range from 0.75 to 0.90), and

$$\dot{M}_{water} = \sum_{i=1}^{n} \dot{M}_{waste\ i} (1 - f_{TS\ waste\ i}) + Q_{water} * \rho_{water} \qquad (26)$$

$$\dot{M}_{TOTAL} = \sum_{i=1}^{n} \dot{M}_{waste\ i} + Q_{water} * \rho_{water} \tag{27}$$

Solving for Q_{water} gives:

$$Q_{water} = \{9 * \sum_{i=1}^{n} \dot{M}_{waste\ i} - 10 * [\sum_{i=1}^{n} \dot{M}_{waste\ i} (1 - f_{TS\ waste\ i})]\}/\rho_{water} \tag{28}$$

Water can be added in a preparation tank ahead of the digester. The preparation tank also serves as an equilibrating/mixing chamber, which homogenizes the substrate. The preparation tank is often a vertical concrete cylinder, or a cylindro-conical standing vessel (Deublein and Steinhauser, 2008). The required volume of the preparation tank can be determined from:

$$V_{PT} = \dot{M}_{TOTAL} * t_{PT}/\rho_{water} * 1.25 \tag{29}$$

where t_{PT} is the desired residence time in the tank and 1.25 is a factor to account for air and fixtures (Deublein and Steinhauser, 2008).

The overall density of the waste slurry is assumed to be approximately equal to that of water, since the water content by weight is 90%. The preparation tank dimensions can then be determined according to the rule of thumb: $H_{PT} = 2 * D_{PT}$, where H_{PT} = height of the preparation tank and D_{PT} = diameter of the preparation tank.

Example 3

Continuing with the information from Examples 1 and 2, estimate the water that must be added to the waste mixture to form a slurry, and size the preparation tank. Assume that the desired t_{PT} is 7 days.

Solution

According to Eq. 28,

$$Q_{water} = \{9 * [\dot{M}_{septage} + \dot{M}_{cow\ manure} + \dot{M}_{poultry\ manure} + \dot{M}_{rice\ straw}] - 10 * [\dot{M}_{septage} (1 - f_{TS\ septage}) +$$

$$\dot{M}_{cow\ manure} (1 - f_{TS\ cow\ manure}) + \dot{M}_{poultry\ manure} (1 - f_{TS\ poultry\ manure}) + \dot{M}_{rice\ straw} (1 - f_{TS\ rice\ straw})]\}/\rho_{water}$$

$$Q_{water} = \{9 * [225+1187+9+432]kg/day - 10 * [225\ kg/day\ (1 - 0.05) + 1187\ kg/day\ (1 - 0.135) + 9\ kg/day\ (1 - 0.45) +$$

$$432\ kg/day\ (1 - 0.375)]\}/(1\ g/mL) * 1000\ g/kg * 1L/(1000\ mL)$$

$$Q_{water} = 1522\ L/day$$

From Eq. 27,

$$\dot{M}_{TOTAL} = \sum_{i=1}^{n} \dot{M}_{waste\ i} + Q_{water} * \rho_{water}$$

$$\dot{M}_{TOTAL} = [225+1187+9+432]\text{kg/day} + 1522\ \text{L/day} * 1\ \text{g/mL} * 1000\ \text{mL/L} * 1\ \text{kg/(1000g)}$$

$$\dot{M}_{TOTAL} = 3375\ \text{kg/day}$$

According to Eq. 29,

$$V_{PT} = \dot{M}_{TOTAL} * t_{PT}/\rho_{water} * 1.25$$

$$V_{PT} = 3375\ \text{kg/day} * 7\ \text{days} /(1\ \text{kg/L}) * 1.25$$

$$V_{PT} = 29{,}531\ \text{L} = 29.5\ \text{m}^3$$

Assuming that the preparation tank is cylindrical, $V_{PT} = H_{PT} * \pi * D_{PT}^2/4$. Assume $H_{PT} = 2 * D_{PT}$. Then,

$$V_{PT} = 2 * D_{PT} * \pi * D_{PT}^2/4 = \pi * D_{PT}^3/2$$

Solving for D_{PT} gives: $D_{PT} = [2 * V_{PT} / \pi]^{1/3} = [2 * 29.5\ \text{m}^3 / \pi]^{1/3} = 2.66$ m, and $H_{PT} = 5.32$ m

4.3 Design the digester/reactor

Steps in design of the digester/reactor are:

a. Choose flow of wastes through the reactor (batch or continuous) and reactor configuration
b. Choose reactor material
c. Size reactor
d. Choose mixing method
e. Determine heating requirements

Each of these steps will now be discussed in turn.

4.3.1 Choose flow of wastes through the reactor (batch or continuous) and reactor configuration

4.3.1.1 Small systems: Batch systems

Most small systems in rural areas are operated as batch systems. In a batch process, the digester is completely filled all at once. The substrate degrades without anything being added or discharged until the end of the residence time (typically around 3 months) (NAS, 1977). Batch systems can be completely mixed or not mixed. Gas production increases until it reaches a maximum at about half the residence time. At the end of the residence time, most of the residue is emptied into the storage tank, with only small amounts remaining to inoculate the next load (Deublein and Steinhauser, 2008). The system is then cleaned. In a batch system, it is desirable to have 2 digesters so that one can always be in operation (NAS, 1977).

Batch processes have advantages over continuous flow processes in that the reactors are cheaper, and the systems are easier to operate. Since all parts of the substrate have the same residence time, the risk of pathogenic organisms exiting the system is reduced. NAS (1977) discusses other options for flow of materials for anaerobic digesters in rural areas, including semi-continuous plug flow systems and digesters with compartments.

A batch system can have a circular or rectangular cross-section (OLGPB, 1978). OLGPB (1978) gives more details about inlets, outlets, separation walls, and gas outlet pipes for batch systems.

4.3.1.2 Large systems: Continuous flow systems

Most larger systems are operated in continuous flow mode. Continuous flow systems have high volume yields, continuously produce a consistent quality and quantity of biogas, and frequently achieve a higher degree of decomposition.

Hydraulic retention time (HRT, also called hydraulic residence time or hydraulic detention time) indicates the time the slurry containing the waste remains in the reactor in contact with the biomass. HRTs can be shorter for simpler wastes that are easily biodegradable, but must be longer for more complex wastes that are more difficult for microbes to metabolize (Khanal, 2008).

Since anaerobes grow slowly, solids retention time is important in continuous flow systems. Solids retention time (SRT, also called mean cell residence time) controls the microbial mass in the reactor. The SRT must be at least 10-15 days for methanogens, which reproduce slowly. Maintaining a high SRT produces more stable operation and better toxic load resistance (Khanal, 2008).

High SRTs can be achieved by simply using a long HRT and SRT, but this leads to a large reactor. Alternatively, approaches that decouple the HRT from the SRT can be used, via separating and recirculating a portion of the microbes/solids, or immobilizing the biomass. Such approaches allow a high SRT to be maintained, thus preventing washout of slow-growing anaerobes, yet allow reduction in reactor size. 4 approaches for decoupling HRT and SRT are (Khanal, 2008):

1. **Biomass immobilization in attached growth systems** – anaerobes attach to support media (plastic, gravel, sand, or activated carbon) to form biofilms. Examples: *anaerobic filter*, rotating anaerobic contactor; expanded/*fluidized bed reactor*.
2. **Granulation and floc formation** – anaerobic microbes agglomerate to form granules and flocs that will settle well in the bioreactor. Examples: *upflow anaerobic sludge blanket reactor*, static granular bed reactor, *anaerobic sequencing batch reactor*; anaerobic baffled reactor.
3. **Biomass recycling** – Feed with high suspended solids (e.g. wood fiber) enables microbes to attach to solids, forming settlable flocs which are then recycled back to the reactor. Example: *anaerobic contact reactor*/clarigester
4. **Biomass retention** – Membrane integration into reactor retains biomass. Example: *Anaerobic membrane bioreactor*.

For high-solids feed streams, a completely-stirred tank reactor (CSTR) must often be used, which means that HRT=SRT. Pretreatment can reduce the detention time and enhance bioenergy production.

Several types of continuous flow systems are now discussed. Metcalf and Eddy (2004) provides more detailed design considerations for the various reactor types. An additional comparison of common anaerobic reactor designs that decouple HRT and SRT is given in Sattler (2011).

4.3.1.2.1 Anaerobic filter

An example of an attached growth system, anaerobic filters contain a support media on which biomass grow in a biofilm, as shown in Figure 4a. Common support media include synthetic plastic or ceramic tiles, with a high void volume and specific surface area. Since the media retains biomass, a long SRT (on the order of 100 days) can be maintained regardless of HRT. Typically, HRT ranges from 0.5 to 4 days and loading rate ranges from 5 to 15 kg COD/m^3/day. Biomass may need to be wasted periodically to prevent clogging. The configuration can be upflow or downflow (Khanal, 2008).

4.3.1.2.2 Fluidized bed reactor

Another type of attached growth system, a fluidized bed reactor (see Figure 4b) contains microbes attached to biocarriers, such as sand, granular activated carbon, shredded tire, or synthetic plastic media. The biocarriers are expanded by the upflow velocity of feed, which must be 10-25 m/h. Its expansion is 25-300% of its settled volume (Khanal, 2008).

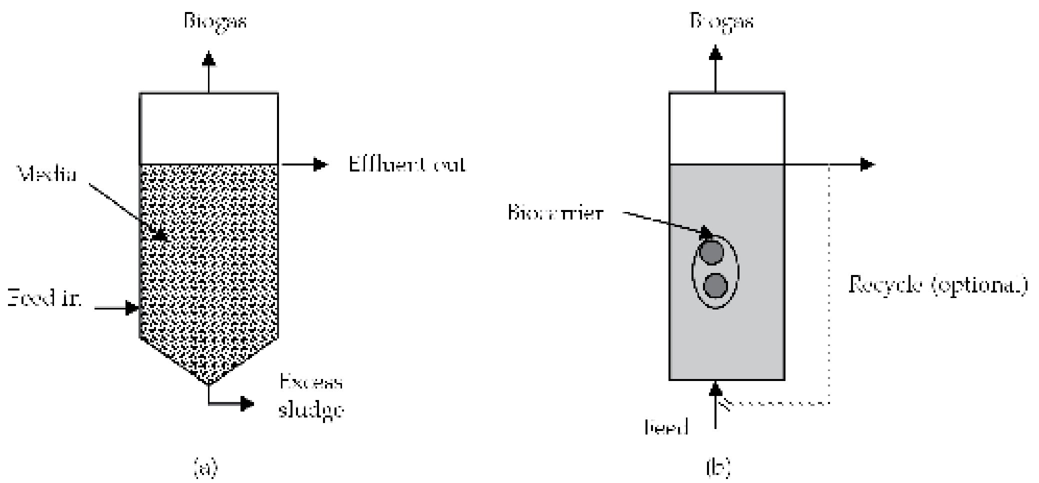

Fig. 4. Schematics of (a) upflow anaerobic filter and (b) fluidized bed reactor (after Khanal, 2008)

4.3.1.2.3 Upflow anaerobic sludge blanket

A type of suspended growth system, an upflow anaerobic sludge blanket reactor (UASB), shown in Figure 5, promotes formation of dense biomass aggregates known as granules, via proper hydraulic and organic loading conditions. Granule diameters range from 1-3 mm, and settle with velocities around 60 m/h. Since the superficial upflow velocity of the waste stream is maintained at <2 m/h, the granules readily settle, forming a sludge blanket at the reactor bottom. Settling of the biomass granules allows decoupling of HRT and SRT. An SRT as long as 200 days can be achieved with HRT as low as 6 hours (Hulshoff Pol et al., 2004). The volumetric organic loading rate (VOLR) can be extremely high: up to 50 kg COD/m^3/day (Khanal, 2008). Khanal (2008) provides more detail about UASB working principles.

4.3.1.2.4 Anaerobic sequential batch reactor

In an anaerobic sequencing or sequential batch reactor (ASBR), all stages of wastewater treatment (filling, reaction, sedimentation, and decanting) happen sequentially in one tank (see Figure 6). Due to sequential operation, a single reactor can serve as a reaction vessel and settling tank, achieving a long SRT regardless of HRT. Biomass is retained due to bioflocculation and biogranulation, similar to a UASB reactor. A larger reactor volume is required than in plants with a continuous process, however, and the process is susceptible to toxic substances. The plants are simpler, though, and are thus often used for industrial wastewater treatment (Deublein and Steinhauser, 2008). The ASBR is highly suited to treatment of animal manure and other biowastes with medium total solids contents (1-4%) (Khanal, 2008).

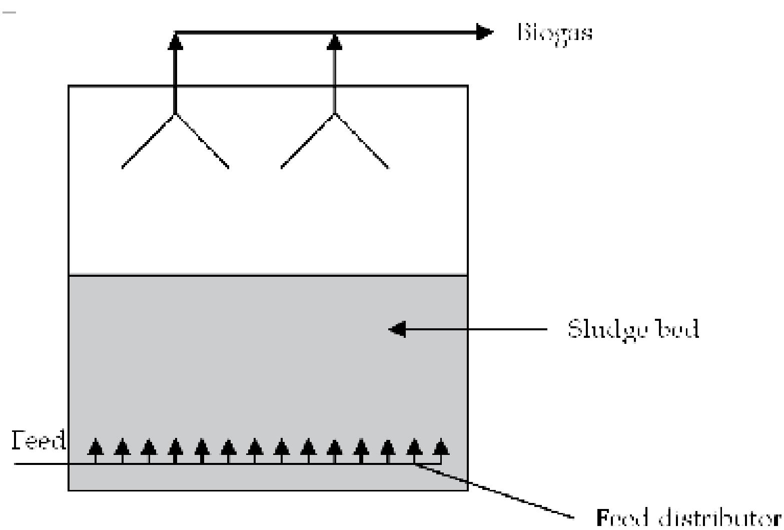

Fig. 5. Upflow anaerobic sludge blanket reactor (adapted from Khanal, 2008)

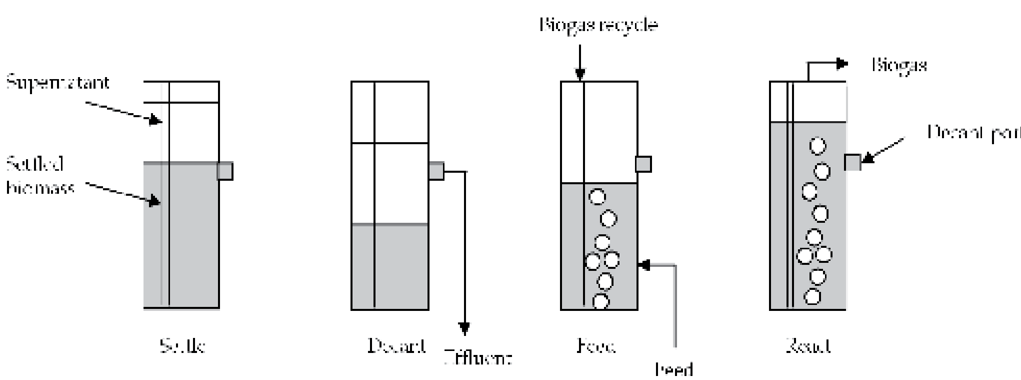

Fig. 6. Anaerobic sequencing batch reactor (adapted from Khanal, 2008)

4.3.1.2.5 Anaerobic contact process

An anaerobic contact process is a CSTR with an external tank to settle biomass, as shown in Figure 7. Settled biomass is recycled back to achieve a long SRT. The degassifier removes

CO_2 and CH_4 bubbles that may attach to biomass and thus prevent settling. The anaerobic contact process is a good choice for feeds with high suspended solids (e.g. wood fiber), which enable microbes to attach to solids and settle. Loading rates range from 0.5 to 10 kg COD/m^3/day (Khanal, 2008).

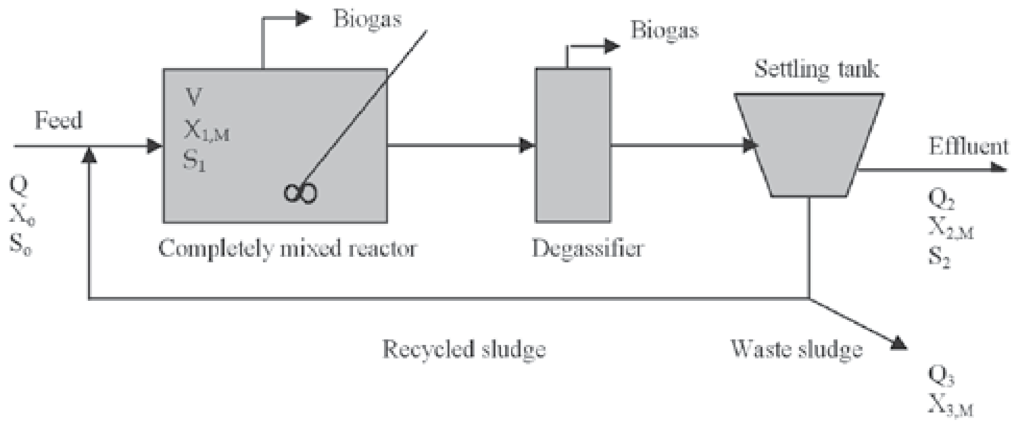

Fig. 7. Anaerobic contact process (after Khanal, 2008)

4.3.1.2.6 Anaerobic membrane bioreactor

An example of a suspended growth system, an anaerobic membrane bioreactor (AnMBR, shown in Figure 8a) uses a membrane, either within the reactor or in an external loop, to aid solids/liquid separation. Since the membrane retains biomass, extremely long SRTs are possible regardless of the HRT (Khanal, 2008).

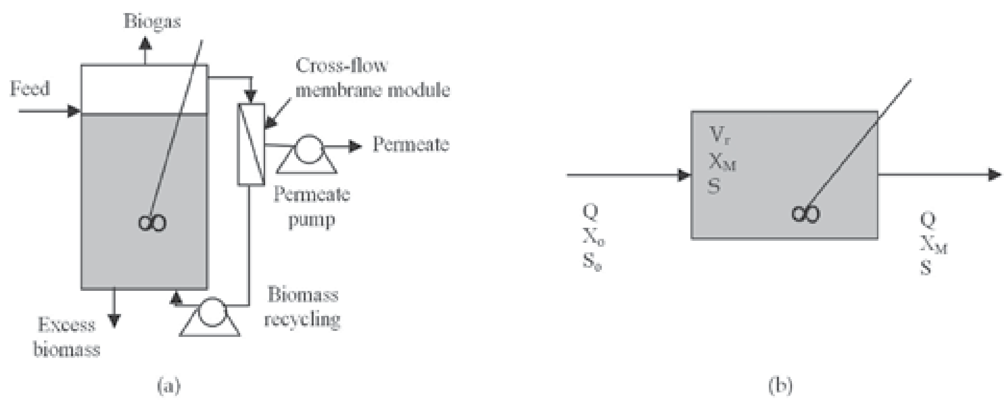

Fig. 8. Schematics of (a) anaerobic membrane bioreactor, with membrane in an external loop, and (b) completely mixed bioreactor (after Khanal, 2008)

4.3.1.2.7 High-rate CSTRs

High-rate anaerobic digesters operated as completely mixed bioreactors, or completely stirred tank reactors (CSTRs), as shown in Figure 8b, have HRT=SRT. They would thus be suitable for high-solids feed streams (TS = 1-6%), including municipal sludge, animal manure, and other biowastes (Khanal, 2008). Required detention time is typically 15 days or less (Metcalf and Eddy, 2004). Mechanical mixing, pumping, and/or gas recirculation can provide mixing.

4.3.2 Choose reactor material

The reactor must be airtight, since the methanogens are obligate anaerobes, and must also prevent liquids from leaking. Considerations in choosing a material for the reactor include:

- Local availability
- Cost
- Ability to maintain heat (thermal insulation capacity)
- Ability to absorb solar radiation (light-colored materials can be painted black to increase solar energy absorption)
- Corrosion-resistant (hydrogen sulfide and organic acids associated with anaerobic degradation can cause corrosion)

Possible materials include:

- Brick and mortar (lime mortar with waterproofing can be substituted for cement where necessary)
- Concrete, sometimes with coating
- Glazed pottery rings cemented together
- Stone
- Glass firer-reinforced plastic
- Fiberglass
- Normal steel with enamel layer or plastic coating for corrosion resistance, stainless steel (high cost may prohibit use in rural areas)
- Thick plastic (for very small tanks only)

Deublein and Steinhauser (2008) provide a more detailed discussion of reactor materials.

4.3.3 Size reactor

The digester size can be estimated using a hydraulic retention time (HRT) or using volumetric organic loading rate (VOLR). Typically, both calculations are performed, and the larger of the two sizes is used, to be conservative.

Typical HRTs for various wastes in anaerobic reactors were given in Table 2. Most of these HRTs are for the mesophilic temperature range. Typical residence times for reactors operated in the mesophilic temperature range are from 20-45 days. Typical residence times for reactors operated in the thermophilic range are around 15 days, since heating increases the rate of microbial activity. From the HRT, the reactor volume V_D can be estimated as (Deublein and Steinhauser, 2008):

$$V_D = Q_{TOTAL} * HRT * 1.25 \tag{30}$$

$$V_D = \dot{M}_{TOTAL} * HRT / \rho_{water} * 1.25 \tag{31}$$

where

Q_{TOTAL} = total waste stream (waste plus water) volumetric flow rate (m^3/day)

1.25 is a factor to account for air and fixtures

M_{TOTAL} = total waste stream (waste plus water) mass flow rate (mass/time)

The volumetric organic loading rate (VOLR) is the mass of dry organic feed/volume of digester/time, or

$$\mathrm{VOLR} = \left(\sum_{i=1}^{n} M_{\text{waste i}} * f_{\text{oTS waste i}} * f_{\text{TS waste i}} \right) / V_D \tag{32}$$

The digester volume can thus be estimated from:

$$V_D = \left(\sum_{i=1}^{n} M_{\text{waste i}} * f_{\text{oTS waste i}} * f_{\text{TS waste i}} \right) / \mathrm{VOLR} \tag{33}$$

This can also be written as:

$$V_D = C_i * Q_{TOTAL} / \mathrm{VOLR} \tag{34}$$

where C_i is the influent waste stream biodegradable COD concentration (mg/L).
The average VOLR for small plants is 1.5 kg oDM/(m^3*d) and for large plants is 5 kg oDM/(m^3*d).
Once the digester volume is found, the dimensions of the digester can then be determined according to the following rule of thumb, assuming a cylindrical digester: $H_D = 0.5 * D_D$, where H_D = height of the digester and D_D = diameter of the digester.

Example 4

Continuing with the information from Examples 1-3, size the reactor.

Solution

First, the digester will be sized based on HRT. From Eq. 31, $V_D = M_{TOTAL} * HRT / \rho_{water} * 1.25$. From Table 2, HRTs for sewage sludge, cow manure, and poultry manure are 35-45, 28-38, and 17-22 days, respectively. (The HRT for rice straw was not given.) Very little of our waste mass is poultry manure. We will choose a 50 day HRT, slightly above the range given for sewage sludge, to be conservative, since a significant portion of the mass of our waste is cow manure.
From Example 3, $\dot{M}_{TOTAL}$ = 3375 kg/day. V_D can then be calculated according to:

$$V_D = 3375 \text{ kg/day} * 50 \text{ days}/(1 \text{ kg/L}) * 1.25 * 1 \text{ m}^3/(1000 \text{ L}) = 211 \text{ m}^3$$

Now, the reactor will be sized based on VOLR. The average VOLR value for small systems, 1.5 kg oDM/(m^3*d), will be used. From Eq. 33,

$$V_D = \left(\sum_{i=1}^{n} M_{\text{waste i}} * f_{\text{oTS waste i}} * f_{\text{TS waste i}} \right) / \mathrm{VOLR}$$

For this example,

$$V_D = [\dot{M}_{\text{septage}} * f_{\text{oTS septage}} * f_{\text{TS septage}} + \dot{M}_{\text{cow manure}} * f_{\text{oTS cow manure}} * f_{\text{TS cow manure}} + \dot{M}_{\text{poultry manure}} * f_{\text{oTS poultry manure}} * f_{\text{TS poultry manure}} + \dot{M}_{\text{rice straw}} * f_{\text{oTS rice straw}} * f_{\text{TS rice straw}}]/\mathrm{VOLR}$$

$$V_D = [225 * 0.05 * 0.65 + 1187 * 0.135 + 9 * 0.45 * 0.75 + 432 * 0.375 * 0.825] \text{ kg/day} / 1.5 \text{ kgoDM/(m}^3\text{*d)} = 203 \text{ m}^3$$

To be conservative, the V_D value of 211 m^3 based on HRT will be used. Assuming that the digester is cylindrical, $V_D = H_D * \pi * D_D^2/4$. Assume $H_D = 0.5 * D_D$. Then,

$$V_D = 0.5 * D_D * \pi * D_D^2/4 = 0.125 * \pi * D_D^3$$

$$D_D = [V_D / (0.125 * \pi)]^{1/3} = [211\ m^3 / (0.125 * \pi)]^{1/3} = 8.13\ m$$

$$H_D = 4.06\ m.$$

4.3.4 Choose mixing method

In large reactors, mixing is useful in exposing new surfaces to bacterial activity and thus maintaining methane production rates. Incorporating an agitator can considerably reduce the size of the reactor. A rule of thumb is that if the volume exceeds 100 m^3, mixer should be used (OLGPB, 1978). Mixing methods include:

1. Daily feeding of the digester (semicontinuous operation),
2. Installing a mixing device operated manually or mechanically,
3. Creating a flushing action of the slurry through a flush nozzle,
4. Creating mixing action by flushing the slurry tangentially to the digester content,
5. Installing wooden conical means that cut into the straw in the scum layer as the surface of the liquid moves up and down during filling and emptying.

Adequate mixing may be difficult to achieve in an undivided large digester (intended to serve an entire community, for example). Compartments may be particularly useful for large digesters producing >500 ft^3 of gas/day.

4.3.5 Determine heating requirements

Heating speeds the rate of methane production; thus, the detention time can be reduced and the digester size can be smaller than for an unheated unit. However, heating takes energy. The operational cost of providing this energy must be weighed against the reduced capital cost of a smaller digester. For small digesters (producing <500 ft^3 of gas per day), heating using fuel may not be desirable due to maintenance requirements. Solar heating or use of waste heat from an engine-generator may be considered (NAS, 1977). Higher temperatures lower the amount of CO_2 dissolved in the liquid phase, according to Henry's law, and thus increases the percent in the gas phase; this lowers the energy content of the biogas per volume.

The heat requirements for the digester include the amount needed (Metcalf and Eddy, 2004):

1. To raise the incoming slurry to desired digestion temperatures (q_{raise}, or q_R),
2. To compensate for heat losses through the reactor floor, walls, and roof (q_{losses}, or q_L), and
3. To make up losses that might occur in piping between the heating source and tank (q_{piping}, or q_P).

The total heat required is thus:

$$q_{TOT} = q_R + q_L + q_P \tag{35}$$

Heat required to raise the slurry temperature can be calculated from:

$$q_R = \dot{M}_{TOTAL}\, c\Delta T \tag{36}$$

where q_R = heat requirement, Btu/h (W)

M_{TOTAL} = mass flow rate of slurry to be heated

c = slurry heat capacity, which can be assumed to be the same as that of water (1 Btu/lb/°F) (Metcalf and Eddy, 2004)
ΔT = difference between the incoming slurry temperature and the desired reactor temperature.
The maximum heat requirement should be calculated for the coldest month of the year. Heat losses through the reactor floor, walls, and roof can be calculated according to:

$$q_L = \sum_{j=1}^{n} U_j A_j \Delta T_j \tag{37}$$

where q_L = heat loss, Btu/h (W)
U_j = overall coefficient of heat transfer for surface j, Btu/ft²/h/°F (W/m²/°C)
A_j = cross-sectional area of surface j through which heat loss is occurring, ft² (m²)
ΔT_j = temperature drop across surface j, °F (°C)
Overall heat transfer coefficients for typical digester materials are given in Table 5. Expanded plastic slabs of polyurethane can provide insulation for the tank bottom. For the upper portion of the tank, expanded polystyrene slabs, mineral wool mats, plastic foam, leaves, sawdust, or straw can be used to insulate the tank and minimize heating requirements.

Example 5

Continuing with the information from Examples 1-4, estimate the heat that would be required to heat the digester from 40°F to 90°F. Assume that the digester is above ground, and made from 12″ thick concrete walls with insulation. The concrete floor is 12″ thick, in contact with dry earth. The fixed concrete cover is 4″ thick and insulated. Assume no losses between the heating source and tank.

Solution

From Eq. 35, $q_{TOT} = q_R + q_L + q_P$. q_P is assumed to be 0. q_R can be calculated from Eq. 36 according to:

$$q_R = \dot{M}_{TOTAL}\, c\Delta T$$

$$q_R = 3375\ \text{kg/day} * (1\ \text{Btu/lb/°F}) * (90°F – 40°F) * (2.2\ \text{lb/kg})$$

$$q_R = 3.71 * 10^5\ \text{Btu/day}$$

q_L can be calculated from Eq. 37 according to:

$$q_L = \sum_{j=1}^{n} U_j A_j \Delta T_j$$

$$q_L = U_{walls}A_{walls}\Delta T_{walls} + U_{floor}A_{floor}\Delta T_{floor} + U_{cover}A_{cover}\Delta T_{cover}$$

From Table 5, taking the mean value in each range, U_{walls} = 0.125, U_{floor} = 0.06, and U_{cover} = 0.245 Btu/ft²/h/°F.
From Example 4, D_D = 8.13 m and H_D = 4.06 m. The areas of the walls, floor, and cover are thus:

$$A_{walls} = \pi * D_D * H_D = \pi * 8.13 \text{ m} * 4.06 \text{ m} = 103.8 \text{ m}^2 = 1117 \text{ ft}^2$$

$$A_{floor} = A_{cover} = \pi * D_D{}^2/4 = \pi * (8.13 \text{ m})^2/4 = 51.9 \text{ m}^2 = 558.7 \text{ ft}^2$$

$$q_L = 0.125 \text{ Btu/ft}^2\text{/h/°F} * 1117 \text{ ft}^2 (90°F – 40°F) + 0.06 \text{ Btu/ft}^2\text{/h/°F} * 558.7 \text{ ft}^2 (90°F – 40°F) + 0.245 \text{ Btu/ft}^2\text{/h/°F} * 558.7 \text{ ft}^2 (90°F – 40°F) = 15{,}505 \text{ Btu/h} = 646 \text{ Btu/day}$$

$$q_{TOT} = = 3.71 * 10^5 \text{ Btu/day} + 646 \text{ Btu/day} = 3.72 * 10^5 \text{ Btu/day}$$

Item	**Btu/ft²/°F/h**
Plain concrete walls (above ground)	
12″ thick, not insulated	0.83-0.90
12″ thick with air space plus brick facing	0.32-0.42
12″ thick wall with insulation	0.11-0.14
Plain concrete walls (below ground)	
Surrounded by dry earth	0.10-0.12
Surrounded by moist earth	0.19-0.25
Plain concrete floors	
12″ thick, in contact with dry earth	0.05-0.07
12″ thick, in contact with moist earth	0.10-0.12
Floating covers	
With 1.5″ wood deck, built-up roofing, and no insulation	0.32-0.35
With 1″ insulating board installed under roofing	0.16-0.18
Fixed concrete covers	
4″ thick and covered with built-up roofing, not insulated	0.70-0.88
4″ thick and covered, but insulated with 1″ insulating board	0.21-0.28
9″ thick, not insulated	0.53-0.63
Fixed steel cover (1/4 ″ thick)	0.70-0.95

Table 5. Overall heat transfer coefficients for typical digester materials (Metcalf and Eddy, 2004)

4.4 Design the gas storage system

Gas can be stored in a digester with floating cover, or gas from a digester with a fixed cover can be piped into an auxiliary gas holder with a floating cover. Materials for the cover can include mild steel, EDPM rubber, or concrete. The volume of the gas holder depends on the daily gas production and usage. It may be as low as 50% of the total volume of daily gas production, if gas usage is frequent.

Example 6

Continuing with the information from Examples 1-5, determine the volume and dimensions for a cylindrical gas holder to be mounted on top of the digester.

Solution

From Example 2, 107.3 m^3 biogas/day would be produced. Since the gas will be used on a regular basis and withdrawn at a relatively constant rate, the gas holder need have only half the volume of the required daily production. Thus, the gas holder needs to have a capacity of 53.7 m^3. For a cylindrical gas holder to fit onto the top of the digester whose dimensions were determined in Example 5, a suitable diameter would be 7.98m, or 15 cm less than the diameter of the digester. The height of the gas holder would then be:

$$H_H = Vol_H/(\pi * D_H^2/4) = 53.7 \text{ m}^3/(\pi * (7.98\text{m})^2/4) = 1.07 \text{ m}$$

4.5 Determine system location

The system location should be:

- At least 50 ft from the nearest drinking water well, to avoid potential contamination (NAS, 1977).
- At least 10 m from any homes, to avoid any methane safety issues (FAO, 1984).
- Out of the sun in hot climates, in the sun in cooler climates (FAO, 1984).
- On firm soil, preferably with a low underground water level (OLGPB, 1978). Away from trees, so roots will not cause cracks (OLGPB, 1978).
- Close enough to place of use to reduce length of connection tubing, and corresponding loss in gas pressure associated with friction with the walls of the tube (OLGPB, 1978).

5. Benefits and limitations of anaerobic processes

Anaerobic treatment processes solve 2 problems at once: waste and energy. Benefits of anaerobic processes compared to aerobic processes are discussed in detail in Sattler (2011), and are summarized briefly here. Benefits of anaerobic systems compared to aerobic systems include:

- Production of usable energy,
- Reduced sludge (biomass) generation/stabilization of sludge,
- Higher volumetric organic loading rate/reduced space requirements,
- Reductions in air pollutants and greenhouse gases,
- Lower capital and operating costs,
- Lower nutrient requirements and potential for selective recovery of heavy metals.

Remaining limitations of anaerobic processes include:

- Requirements for post-treatment,

- Methane loss in the effluent,
- Sensitivity to low temperatures, and
- Attention required during start-up.

6. Summary

Steps in anaerobic degradation of organic material by bacteria include polymer breakdown (hydrolysis), acid production (acidogenesis), acetic acid production (acetogenesis), and methane production (methanogenesis). Various factors associated with the waste impact both the quantity and rate of methane production, including waste composition/degradable organic content, particle size, and organic loading rate (kg/(m^3*d)). Environmental factors impacting the rate of methane generation include temperature, pH, moisture content, nutrient content, and concentration of toxic substances.

Steps in design of a gas production system include:

1. Determine biogas production requirements,
2. Select waste materials and determine feed rates; size waste storage; determine rate of water addition and size the preparation tank,
3. Design the digester/reactor,
4. Design the gas storage system,
5. Determine system location.

Benefits of anaerobic systems compared to aerobic systems include production of usable energy, reduced sludge (biomass) generation/stabilization of sludge, higher volumetric organic loading rate/reduced space requirements, reductions in air pollutants and greenhouse gases, and lower capital and operating costs.

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13

Management of Phosphorus Resources – Historical Perspective, Principal Problems and Sustainable Solutions

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1. Introduction

Phosphorus, a common element ranking 11th in order of abundance in the Earth's crust, is essential for life and part of all biological systems. It is a major component of the vertebrate skeleton, an important element of gene pools, a component of cell membranes and an essential element for energy transfer. Consequently, phosphorus is a major plant nutrient. Today, about 80% of the phosphate rock mined is converted into mineral fertilisers in order to sustain world food production (Heffer et al., 2006).

Mineable phosphate rock is a non-renewable resource. However, a main proportion of the phosphorus fertiliser present in food ends up in urban wastes such as sewage sludge and slaughterhouse waste. Urbanisation and population growth impose specific challenges for phosphorus recirculation. At the global scale, more than 50% of the human population (3.3 billion) lives in urban locations and urbanisation is increasing (United Nations, 2010). In future, it will be of the utmost importance to recycle and reuse the phosphorus present in waste in order to minimise losses and conserve existing resources. In fact, phosphorus recirculation in society already has a high priority in national environmental programmes. However, the re-use of municipal wastes in agriculture is currently impeded by problems such as: (i) the presence of unwanted metals, organic pollutants and pathogens, limiting recycling of municipal wastes (sewage sludge, slaughterhouse wastes and household compost) to agricultural land; (ii) logistical difficulties in re-distributing surplus municipal wastes such as sewage sludge from urban areas back to arable land; and (iii) a low fertiliser value.

Two contrasting situations for nutrient recirculation can be identified: huge urban centres with large-scale treatment of wastes requiring long-distance transportation of nutrients back to arable land; and rural settlements with small-scale, on-site waste collection/treatment and sufficient arable land nearby for soil application. This chapter mainly focuses on phosphorus recirculation from densely populated areas.

The chapter begins by reviewing earlier waste treatment in society, the production of phosphorus fertilisers and foreseeable problems. The conditions necessary to achieve recirculation of municipal wastes are then described and possible technical solutions that fulfil these conditions are presented.

2. Historical perspective

2.1 Lesson from waste treatment in the past – limited recycling of human waste to soil

It could be assumed that in the pre-industrialised age, complete nutrient cycling was achieved through spreading human, animal and plant residues onto agricultural land. However, recycling of human waste to land was limited in early societies.

Urban settlements require wastes to be handled in a planned manner, which was the case even in early history. The Indus and Harappa cultures, which settled along the Indus river (today Pakistan) around 3000 BC, seem to have used water to remove toilet wastes and conducted the wastewater into recipient water bodies (Glover & Ray, 1994). Houses with water toilets, bathrooms and outflows connected to brick-covered channels in streets have been found. The Minoan culture on Crete in 2000-1500 BC also used water toilets, which were connected to sewage channels. Stone-walled pits of about 5 m in diameter found at Knossos were probably used for solid waste treatment through deep litter decomposition (Joyner, 1995). In the Greek and Roman cultures, town planning, water supply, sewage discharge and waste treatment were highly developed services. Sewage water from Athens in 500 BC was applied to open fields in rural surroundings (White-Hunt, 1980a), while drains and sewers of Nippur and Rome, among the great structures of antiquity, were used to carry away storm runoff, toilet wastes and street washing water. From the Cloaca Maxima in Rome (the main sewage tunnel), effluents were transported through channels to far outside settlements for both discharge and infiltration (Dersin, 1997). Solid, settled waste material, 'black gold', was recovered from sewage systems and ponds and recycled to arable land.

Type of material	Mean water content (%)	Cadmium content	
		(mg kg^{-1} dry weight)	(mg kg^{-1} phosphorus)
Household compost	65	1.3	220
Human urine	99	0.02	1
Sewage sludge	75	1.05	35
Ash (sewage sludge)	<3	1.58	35
Harvested field crops:			
Wheat	16	0.04	12
Barley	16	0.02	6

Table 1. Some characteristics of municipal wastes compared with harvested field crops. Data complied from: Kirchmann and Pettersson (1995), Kirchmann and Widen (1994), Cohen (2009), Eriksson (2009) and Svanberg (1971).

In contrast, historical documents from China, Korea and Japan show comprehensive and effective handling and treatment systems for organic human wastes not using water for sewage transport (King, 1911). Instead, careful collection and extensive transport of latrine, organic wastes, ash, etc. from some large cities back to agricultural land by human- or animal-drawn carts and manure boats is described. Extensive collection was followed by careful storage and treatment. Application of urine, pulverised human excreta, ash and composts, often mixed with sod or mud, canal sediments, etc., to arable land ensured a high degree of nutrient recirculation and maintenance of soil fertility. It should be noted that the volume of waste transported back to agricultural land was larger than the volume of food consumed owing to the higher water content in different wastes compared with major food types (Table 1).

The Middle Ages were characterised by a decline in hygiene and sanitation standards in cities and towns in Europe. Failure to remove the wastes from houses and streets, overloaded ditches and sewer channels in and around cities caused heavy pollution of watercourses in many places, for example London (White-Hunt, 1980a; 1980b). Wastes could be stored in tanks in the bottom of buildings or discharged into narrow lanes between houses from toilets placed above (the narrow alleys present in romantic medieval town structures) and the removal intervals could be long. The absence of an effective sewage and waste handling system was a major hindrance in combating diseases in European cities of that era. Furthermore, even animal wastes were not necessarily applied to arable land, as a significant but unknown amount was leached to produce nitrate for use in gunpowder.

In summary, early cultures discharged or infiltrated wastes and wastewater from water-based sewage systems outside urbanised areas and thus the nutrients they contained were not recycled to arable land. Estimates show that at least 50% of total nutrients present in toilet wastes were lost, representing the proportion present in urine (see compilation by Kirchmann et al., 2005). The key lesson from this historical review is that recirculation of human wastes to soil was limited. As a result, the stock of nutrients in agricultural soils was gradually depleted and soil fertility decreased.

2.2 History of phosphorus fertiliser production - from bones to non-renewable resources

To slow down nutrient depletion in arable soils, especially of phosphorus, animal bones consisting of calcium phosphate were applied during earlier times. Several 17th Century publications in Europe mention the beneficial effect of bones. In 1769, the Swedish scientist J.G. Gahn discovered that calcium phosphate is the main component of bones, but the role of phosphorus as a major plant nutrient was still not known. Field trials demonstrated that bones should be crushed and applied in the form of powder, but the positive effect obtained was ascribed to organic components in bones. Attempts were made to improve the efficiency of bones by (i) composting them together with animal and plant wastes, (ii) boiling them in water; or (iii) treating them with steam under pressure. The widespread use of bones led to the idea of chemical treatment of bone material. H.W. Köhler of Bohemia was probably the first to suggest such a treatment and filed a patent for using acids (especially sulphuric acid) to process and produce commercial phosphate fertilisers (1831). In 1840, Justus von Liebig published work showing that plants take up nutrients in the form of inorganic components and carbon from air. Until then, academics from Aristotle (384-322 BC) to Thaer (1752-1828 AD) had considered organic matter in soil (humus) to be the source of plant dry matter. Liebig's findings contributed to the acceptance and development of phosphorus fertilisers. Together with the English businessman J. Muspratt, Liebig developed and patented a method to produce a combined phosphorus and potassium fertiliser. However, the fertiliser they produced was a complete failure, since the phosphate and potassium present were insoluble in water and therefore unavailable to plants. When the initial failure of this fertiliser and the insignificant effect of bone powder as a fertiliser became understood, the importance of the water solubility of plant nutrients was fully recognised and the concept of producing water-soluble fertilisers was introduced (Finck, 1982).

Lack of bone material as a phosphorus source led to the import of guano from Peru around 1840. The discovery of low-grade mineral phosphates (apatite) in France and England eased the situation. The first 'artificial' fertiliser, superphosphate, was produced in England in 1843 from apatite and sulphuric acid (see reaction below).

$$2\ Ca_5F(PO_4)_3 + 7\ H_2SO_4 \rightarrow 3\ Ca(H_2PO_4)_2 + 7\ CaSO_4 + 2\ HF\ \text{(superphosphate)} \quad (1)$$

Superphosphate is a mixture of mono-calcium phosphate and gypsum, with a mean phosphorus content of 7-9.5%. In 1855, superphosphate was also produced in Germany and in 1860 the first plant was built in Sweden (Klippan). Due to increased use of artificial phosphorus fertilisers, cereal yields almost doubled between 1840 and 1880 from about 0.8 to 1.4 tons per hectare. Use of phosphoric instead of sulphuric acid for apatite dissolution resulted in triple superphosphates being commercialised in 1890. These also consisted of mono-calcium phosphate, but without gypsum (see reaction below) and had a phosphorus content of 17-23%.

$$Ca_5F(PO_4)_3 + 7\ H_3PO_4 \rightarrow 5\ Ca(H_2PO_4)_2 + 2\ HF\ \text{(triple superphosphate)} \quad (2)$$

The development of the phosphate industry was secured by the discovery of large sedimentary phosphate deposits in South Carolina (USA). Mining began in 1867 and by 1889 the USA was supplying 90% of the apatite used worldwide for phosphate fertiliser production.

In 1917, a new phosphorus fertiliser was developed in the USA by reacting phosphoric acid with ammonia gas to form mono- and di-ammonium phosphate (see reactions below).

$$H_3PO_4 + NH_3 \rightarrow NH_4H_2PO_4\ \text{(mono-ammonium phosphate)} \quad (3)$$

$$H_3PO_4 + 2\ NH_3 \rightarrow (NH_4)_2HPO_4\ \text{(di-ammonium phosphate)} \quad (4)$$

Mono-ammonium phosphate is the inorganic phosphate salt with the highest phosphorus concentration (up to 26%). The production of ammonia on a major industrial scale from nitrogen gas in air and hydrogen gas in coal through the Haber-Bosch process boosted the production of ammonium phosphate fertilisers. In 1926, IG Farbenindustrie in Germany announced the development of a series of multi-nutrient fertilisers based on crystalline ammonium phosphate. In the late 1920s, the nitro-phosphate process was developed in Norway. In this process, phosphate rock is treated with nitric acid and calcium nitrate and ammonium phosphate are produced (see reaction below).

$$Ca_5F(PO_4)_3 + 10\ HNO_3 \rightarrow 5\ Ca(NO_3)_2 \downarrow + 3\ H_3PO_4 + HF\ \text{(calcium nitrate)} \quad (5)$$

$$H_3PO_4 + NH_3 \rightarrow NH_4H_2PO_4\ \text{(mono-ammonium phosphate)} \quad (6)$$

Reviews carried out by Finck (1982), Kongshaug (1985), and Mårald (1998) show that phosphate rock, a limited mineable resource, has been the main source for phosphorus fertiliser production since 1867.

3. Relevant issues

3.1 Rock phosphate and the cadmium and uranium problem

About 80% of the phosphate rock currently mined is used to manufacture mineral fertilisers. Use for detergents, animal feeds and other applications (metal treatment, beverages, etc.) accounts for approx 12, 5 and 3 %, respectively (Heffer et al., 2006). The global production of rock phosphate amounted to 174 million tons in 2008 (IFA, 2010a). How long existing phosphorus reserves will last is difficult to forecast. Some estimates vary between 50 to 100 years, assuming peak phosphorus (Cordell et al., 2009; Cordell, 2010) and excluding reserve bases currently not economical to mine (Steen, 1998; Driver et al., 1999; Stewart et al., 2005; Buckingham & Jasinski, 2006). Other estimates are around 350 years, based on current production capacity and excluding increased demand for phosphorus (IFDC, 2010; USGS, 2011).

Depending on its origin, phosphate rock can have widely differing mineralogical, textural and chemical characteristics. Igneous deposits typically contain fluorapatites and hydroxyapatites, while sedimentary deposits typically consist of carbonate-fluorapatites collectively called francolite. Sedimentary deposits account for about 80% of global production of phosphate rock (Stewart et al., 2005). As high-quality deposits have already been exploited, the quality of the remaining sedimentary phosphorus reserves is declining and the cost of extraction and processing is increasing, mainly due to lower phosphorus content in the ore (Driver et al., 1999). Associated heavy metals such cadmium and uranium substituting for calcium in the apatite molecule are often present at high levels in phosphate rock, especially that of sedimentary origin. Rock phosphate may contain up to 640 mg cadmium per kilogram phosphorus (Alloway & Steinnes, 1999) and up to 1.3 g uranium per kilogram phosphorus (Guzman et al., 1995). Only a minor proportion of phosphorus reserves have low cadmium content (Fig. 1). Most (85-90%) of the cadmium and uranium in rock phosphate ends up in fertilisers (Becker, 1989).

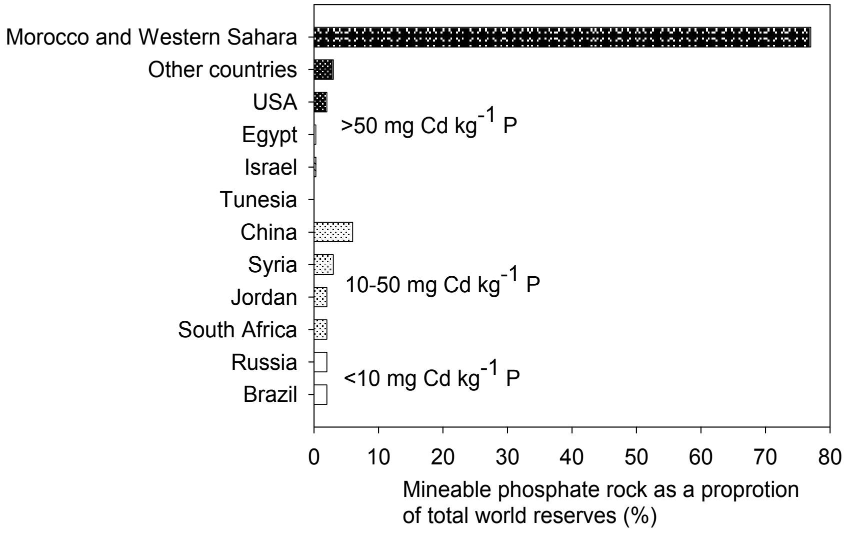

Fig. 1. Mineable phosphate rock and cadmium content. Estimates of mineable amounts taken from US Geological Survey (USGS, 2011) and cadmium contents from McLaughlin & Singh (1996).

Recent studies show that uranium originating from fertilisers accumulates in soils, leading to uranium losses to natural waters (Schnug & Haneklaus, 2008). The biochemical toxicity of uranium has been shown to be six orders of magnitude higher than the radiological toxicity (Schnug & Haneklaus, 2008). Uranium in soil enters the food chain mainly through consumption in drinking water.

A new standard for low cadmium content in phosphorus fertilisers is likely to become an issue, since the European Food Safety Authority recently reduced the recommended tolerable weekly intake of cadmium from 7 to 2.5 micrograms per kilogram body weight, based on new data regarding the toxicity of cadmium to humans (EFSA, 2009). Several countries already restrict cadmium levels in phosphate fertilisers and there is a need for exclusion of cadmium and uranium from phosphorus fertilisers for safe food production.

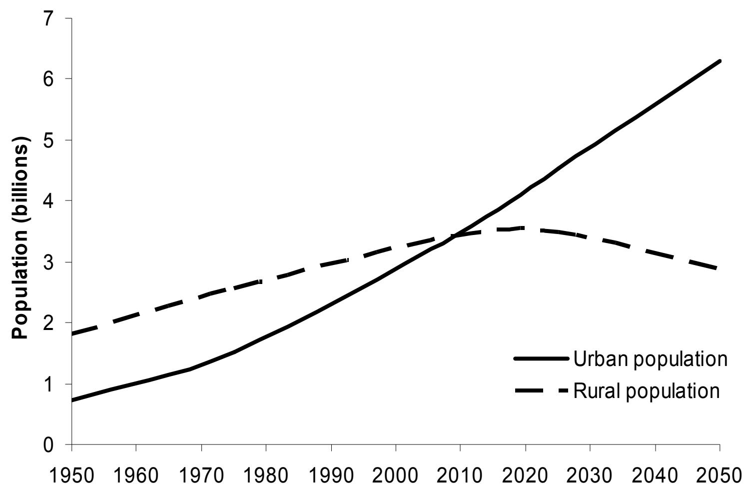

Fig. 2. Urban and rural population of the world, 1950-2050. Data from United Nations (2010).

3.2 Population growth and urbanisation

The global population is rapidly increasing. Between 1950 and 2009 the population increased from 2.5 billion to 6.8 billion and it is expected to reach 9.1 billion by 2050 (United Nations, 2009). In addition, the 20th Century witnessed rapid urbanisation in the world. The proportion of urban population increased from 13% in 1900 to 29% in 1950 and reached 50% in 2009 (United Nations, 2010). Population growth is expected to occur mainly in urban areas (Fig. 2), the population of which is projected to increase from 3.4 billion in 2009 to 6.3 billion in 2050. Cities in less developed regions will become centres of population growth. Table 2 shows the expected population growth for some large cities between 2010 and 2025. Statistics show that 1.4 billion people live in 600 cities, excluding suburban areas with a population larger than 0.75 million inhabitants (mean population of 2.3 million per city) (GeoHive, 2010).

Urbanisation and population growth impose specific challenges for phosphorus use: (i) long-distance recycling of nutrients from large cities back to arable land to avoid contamination of surrounding areas and to ensure long-term supply of P fertiliser; (ii) increase in crop production by at least 50% by 2030 to ensure sufficient food supply

(Bruinsma, 2003); and (iii) increased bio-fuel production to replace fossil fuels. As a result, agriculture will be intensified and the demand for phosphorus fertilisers will increase. The increase in phosphorus demand is estimated to be 2.8% per year (FAO, 2008).

3.3 Phosphorus in waste flows in society

All forms of agriculture remove plant nutrients from fields via the harvest of crops. The nutrients removed from fields flow through one or more of three cycles: the fodder cycle, the food cycle, and the industrial cycle (Fig. 3). The fodder cycle is the flow through housed animals, on or off the farm, which results in manures, slurries, urine, feed-lot wastes and deep-litter wastes. The food cycle concerns human consumption of food of plant or animal origin, and the resulting wastes. The industrial cycle concerns processing of animal and vegetable products into food and the resulting industrial residues.

The fodder cycle in the past was more or less closed, since manures were normally recycled to arable land except for the portion used for nitrate production for gunpowder. Today, however, transfer of fodder to a livestock farm can result in nutrient accumulation that far exceeds the absorption capacity of nearby farmland. Manure surpluses occur in many regions of Europe, Asia and the USA. For example, Haygarth et al. (1998) calculated that a typical intensive dairy farm of 57 ha in the UK with 129 lactating cows results in a net annual accumulation of approximately 26 kg phosphorus per hectare. The Netherlands has an estimated national surplus of about 8000 tons of phosphorus per year (Greaves et al., 1999). Incineration of manure to minimise the logistical difficulties of handling surplus manure and to recover energy is now practised in regions with a high animal density (Kuligowski & Poulsen, 2010).

Cities ranked according to expected size in 2025	**Population (million)**		
	2000	2010	2025
Tokyo, Japan	34.4	36.7	37.1
Delhi, India	15.7	22.2	28.6
Mumbai (Bombay), India	16.1	20.0	25.8
São Paulo, Brazil	17.1	20.3	21.7
Dhaka, Bangladesh	10.3	14.6	20.9
Ciudad de México, Mexico	18.0	19.5	20.7
New York-Newark, USA	17.8	19.4	20.6
Kolkata (Calcutta), India	13.1	15.6	20.1
Shanghai, China	13.2	16.6	20.0
Karachi, Pakistan	10.0	13.1	18.7
Lagos, Nigeria	7.2	10.6	15.8
Beijing, China	9.8	12.4	15.0
Manila, Philippines	10.0	11.6	14.9
Buenos Aires, Argentina	11.8	13.1	13.7
Los Angeles, USA	11.8	12.8	13.7
Al-Qahirah (Cairo), Egypt	10.2	11.0	13.5
Rio de Janeiro, Brazil	10.8	11.9	12.7
Istanbul, Turkey	8.7	10.5	12.1

Table 2. Population growth for some large cities 2000-2010 and prediction for 2025. Data from GeoHive (2010).

The food cycle suffers from severe problems regarding return of nutrients from cities back to arable land. Urban growth has resulted in centres of consumption, and hence accumulation of human wastes, that are far away from areas of agricultural production. Nutrients removed from the fields enter cities in the form of food of plant or animal origin, resulting in the production of municipal wastes such as toilet waste in the form of sewage sludge, and organic household waste in the form of compost or biogas residues. These organic wastes typically have high water and low nutrient contents. For example, dewatered sewage sludge contains 70-80% water and the phosphorus content is only about 3% of dry matter. Waste accumulation around cities leads to logistical difficulties in re-distributing human waste to arable land. The volume of urban waste is three- to five-fold larger than the volume of most harvested crops (Kirchmann et al., 2005). Lack of available arable land for organic waste application within reasonable distance from cities requires strategies for reducing the volume of urban wastes. In many cities sewage sludge is incinerated, whereby the volume of dewatered sewage sludge can be reduced by approx. 90%.

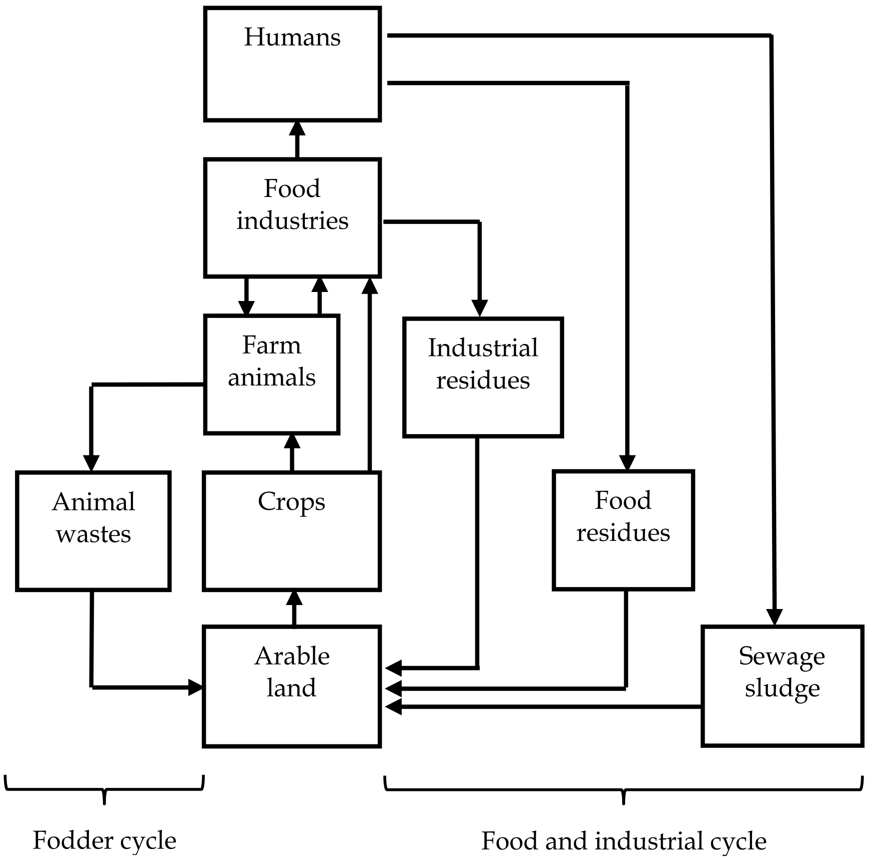

Fig. 3. Plant nutrient cycling in society can be divided into fodder, food and industrial cycles.

Fertilisation with sewage sludge has dramatically declined in many countries due to the logistical difficulties in handling surplus sewage sludge and the unwillingness of farmers to apply sewage sludge on arable land because of the presence of heavy metals and organic contaminants. In some European countries, use of sewage sludge on arable land is completely prohibited (e.g. Switzerland). In addition, landfilling of organic material has been prohibited in EU countries since 2005. As a consequence, sewage sludge is increasingly being incinerated. As the ash is rich in heavy metals and its value as a phosphorus fertiliser is low, it is mainly landfilled.

The industrial cycle also suffers from problems regarding phosphorus recycling. The outbreak of Bovine Spongiform Encephalopathy (BSE) disease led to a ban on the reuse of meat and bone meal (MBM) in animal feed. Furthermore, many EU countries prohibit soil application of MBM. Again, incineration or gasification to destroy potential BSE infective material remains an option, normally followed by landfilling.

The amount of phosphorus in sewage sludge and MBM within the EU is estimated to be 250,000 and 133,000 tons per year, respectively (Werner, 2003). Approximately 25% of the sewage sludge and 44% of the MBM produced in the EU were incinerated in 2003 (Werner, 2003). A calculation for Sweden shows that phosphorus in sewage sludge and slaughterhouse wastes comprises 63% of the phosphorus applied through inorganic fertilisers (Table 3). In the USA, there are approximately 170 sewage sludge combustion plants incinerating approx. 20% of the sewage sludge and producing between 0.45 and 0.9 million tons of sludge ash per year (US Department of Transportation, 2005). In Japan and the Netherlands, all sewage sludge is incinerated. Donatello et al. (2010) estimated that about 1.2 million tons of sewage sludge ash are produced every year in North America and the EU and a further 0.5 million tons in Japan. Since ash is landfilled as a rule, an undesirable flow of phosphorus has arisen from fields, through cities, to landfill.

4. Possibilities for phosphorus recycling from toilet wastes

There are four main options available for recycling phosphorus from toilet wastes: a) spreading sewage sludge on arable land; b) separating human urine from faeces in special toilets and using the urine as a fertiliser; c) recovering phosphorus from sewage water in wastewater treatment plants; and d) recovering phosphorus from the ash of incinerated sewage sludge. Each option has advantages and disadvantages, as briefly discussed below, and the best choice depends on the conditions present.

Type of material	kg P ha^{-1} yr^{-1}
Crops	15
Animal wastes	11
Inorganic fertilisers	6
Sewage sludge	2.6
Slaughterhouse wastes	1.2
Food residues	0.4

Table 3. Estimated amounts of relevant phosphorus (P) flows in Swedish society. Data from Kirchmann et al. (2005) and Swedish EPA (2002).

4.1 Spreading sewage sludge on arable land

The main advantage of spreading sewage sludge on arable land is that the majority of the phosphorus in toilet wastes can be returned to arable land. More than 90% of the phosphorus entering a modern sewage treatment plant is normally incorporated into sludge (Palmgren, 2005). Additional advantages are that about 20% of the nitrogen content in sewage water is recycled and organic matter is added to the soil. The main disadvantage of phosphorus recycling via sewage sludge is the logistical problem of handling the large amounts of sludge produced in cities with limited arable land available within a reasonable distance. Another disadvantage is that sewage sludge may contain high concentrations of pollutants (metals, organic compounds, pathogenic organisms, pharmaceutical residues, viruses, etc.). An additional disadvantage is that the phosphorus in sewage sludge is mainly bound to iron or aluminium, which are used for phosphorus precipitation in wastewater treatment plants. Iron/aluminium phosphates have very low solubility and the plant availability of the phosphorus in sewage sludge is usually low.

4.2 Urine separation

About 75% of the nitrogen and 50% of the phosphorus and potassium in sewage water originate from urine (Lentner et al., 1981; Kirchmann & Pettersson, 1995; Viessman & Hammer, 1993; Bitton, 1994; Droste, 1997). Urine is a suitable fertiliser but urine separation is not a suitable option for phosphorus recycling from urban areas. The main reason is that urine is a very dilute solution, with a salt content of less than 1% and a phosphorus concentration of about 0.05% (0.36-0.67 grams phosphorus per litre), (Kvarnström et al., 2006). For a large city this would require storage and transport of very large volumes. For example, separating the urine from the London urban zone (approx. 11.9 millions inhabitants) would mean storage and transportation of approx. 6.5 million cubic metres of urine per year. This would require the equivalent of 1,300 Olympic swimming pools for urine storage, while spreading would involve 165,000 40-ton tanker loads per ca 3 month. In addition, separate collection of urine causes precipitation of phosphate salts (struvite, calcium phosphate, etc.) in pipes due to increased pH level (> 9) which leads to blockages (the pH in urine increases due to enzymatic splitting of urea to ammonia and bicarbonate). A recent study showed that around 45% of the phosphorus in urine precipitates in storage tanks (Wohlsager et al., 2010). It is therefore difficult to transport urine in pipes over longer distances. Richert Stintzing et al. (2007) reported that the maximum distance for transporting urine in pipes should not exceed 10 metres in order to minimise scaling problems. Other disadvantages of urine separation are the requirement for a separate pipe system and special toilets, losses of nitrogen due to ammonia volatilisation, possible contamination with pharmaceutical residues and the continuing need for wastewater treatment of faecal water, greywater (laundry, dishwashing and bathing) and industrial wastewater. Thus, half the phosphorus in wastewater will still end up in sewage sludge. The conclusion is that urine separation is a suitable recycling strategy for rural settlements and small villages lacking sewage treatment infrastructure (e.g. in developing countries) but having agricultural land adjacent to housing.

4.3 Phosphorus recovery from wastewater

Phosphorus recovery from wastewater is mainly based on precipitation of phosphorus from side-streams within sewage treatment plants. This produces calcium phosphate without organic matter or other impurities (van Dijk & Braakensiek, 1984; Eggers et al., 1991; Seckler et al., 1996a,b,c; Angel, 1999; Giesen, 1999) or struvite (magnesium ammonium phosphate) (Ueno & Fujii, 2001; Parsons et al., 2001; Britton et al., 2009). Calcium phosphate, which is equivalent to rock phosphate, can be processed industrially (Schipper et al., 2001). Struvite cannot be processed industrially but can be used as a slow-release fertiliser (Johnston & Richards, 2003a). The main disadvantage of recovering phosphorus directly from sewage water is that only the phosphorus present in the liquid phase can be recovered, which reduces the efficiency of phosphorus recovery considerably. Anaerobic digested sludge usually contains 40-80% of the phosphorus present in wastewater and therefore only 20-60% of total phosphorus in sewage water can be recovered as inorganic salts from side-streams in treatment plants (Murakami et al., 1987; Wild et al., 1997; Sen & Randall, 1988; Gaastra et al., 1998; Strickland, 1999; Brdjanovic et al., 2000; Piekema & Giesen, 2001; Balmer et al., 2002; Hao & van Loosdrecht, 2003). Phosphorus can precipitate *in situ* during anaerobic digestion as struvite, calcium phosphate and/or iron/aluminium phosphate and is incorporated into sewage sludge and thereby withdrawn from the liquid phase. Another disadvantage of phosphorus recovery from wastewater is that the cost of the chemicals required for phosphorus precipitation as struvite or calcium phosphate currently exceeds the value of the phosphorus products recovered (Dockhorn, 2009).

4.4 Phosphorus recovery from sewage sludge ash

Sewage sludge contains more than 95% of the phosphorus entering a modern wastewater treatment plant. For example, the Käppala sewage treatment plant in Stockholm, Sweden, has a phosphorus removal efficiency of 97% (Palmgren, 2005). Incineration of sewage sludge at 800-900^{o}C does not cause significant phosphorus losses through volatilisation and the phosphorus remains in the ash. Thus, the potential for phosphorus recovery from sludge ash is high. The main disadvantage of this option is the need for investment in sludge incineration. However, a considerable amount of sewage sludge is already being incinerated and sludge incineration is expanding due to the difficulties in handling large volumes of sewage sludge. The phosphorus concentration in ash of incinerated sewage sludge usually varies between 7 and 13% by weight (Cohen, 2009; Schaum et al., 2004) and is only slightly lower than the phosphorus concentration in beneficiated phosphate rock (12-16% by weight), indicating that ash of sewage sludge is a concentrated phosphorus source.

Several processes have been suggested for recovering phosphorus from sewage sludge ash. In some approaches, phosphorus is leached from the ash using an acid, followed by precipitation as iron phosphate (Takahashi et al., 2001) or aluminium phosphate (Schaum et al., 2004). The drawback of phosphorus recovery techniques based on chemical precipitation is that the products recovered, such as iron phosphate and aluminium phosphate, have a very low solubility and thus cannot release phosphorus at rates sufficient for crop demand. Their fertiliser value is therefore low. Furthermore, the separated precipitates cannot be processed by the phosphate industry, since iron and aluminium cause undesirable reactions. In other approaches, phosphorus is leached from the ash using an alkali, followed by precipitation as calcium phosphate (Stendahl & Jäfverström, 2003, 2004; Nishimura, 2003) or sodium phosphate (Ek, 2005). Ash dissolution with an alkali is inefficient as only a minor proportion of the phosphorus (< 50%) can be leached, whereas dissolution with an acid achieves almost complete phosphorus leaching (Cohen, 2009). Another process for phosphorus recovery involving dissolution of ash from sewage sludge in acid using ion exchange and recovering the phosphorus in the form of phosphoric acid has been suggested by Jensen (2000). Hong et al. (2005) describe how phosphoric acid can be extracted with organic solvents after dissolution of incinerated sewage sludge ash with sulphuric or hydrochloric acid. Another process is based on heating the ash up to 1,400°C to vaporise elemental phosphorus, which is condensed in water and oxidised to phosphoric acid (Japanese patent 9145038, 1997). Heating sludge ash to evaporate the phosphorus requires large amounts of energy and the efficiency of phosphorus recovery is moderate due to the formation of iron phosphate slag (Schipper et al., 2001). A process for thermochemical removal of heavy metals from sludge ash and use of the residue as a fertiliser has been developed by the company AshDec (Herman, 2009; Adam et al., 2009; Mattenberger et al., 2008, 2010). A new process for production of ammonium phosphates, called CleanMAP™ Technology, has been developed by the company EasyMining Sweden AB (EasyMining, 2011). The technology enables production of pure mono-ammonium (MAP) or di-ammonium phosphate (DAP), irrespective of the quality of the phosphorus raw material, and is based on selective liquid-liquid extraction coupled with precipitation. Cadmium, uranium and other metals are separated out and not incorporated into the fertiliser. Furthermore, the costs are lower than those of state-of-the-art technology for phosphorus fertiliser production and no energy is required for water evaporation. Energy savings of around 5 tons steam per ton phosphorus are achieved compared with state-of-the-art fertiliser production. The technology can be used for phosphorus extraction from phosphate rock and other raw materials such as ash (of incinerated sewage sludge, slaughterhouse wastes or incinerated manure). Processing the ash of incinerated sewage sludge includes the following steps: (i) ash is dissolved in sulphuric acid and insoluble material is separated out and washed; (ii) phosphate ions are recovered from the leach solution as mono-ammonium phosphate using the CleanMAP™ Technology; (iii) iron and aluminium ions mainly originating from phosphorus removal during wastewater treatment are recovered in hydroxide or sulphate forms to be reused for phosphorus precipitation in wastewater treatment plants; and (iv) remaining dissolved heavy metals are removed from solution as sulphides upon precipitation with sodium sulphide. Outgoing water from the process has neutral pH and a low phosphorus and metal content. In summary, the advantages of this process are that the chemical used for phosphorus separation (ammonia) becomes part of the product, the product is concentrated and water-soluble, and metals are separated out during the process and the fertiliser is of high quality.

5. Soluble phosphorus fertilisers – essential for efficient use in agriculture

Rock phosphate is the raw material from which all types of phosphate fertilisers are produced. Most rock phosphates are not suitable for direct application to soil, since they are insoluble in soil and water. Phosphate rock is therefore processed by the fertiliser industry into soluble fertiliser with high plant availability (see section 2). The four most common phosphorus fertilisers are mono-ammonium phosphate (MAP), di-ammonium phosphate (DAP), single superphosphate (SSP = mono-calcium phosphate + gypsum), and triple superphosphate (TSP = mono-calcium phosphate), all with high water solubility. Ammonium phosphates dominate worldwide (Fig. 4), and the phosphorus in NPK compound fertilisers is usually based on one of these compounds.

It is commonly believed that the use efficiency of phosphorus fertiliser by crops is low, ranging from 10 to 25% based on calculations of the difference between crops fertilised with phosphorus and unfertilised controls (Crowther et al., 1951; Mattingly & Widdowson, 1958, 1959; Johnston & Richards, 2003b). However, through calculations based on the balance between inputs and outputs of phosphorus, Syers et al. (2008) showed that a use efficiency of 95% could be obtained at optimal phosphorus levels by replacing the amount removed with harvest with soluble phosphorus fertiliser. A substantial proportion of the phosphorus added with fertilisers to soil was found to be utilised by crops during following years, which means that some fertiliser phosphorus accumulates in the soil in reversible residual forms. Thus, the objective should be to maintain the amount of readily plant-available soil phosphorus at the optimum level. It is therefore important that phosphorus fertilisers are highly soluble in water.

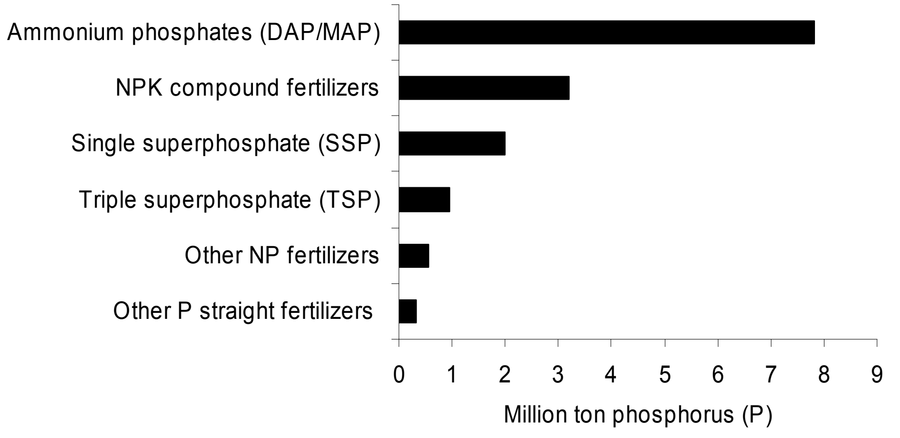

Fig. 4. World consumption of phosphorus fertilisers in 2008. Data from IFA (2010).

6. Conditions necessary to achieve efficient nutrient cycling

An environmental target in modern societies is to recycle nutrients back to agricultural land in a sustainable way. Therefore, municipal wastes must be 'safe and clean'. In order to achieve this target, a number of actions have been taken. For example in EU countries, landfilling of organic material has been prohibited. The use of certain metals (e.g. cadmium, mercury) has been prohibited or is highly restricted to reduce contamination of wastes. Industries connected to sewage treatment plants must keep discharge of pollutants at a minimum to avoid contamination of sewage sludge. Source-separation of household wastes has been introduced to produce composts without contaminants. These efforts have improved the quality of municipal wastes. For example, the cadmium level in sewage sludge in Sweden has declined from rather high concentrations to only 20-40 milligrams per kilogram phosphorus (Eriksson, 2009).

However, it is questionable whether these commendable improvements will result in long-term use of municipal wastes on arable land, considering that a number of conditions must be fulfilled for sustainable recycling. Table 4 summarises the most important conditions that must be fulfilled to achieve sustainable recycling of municipal wastes back to soil, including: (i) 'safe and clean' wastes that have a negligible effect on the soil and environment (refers to their possible content of metals, organic contaminants and pathogens); (ii) high plant availability of nutrients in wastes to give a significant fertiliser effect (i.e. if nutrients in wastes are bound in less soluble or insoluble form, recycling will not replace inorganic fertilisers); and (iii) redistribution of nutrients to arable land through wastes must be related to nutrient removal (i.e. the 'law of nutrient replacement' should be followed).

Nutrients removed from soil through harvest and losses should be replenished with equivalent amounts. Application of excessive amounts to arable land is unacceptable and long-distance transportation would be required to achieve equitable redistribution while avoiding accumulation of nutrients in arable land surrounding cities. It seems that all these conditions can only be achieved if handling of municipal organic wastes in society is greatly improved.

Condition for nutrient cycling
No adverse effect on food quality and the environment
Low levels of unwanted metals
Low levels of organic pollutants
Low levels of pharmaceuticals
Low levels of pathogens
Efficient nutrient supply
High plant availability
Low nutrient losses
Equitable redistribution and spreading on arable land
Long-term transportation
Energy-saving compared with mineral fertiliser use

Table 4. Defining conditions for recirculation of nutrients in wastes back to agricultural land to achieve sustainable management.

7. Closing the phosphorus cycle in society through incineration of phosphorus-containing wastes and fertiliser production from ash

As pointed out above, the demographic trend for increasing urbanisation makes towns and cities hot-spots for accumulation of nutrients and metals, which can cause biogeochemical imbalances (Grimm et al., 2008). The accumulation of wastes in mega-cities is managed through dumping, partial recycling (e.g. Færge et al., 2001) and incineration (Donatello et al., 2010). Re-applying nutrients to arable land only as recycled organic wastes is not a viable option any longer (Fig. 3). The trend for incinerating more sewage sludge, not only in mega-cities but also in cities and towns, means that spreading of sewage sludge will decrease in future. We consider ash to be the main waste product from increasing urbanisation and processing of ash for nutrient extraction to be an important step to close nutrient cycling in society (Fig. 5). The approach of not recycling urban organic wastes as such but producing

inorganic fertilisers from nutrients present in wastes has been proposed earlier (Kirchmann et al., 2005).
Incineration of manure to recover energy is also increasing, as other treatment options are often more expensive when large volumes have to be handled. Phosphorus and potassium can both be extracted from ash of incinerated manure. For example, ash of poultry litter can contain up to 10% phosphorus and 8% potassium (Blake & Hess, 2011). A considerable amount of the potassium in ash of poultry litter originates from bedding materials such as straw. The company Kommunekemi AS has developed a process for production of pure potassium fertilisers (potassium chloride or sulphate) from straw ash contaminated with cadmium (Ottosson et al., 2009). Potassium and phosphorus are major plant nutrients that are obtained through mining of non-renewable minerals. In the past, potassium was in fact obtained from ash. The term 'potash', which commonly refers to potassium-containing minerals, originates from the old method of leaching wood ash and evaporating the solution in large iron pots to obtain potassium carbonate (Prud'homme, 2011).
The potassium content of sewage sludge is very low (0.1% of dry matter in activated sludge) (Binnie et al., 1995), since potassium is generally water-soluble and not incorporated into the solid phase. Therefore, recovery of potassium from sludge ash is not relevant. However, about 20% of the nitrogen in sewage water is incorporated into sewage sludge (Palmgren, 2005). This cannot be recycled if sewage sludge is incinerated, since during sludge incineration the nitrogen in organic matter is oxidised into nitrogen gas. However, nitrogen is not a limited resource like phosphorus or potassium, since it is recovered from air (air contains approx. 78% nitrogen) and thus the nitrogen lost during incineration of sewage sludge has little relevance for agriculture. For example, the total sewage sludge produced in Sweden every year contains around 5000 tons of nitrogen, which is equivalent to around 2 kilograms per hectare arable land. The nitrogen demand of crops usually varies between 80 and 150 kg per hectare and year. Furthermore, during sludge incineration organic matter is oxidised into carbon dioxide but as sludge is based on renewable biomass, it is a carbon-neutral fuel. The role of sewage sludge as a source of organic matter for agriculture is minor. For example, the organic matter content of sewage sludge is about 60% of dry matter

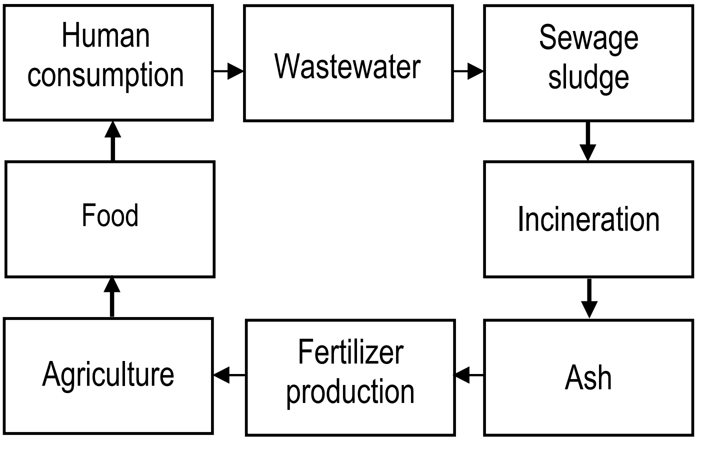

Fig. 5. Incineration of sewage sludge followed by nutrient extraction and fertiliser production from ash as a way to close the food cycle in society.

(Palmgren, 2005), which for Sweden corresponds to a supply of 48 kilograms per hectare arable land and year through sewage sludge. The average supply of organic matter through crop residues is about 2000 kilograms per hectare and year. The conclusion is that incineration of sewage sludge does not represent a great loss of valuable resources. Recovery of only phosphorus and precipitation chemicals seems to be an acceptable strategy.

8. Conclusions

Recycling of the phosphorus present in wastes back to arable land is a key issue in achieving sustainable phosphorus management. According to the conditions and difficulties described above, it appears that recirculation of extracted phosphorus from municipal wastes rather than whole waste redistribution on arable land is a logical way forward.

One approach would be to incinerate phosphorus-rich wastes, followed by further treatment of the ash for phosphorus extraction. The extracted phosphorus should be concentrated to enable redistribution to all arable land, allowing long-distance transportation, and non-contaminated to ensure safe food production. In addition, the recycled fertiliser should be water-soluble, containing plant-available nutrients and having the same fertiliser value as mineral fertiliser. Extraction of phosphorus fertiliser from ash has the potential to partly replace rock phosphate-based fertilisers. The trend for increasing incineration of municipal wastes is in line with this approach.

In future, wastes will be re-used, replacing valuable raw materials. We believe that continued technological development of waste treatment will lead to sustainable nutrient recycling in society, characterised by efficient processes and high quality products.

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14

On-Farm Composting of Dead Stock

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1. Introduction

Rendering and on-farm burial are the predominant methods used by farmers for the routine disposal of domesticated farm animals. However, throughout many developed countries, knackery and rendering services have been contracting and many farmers are seeking alternative stock disposal options. In parts of Australia, for example, where many knackeries have been closing in recent years, the illegal dumping of dairy cattle in waterways has become a serious problem. Bonhotal et al. (2002) reported that the improper disposal of dead stock in New York State and Pennsylvania was becoming more widespread as farmers no longer had access to affordable rendering services. Dumping of stock is not only a risk to water quality but is also a biosecurity hazard and the source of many complaints to environment protection agencies from neighbours and downstream users.

On-farm burial is one of the simplest and most cost-effective methods of carcass disposal, but this too is becoming restricted as environment protection agencies seek to protect water resources from contamination. It has effectively been eliminated as an option for mass disposal in Virginia following the unearthing of intact 15 year old avian influenza-affected poultry carcasses at a trench burial site in the late 1990s (Malone, 2005). On-farm burial is also not possible in many irrigated areas where the watertable is close to the surface or where surface waters are close to the disposal site.

Composting is a natural biological decomposition process that takes place under aerobic, thermophilic conditions. It can be used for the day-to-day management of mortalities on farms and for carcass disposal in emergency animal disease (EAD) outbreaks. In mortality composting, carcasses are placed in piles or bins together with supplemental carbon sources such as sawdust, litter, straw or wood shavings. Composting is particularly suitable for broiler-farm mortalities and litter. In the case of EAD outbreaks, composting can be conducted either inside or outside the poultry house following euthanasia (Kalbasi et al., 2005; Mukhtar et al., 2004).

2. The mortality composting process

2.1 Brief comparison with conventional composting

In conventional composting systems, raw materials are mixed together to form a pile of relatively uniform nutrient content, particle size, porosity and moisture content. Mesophilic microorganisms first use the readily degradable substrates such as sugars, starch and proteins, and provided that the pile is of sufficient volume (usually >1m^3), temperatures rise rapidly. The materials may be turned every few days to move the cooler outside layers into the centre of the pile, and to allow air to move more freely into the pile. In other systems, air is forced into the pile by a thermostatically controlled fan.

This first stage of composting (6–12 weeks duration) is characterised by high temperatures and rapid rates of decomposition and is usually termed the thermophilic stage or period of 'intensive decomposition' (Haug, 1993). These conditions result in the elimination of nuisance odours and destruction of pathogens and weed seeds. It is during this stage that substrates such as fats, hemicellulose and cellulose are degraded. As the composting process proceeds and the availability of substrate become more limiting, temperatures begin to fall. This second stage of composting (lasting for 4+ weeks), called the maturation or curing phase, takes place under mesophilic conditions (under 45°C) and is characterised by lower rates of biological decomposition under which aeration is no longer a limiting factor. During this stage, the biologically resistant substrates such as lignocellulose and lignin are degraded. The maturation phase of composting has a large bearing on the suitability of the end product for a particular use (Brewer & Sullivan, 2003; Wilkinson et al., 2009).

Many authors have defined various optima for the composting process, including a carbon to nitrogen ratio (C:N) of between 25:1 and 30:1, moisture content within the range of 50–60% (w/w), porosity of 35–45% and oxygen levels of >10% by volume (Table 1). But these optima were developed for relatively homogenous organic materials such as manures, green waste, food wastes and biosolids and have questionable relevance to mortality composting.

Characteristic	Optimum	Reasonable range
Carbon to nitrogen ratio (C:N)	25–30:1	20–40:1
Moisture content	50–60% (wet basis)	40–60% (wet basis)
Porosity	35–45%	30–50%
Oxygen concentration	>10%	>5%
Bulk density		<640 kg/m^3
pH	6.5–8.0	5.5–9.0

Table 1. Desirable characteristics for composting (modified from Keener et al., 2006; Northeast Regional Agricultural Engineering Service, 1992).

In contrast, a livestock mortality composting pile is a heterogenous mixture, so strict application of the principles discussed above is not possible. A mortality compost pile may contain an animal of large mass, having a high moisture content, low C:N ratio and nearly zero porosity, surrounded by a material (the carbon source) with a high C:N ratio, moderate moisture level and good porosity (Keener & Ellwell, 2006). Kalbasi et al. (2005) aptly described mortality composting as the above ground burial of dead animals in a mound of supplemental carbon such as sawdust, litter, straw or wood shavings. Sufficient supplemental carbon is required around the carcass to absorb bodily fluids and to prevent odours from escaping from the pile.

The decomposition process is initially anaerobic in and around the carcass layer, but as odorous gasses are produced and diffuse away, they enter an aerobic zone where they are degraded to CO_2 and water (Keener & Ellwell, 2006). In contrast to conventional composting systems, temperatures in mortality composting are initially higher in the outer aerobic layers of the pile compared to the interior (Fig. 1). Oxygen diffuses only slowly into the interior of the pile as the carcass layer degrades, resulting in a delay of 2-3 days before thermophilic conditions are reached.

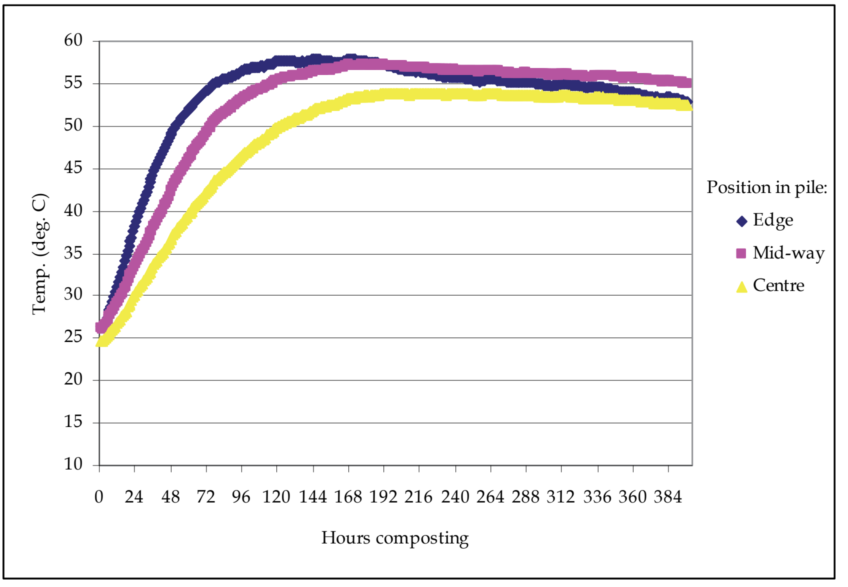

Fig. 1. Temperature development in poultry mortality composting according to position in the pile. Source: Wilkinson, unpublished.

Mortality composting is generally conducted in 3 stages. In the primary stage of composting, the pile is left undisturbed as soft tissue decomposes and bones partially soften. The compost is usually then moved, turned or mixed to begin the secondary stage, during which time the remaining materials (the remaining meat and bones) break down further. Following completion of the secondary phase, the composting process is completed during a curing or storage phase.

Some bones of large mature animals may remain after completion of the secondary and/or storage stages of composting, but these are usually quite brittle and pose no health risk and will not damage farm equipment when applied to land (Keener & Ellwell 2006; Mukhtar et al., 2003). Nevertheless, Murphy et al. (2004) observed that the moisture content of a composting pile has a major bearing on the rate of decomposition of bones from cattle mortalities. If the pile is allowed to dry out, bones become very hard and appear to cease decomposition. Continued decomposition of the bones is achieved by wetting the pile on a monthly schedule for a period of about 6–9 months.

The time to completion of composting varies with the size of the animal, the compost formulation (e.g. type of carbon (C) sources used) and the management of the pile (e.g. mixing, turning and watering). As a general rule, the first stage of composting is complete in as little as 10 days for small animals such as poultry, about 90 days for medium sized animals such as pigs and over 6 months for large carcasses (Mukhtar et al., 2004).

3. Mortality composting system design and layout

3.1 Main systems

Mortality composting began in the poultry industry in the USA in the early 1980s and soon spread to other industries and has also been used for road kill. The basic forms of mortality composting are conducted either in bins or piles/windrows.

Bin composting is usually conducted in a three-sided enclosure on a hard stand (e.g. concrete or compacted soil). It may or may not be covered by a roof, though a roof is usually required in high rainfall areas. Designs are available on-line for purpose-built constructions with concrete floors, roofs and wood or concrete side-walls (Fig. 2). In its simplest form, the walls can be constructed of hay bales or any such material that can adequately confine the composting pile (Mukhtar et al., 2003). Simple bins can also be constructed from pallets or wood and plastic mesh. These are sometimes termed 'mini-composters' and are suitable for small animals such as poultry, rabbits, piglets and fish (Brodie & Carr, 1997).

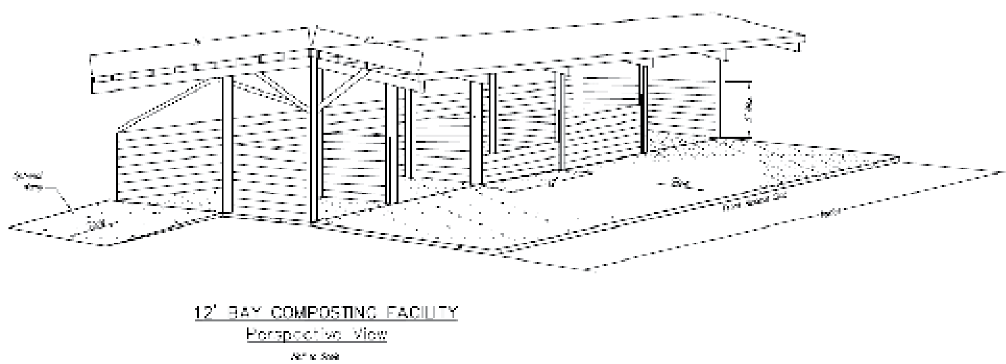

Fig. 2. Diagram of a dead bird composting facility. Additional detailed drawings can be found at the USDA National Resources Conservation Service website, http://www.oh.nrcs.usda.gov/technical/engineering/cadd2_dwg_a_to_c.html.

At least 3 bins are usually in operation at any one time—one being filled, another in the primary stages of composting and the other in the secondary stages of composting. A pile is sometimes substituted for the secondary bin in two bin systems (Keener et al., 2000). Bins are usually only used to compost small-/ and medium-sized carcasses. As a general guide, 10 m^3 of bin space is required for every 1,000 kg of carcass (Mukhtar et al., 2004).
Piles for mortality composting are usually constructed in the open on a hard stand. Placing a plastic or geotextile liner under windrows as a moisture barrier is recommended when a concrete pad is not available. Access to the pile from all sides should be possible and the pile is shaped to shed rainfall. Windrows are formed by continually extending the length of the pile with the addition of further mortalities and supplemental carbon. The length of the windrow is determined by loading rates and site layout. Mukhtar et al. (2004) described the recommended dimensions of windrows according to the relative sizes of carcasses:

- Small carcasses (<23 kg): bottom width, 3.6 m; top width, 1.5 m; and height, 1.8 m.
- Medium carcasses (23–114 kg): bottom width, 3.9 m; top width, 0.3 m; height, 1.8 m.
- Large and very large carcasses (>114 kg): bottom width, 4.5 m; top width, 0.3 m; height, 2.1 m.

New poultry operations in the United States frequently build mortality composting facilities along the side of a manure shed (Fig. 3). The roof-line is simply extended to create a channel down one side of the shed. Piles of compost can then be constructed under it using the manure which is stored in the main shed adjacent to it.
In-vessel composting systems have also been used for composting carcasses. In-vessel systems enclose composting materials in a sealed chamber or vessel where environmental

parameters such as temperature and aeration can be better controlled than in a pile or windrow. Examples include rotary composters, the BiobiN™ and the Ag-Bag® in-vessel system. The BiobiN™ system is offered as a contracted service to the poultry industry in Australia. Bins of up to 9 m^3 in size are delivered to the poultry facility and, when full, are transported to a licensed composting facility to complete composting. The BiobiN™ is a fully enclosed system with forced aeration and a biofilter to control odours and leachate.

Fig. 3. Composting facility constructed on the side of manure sheds at poultry facilities, Delmarva Peninsula, USA. Photos: K. Wilkinson.

The Ag-Bag® in-vessel system was used for the disposal of 1 million avian influenza-negative birds during an EAD outbreak in British Columbia in 2004 (Spencer et al., 2005). The poultry carcasses and C source were mixed together and pushed into the Ag-Bag®. The Ag-Bag® composting system was also used to dispose of 43,000 birds in the low-pathogenic avian influenza outbreak in Virginia during 2002.

3.2 Site selection and layout

The following general principles apply to site selection and layout for on-farm composting of mortalities (Mukhtar et al., 2004; Keener et al., 2006):

- The site should be in an elevated area of low permeability, at least 1–2 m above the watertable and not within 100 m of surface waters (e.g. streams, lakes, wells etc).
- The site should have an adequate slope (1–3%) to allow proper drainage of leachate and prevent pooling of water.
- Consideration should be given to prevailing winds and the proximity of neighbours to minimise problems associated with odour and dust.
- Run-off from the compost facility (e.g. from a 25-year, 24 hr rainfall event) should be collected and directed away from production facilities and treated through a vegetative filter strip or infiltration area.
- The site should have all-weather access and have minimum interference from other traffic.
- Maintaining an effective cover of C source over compost piles is usually sufficient to eliminate scavenging animals and vermin. But animals will dig into piles when they know mortalities are contained in them, so fencing should be installed around piles and bins to minimise this problem.

4. The mortality composting process in detail

4.1 Carbon sources

A wide range of carbon (C) sources can be used for mortality composting, including sawdust, wood shavings, green waste, chopped straw, manure, poultry litter and other bedding materials. The three most important properties that influence the performance of different carbon sources in mortality composting are available energy (biodegradability), porosity and moisture absorbency.

Sawdust is probably the most common C source used for mortality composting, as it is highly absorbent, allows high temperatures to be sustained and sheds rainwater when used for uncovered piles. According to Imbeah (1998), carbon sources like sawdust and rice hulls are ideal for mortality composting because their particle size allows them to settle intimately around the carcass to provide optimum contact.

Researchers rarely identify the type of C source beyond the generic term ‘sawdust’ despite the fact that the biodegradability of sawdust between timber species can differ by a factor of more than 10. Data from Allison (1965) showed that hardwoods had significantly higher biodegradability than softwoods but there was considerable variation between various species, especially in the softwood family.

The absorbency of different types of bedding materials is also known to differ greatly (Burn & Mason, 2005; Misselbrook & Powell, 2005). In general, softwood sawdusts are more absorbent than hardwood sawdusts. The absorbency of a C source will influence the depth of the base layer that is needed to absorb liquids during composting, but also the performance of the outer layers as a biofilter.

Research by Ohio State University found that some C sources such as chopped straw or cornstover can be used in mortality composting piles, but they require periodic addition of water to maintain composting conditions (Keener & Elwell, 2006). King et al. (2005) compared the performance of 11 different types of C sources for composting large carcasses (horses and cows). They reported that coarsely structured C sources such as wood shavings or wood chips experienced problems with odour, leachate and vector attraction. Glanville et al. (2005) studied straw/manure, corn stalks and corn silage as C sources for 450 kg cattle carcasses in windrows. From a biosecurity standpoint, corn silage performed best as it consistently produced the highest internal temperatures and sustained them for the longest time but it did not result in noticeably shorter carcass decay times.

In practice, a wide range of carbon sources can be successfully used in mortality composting. The choice of material is likely to be based on cost, availability and performance. It is commonly advised to incorporate up to 50% of finished compost into the base and cover C sources (Kalbasi et al., 2005; Keener & Elwell, 2006; Mukhtar et al., 2004). The recycling of finished compost in this manner reduces the cost of purchase of raw materials, speeds up the initiation of composting conditions and reduces the space required for storage of finished compost. To facilitate faster rates of decomposition, some researchers recommend that carcasses should be added to C sources that are actively composting or those that have an ideal C:N ratio for composting (Kalbasi et al., 2005; King et al., 2005). The inclusion of too much finished compost in the initial mixture sometimes reduces decomposition rates because of a lack of available energy in the compost or reduced porosity in the final mix (Keener & Elwell, 2006; Murphy et al., 2004).

4.1.1 Determining requirement for carbon

Recommendations differ on the amount of carbon required to compost mortalities. These include:

- A 12:1 sawdust to mortality volume ratio for all types of mortality (Keener et al., 2000).
- About 9.5m^3 of C source for fully-grown cattle (Bonhotal et al., 2002).
- A carcass:straw:manure volume ratio for poultry of 1:0–1.2:4–8 (Natural Resources Conservation Service, 2001).
- A 2:1 C-source to mortality volume ratio for poultry, not including the requirement for base layer and capping (Tablante & Malone, 2005).

The requirement for carbon can be estimated for composting all types of mortalities in either bins or static piles/windrows when the annual mass of mortality is known. The annual sawdust requirement in m^3/yr, V_s, is

$$V_s = YL \times 0.0116 \quad (1)$$

where YL is the yearly mortality loss in kg/yr (Keener et al., 2000). Equation 1 gives the total annual requirement, but up to 50% of this can be met by replacement of fresh sawdust with finished compost.

4.2 Pre-treatment of carcasses

The burial of mortalities above the ground in a pile of carbonaceous material does not necessarily result in optimum conditions for composting because of the heterogenous nature of the mix. But leaving the carcasses undisturbed until they are largely broken down has obvious advantages for biosecurity, particularly in an EAD outbreak. Nevertheless, Rynk (2003) demonstrated that chopping large carcasses in a vertical grinder-mixer (the type used for grinding hay and mixing feed rations) produces a homogenous mixture for composting and reverses the normal requirement of C source to mortalities from 4:1 to 1:4 by mass. Finely chopping large carcasses also results in a significant reduction in required composting time from about 180 days down to as low as 75 days. All of this has a significant effect on the economics of mortality composting. The advantages of chopping the carcasses of smaller animals, like poultry, are less clear because they typically break down much more quickly than large carcasses.

Combining chopping and/or mixing of carcasses with the use of in-vessel type composting systems (e.g. the Ag-Bag® system) could be feasible for disposing of non-diseased birds in an EAD outbreak.

Rynk (2003) described the advantages of this sort of approach to include:

- Mortalities are isolated from the environment, reducing the risk of odours and scavengers plus the effects of the weather.
- The containment reduces the amount of C source required because the carcasses do not need to be fully covered and the need to absorb liquids is not as critical.
- The added degree of process control in in-vessel type composting systems (e.g. forced aeration) tends to accelerate the composting process compared to passively aerated systems.

4.3 Bin composting

A base of sawdust or other suitable C source of 20-30 cm thickness should be placed on the floor of the bin to collect liquids that are released during composting. Larger animals may require a deeper base layer (up to 60 cm deep). Mukhtar et al. (2004) suggested that the ideal base layer is pre-heated litter, put in place about 2 days before carcasses are added. Carcasses can be layered within the bin with about 15–30 cm of absorbent bulking material (e.g. litter or sawdust) placed between each layer of mortalities. Mortalities must not be placed within 20–30 cm of the sides, front or rear of the bin. A final cover of damp sawdust or litter to a depth of about 60 cm should be placed on the top of the pile (Fig. 4). This final cover acts as a biofilter for odour control and to insulate the heap. When the cover material is too dry or too wet, odours may be released and scavenging animals may be attracted to the pile (Keener & Elwell, 2006).

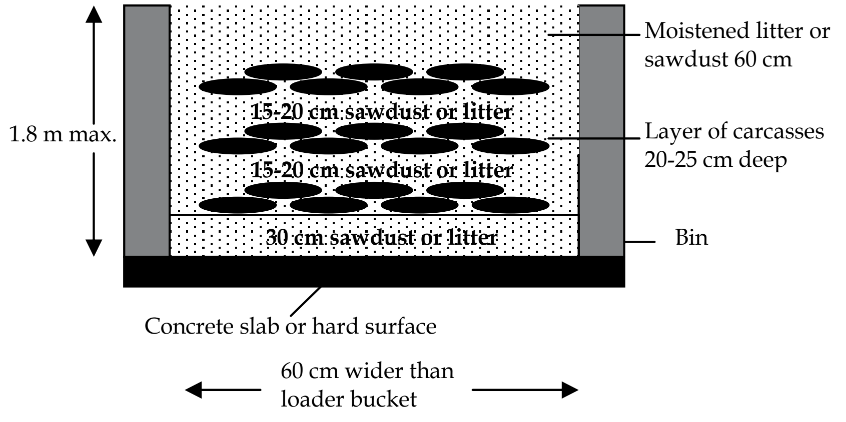

Fig. 4. Typical layout of a mortality composting bin for small animals (adapted from Keener & Elwell, 2006; Tablante & Malone, 2005).

The pile is moved to a secondary bin when the last layer of mortalities is almost completely decomposed. To ensure that the pile reheats, it is watered and re-mixed. An additional 10 cm of co-composting cover material is added to ensure that any carcass pieces remaining are covered and odours are minimised. When additional animals are to be added to a partially filled bin, half of the cover material is removed and a new layer of animals is placed on top. The new layer of mortalities is then covered with 60 cm of damp C source.

Stanford et al. (2000) used a bin (2.4 x 2.4 x 2.4 m) constructed of pressure treated timber to successfully compost lambs and mature sheep in both summer and winter conditions of Alberta, Canada. Alternate layers of composted sheep manure, barley straw and fresh sheep manure were used above and below a layer of mortalities. The expected heating pattern was not observed in one trial due to the excessive moisture content (31% dry matter) of the fresh sheep manure that was added to the bin. In this trial, 6 wethers (mean mass of 97.5 kg) were composted in a single layer over autumn and winter. Foul odours were observed when the contents of the bin were transferred to the secondary bin after 79 days. However, turning the compost into the secondary bin salvaged the pile and temperatures reached over 60°C even though the average ambient temperature was only -6.7°C (with a low of -35°C).

4.4 Pile or windrow composting

Large and very large animals (e.g. mature cattle and pigs) are most suited to the windrow composting method. It is also the system that is most likely to be used in any mass mortality composting process. Keener et al. (2000) stated that for mature cattle or horses, it is preferable to construct a separate pile for each carcass.

Mukhtar et al. (2004) suggested that a base layer of C source should be 30 cm thick for small carcasses, 45 cm for medium carcasses and 60 cm for large carcasses. An ideal base layer for this purpose has been described as absorbent organic material containing sizeable pieces 10–15 cm long such as wood chips (Bonhotal et al., 2002). Another layer (15–30 cm thick) of highly porous, pack-resistant bulking material can be added on top of the base layer to absorb moisture from the carcasses and to maintain adequate porosity. The dimensions of these base materials must be large enough to accommodate the mortalities with >60 cm space around the edges (Figs. 5 & 6).

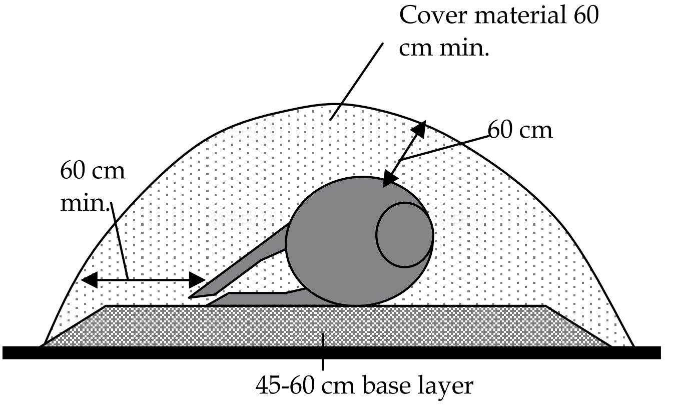

Fig. 5. Cross-section of a typical windrow or static pile for larger carcasses.

An evenly-spaced layer of mortalities can then be placed on top of this and covered with between 30 cm and 60 cm of C source. Some guidelines recommend the use of a dry cover (e.g. Bonhotal et al., 2002), whereas others claim a moist C source reduces odours and assists in the breakdown of bones (Keener & Elwell, 2006; Murphy et al., 2004).

Small-/ and medium-sized carcasses can be layered in windrows with at least 30 cm of C source placed between each layer until the windrow reaches a height of approximately 1.8 m. With larger carcasses, only a single layer of mortalities should be placed in a windrow before it is capped with C source (Fig. 6).

For ruminants larger than 136 kg, it is usually recommended to lance the rumen and/or thoracic cavity to avoid bloating and possible explosion (Bonhotal et al., 2002).

Straw bales were used by Murphy et al. (2004) to confine a U-shaped site of dimensions 2.6 m by 2.6 m and 1 m deep for composting beef cattle (275–450 kg). As base layers and covers, they used straw, manure compost and sawdust separately and in combination (i.e. 2 C sources in equal quantities). All six permutations of C sources produced an acceptable decomposition of the cattle mortality and no odours were observed. However, it was noted that straw and sawdust piles produced a more rapid rise in temperature and shorter times of decomposition.

Mukhtar et al. (2003) investigated a low-maintenance approach to composting cattle and horses in spent horse bedding (pine wood shavings and horse manure). The animals were composted in the bedding with or without wooden pallets under them (both on a 46 cm base layer). It was assumed that the air spaces between the pallets and the bedding layer underneath them would continue to aerate the static pile and that these piles would require less turning. The effect of the pallets was inconclusive as both methods worked successfully and the animals composted were of different sizes. Nevertheless, the trials showed that peak temperatures were often associated with the moist bottom layers of the pile as the upper layers dried out. Temperatures in the upper layers of the pile increased in response to rainfall.

Fig. 6. Construction of compost pile for a large carcass. Photos: J. Biala & K. Wilkinson.

In static piles of poultry mortalities, straw and hen manure, González & Sánchez (2005) found some influence of ambient temperatures and different mixes on the progress of composting. During summer, the carcasses were exposed to temperature above 60°C for between 4 and 20 days depending on the particular mix used. In winter, peak temperatures were lower, but still exceeded 55°C in each pile.

4.5 Monitoring composting conditions

The progress of composting is monitored primarily with a temperature probe. Temperature is the single most important indicator of the stage of degradation, the likely pathogen kill and the timing of turning events (Keener & Elwell, 2006). Temperatures should be taken at several points near the carcasses in a pile—for example with the use of a stainless-steel temperature probe 90–100 cm in length. A logbook should also be used to record data such as dates, mass of carcasses, temperature, amount and types of C sources used and dates when compost is turned (Mukhtar et al., 2004).

4.6 Managing environmental and public health impacts

Improper carcass disposal may cause serious environmental and public health hazards, including:

- Generation of nuisance odours resulting from the anaerobic breakdown of carcasses.
- Leaching of nutrients from carcasses to ground and surface water.
- Spread of pathogens from infected carcasses via equipment, personnel, air, soil or water.

- Flies, vermin and scavengers disrupting operations and acting as potential vectors of harmful diseases.

Many of these potential hazards are managed by paying careful attention to site design and layout. The biological risks associated with mortality composting are principally managed by proficient operation of the composting process.

The environmental impacts of cattle carcass composting were investigated by Glanville et al. (2005). Trials were conducted in 6 m x 5.5 m x 2.1 m windrow-type test units containing four 450 kg cattle carcasses on a 60 cm thick base layer of C source. C sources included corn silage, ground cornstalks or ground straw mixed with feedlot manure.

During the first 4–5 weeks after construction, air samples were collected on a weekly basis from the surface of the test units and compared with stockpiles of cover materials (i.e. not containing mortalities). Threshold odour levels were determined by olfactometry using experienced odour panellists and standard dilution procedures. It was found that 45–60 cm of cover material was generally very effective at retaining odorous gasses produced during composting. Threshold odour values for the composting test units were often very similar to the odour intensities found in the cover material stockpiles.

Chemical analysis of the leachate collected in PVC sampling tubes installed at the base of the test units showed that it had high pollution potential (Glanville et al., 2005). The leachate had mean ammonia concentrations of 2,000–4,000 mg/L, total organic C of 7,000–20,000 mg/L and total solids of 12,000–50,000 mg/L. Nevertheless, the base and cover materials were highly effective in retaining and evaporating liquids released during composting as well as that contributed by seasonal precipitation. Following a 5-month monitoring period after the set up of the trial, the test units received nearly 546 mm of precipitation yet released less than 9 mm of leachate each.

In Nova Scotia, Rogers et al. (2005) investigated the environmental impacts of composting pigs in sawdust and pig litter (manure plus bedding). Leachate and surface run-off were collected and analysed for various water quality parameters. Highest temperatures and better carcass decomposition were observed with sawdust in both the primary and secondary stages of composting. The sawdust cover also had lower leachate and surface run-off volumes and annual nutrient loadings compared to the pig litter treatments.

Finished mortality compost should be applied to land in a manner similar to manure so that the nutrient uptake capabilities of the crop being grown is not exceeded. A comparison of the nutrient composition of poultry litter and mortality composts is shown in Table 2.

Poultry mortality compost often has a higher nutrient content than other composts, probably as a result of the high nutrient content of poultry litter (Table 2). During composting, much of the available nitrogen is converted to organic forms and becomes unavailable in the short-term to plants.

Murphy & Carr (1991), for example, demonstrated much slower rates of N mineralisation in a loamy sand amended with poultry mortality composts compared to manure. Thus there is a lower risk of nutrient leaching with compost compared to uncomposted manures and mortalities. Nevertheless, it is advisable not to spread mortality compost in sensitive areas such as watercourses, gullies and public roads.

5. Mass mortality composting

The use of mortality composting as the main method of carcass disposal on a mass-scale (known as mass mortality composting) is probably only likely for small/- to medium-size carcasses. Until recently, most mass mortality composting operations were conducted after

	Lamb mortality compost[1]		Sheep mortality compost[1]		Poultry litter[2]	Poultry mortality compost[3]	Poultry mortality compost[4]
	Starting compost	Finished compost	Starting compost	Finished compost	Un-composted	Finished compost	Finished compost
	Mean (SD)		Mean (SD)		Mean (SE)	Mean (SD)	Mean (SD)
DM (%)	52.7 (8.1)	65.3 (5.5)	64.6 (1.4)	50.6 (5.4)	80.5 (0.58)	85.41 (11.31)	63.8 (10.62)
Total C (%)	23.5 (0.8)	23.1 (2.0)	23.5 (1.4)	28.3 (2.9)		27.40 (15.75)	36.3 (3.83)
Total N (%)	1.6 (0.1)	1.8 (0.2)	2.00 (0.2)	2.3 (0.2)	4.00 (0.72)	2.42 (0.93)	3.80 (0.55)
C:N ratio	14.3 (0.8)	12.7 (2.1)	11.9 (0.4)	12.2 (2.0)		10.96 (2.01)	9.8 (0.16)
Total P (%)	0.6 (0.0)	0.8 (0.1)	0.8 (0.1)	0.9 (0.1)	1.56 (0.047)	3.1 (0.91)	1.8 (0.55)
Total K (%)	2.42 (5.0)	12.16 (2.28)	14.31 (2.62)	13.55 (1.35)	2.32 (0.059)	2.88 (1.82)	2.1 (0.55)

[1]Stanford et al. (2000). Compost composed of mortalities, straw, manure and composted manure. Number of samples not given.
[2]Stephenson et al. (1990). Analysis of 106 broiler litter samples collected in Alabama, USA.
[3]González & Sánchez (2005). Analysis of 8 samples of compost with different ratios of straw, hen manure and poultry mortalities.
[4]Cummins et al. (1993). Analysis of 30 poultry mortality composts collected from farms in Alabama, USA.

Table 2. Nutrient composition of lamb and sheep mortality compost, poultry litter and poultry mortality compost.

catastrophic events such as poultry flock losses due to heat stress or herbicide contamination (Malone et al., 2004). However, it is now increasingly being used to successfully manage the disposal of carcasses in EAD outbreak, particularly in North America.

5.1 Mass poultry mortality composting[1]

Composting is particularly suitable for the emergency management of broiler-farm mortalities and poultry litter. Composting can be conducted both inside and outside the poultry house following euthanasia. Additional litter, sawdust or other carbon source can be delivered to the farm when the volume of litter in the poultry house is insufficient to complete the composting process. As a general rule, 4 to 5 mm of litter is required per kg of carcass per m^2 of poultry-house floor space (Tablante & Malone, 2005).

Poultry carcasses can be layered in windrows using essentially the same procedure as described above for the routine management of mortalities. A skid-steer loader is used to layer carcasses in a windrow with dimensions of 3-4 m at the base and up to 1.8 m high. Each layer of mortality should be no deeper than 25 cm with 15 to 20 cm of litter/sawdust between each layer. The final windrow is capped with 15 to 20 cm of litter/sawdust and to ensure that all carcasses are covered. Each layer of birds is moistened with water at a rate of 1 litre/kg of carcass (Tablante et al., 2002).

Alternatively, birds can be mixed and piled up together with the available carbon source. Firstly, the birds are spread evenly across the centre of the shed. The carcasses are rolled up together with litter to form windrows 3-4 m wide at the base. The litter from along the sidewalls (or additional supply of carbon, if needed) is then used to cap the windrows (15 to 20 cm thickness). Experience in the United States has shown that this method involves the least time, labour and materials. In addition, current research in Australia has confirmed anecdotal evidence that windrows constructed in this manner result in faster carcass

[1] This section has largely been adapted from Wilkinson (2007).

decomposition and higher temperatures than windrows constructed using the layering method (Wilkinson et al., 2010; Fig. 7).

Where larger birds such as turkeys are involved, or where there is a desire to speed-up decomposition, carcasses can be shredded by rotary tiller or crushed by loader prior to constructing the windrows. Bendfeldt et al. (2005b) demonstrated that temperatures above 60°C were achieved within 5 days in windrows constructed with crushed or shredded turkeys and 16 days for whole carcasses. In addition, they reported that to compost crushed or shredded carcasses, 30% less carbon material was required compared to whole carcasses. Windrows formed from crushed or shredded carcasses also do not require additional water to be added.

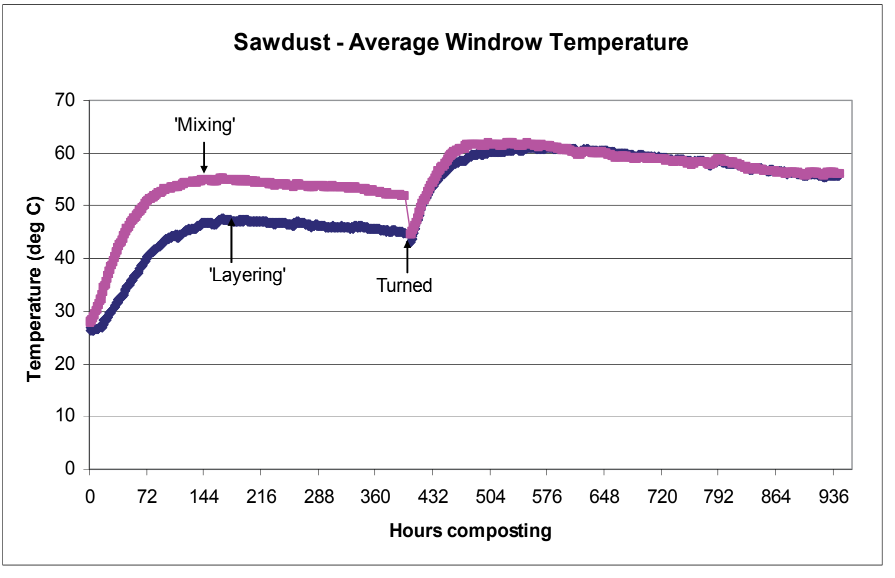

Fig. 7. Average temperatures in poultry mortality composting windrows constructed using the layering and mixing method and sawdust as the carbon source (Wilkinson et al., 2010).

Temperatures in excess of 55°C are usually reached within 5 days of windrow construction. When temperatures begin to decline after 10 to 14 days, the windrows can either be turned inside the poultry house, or reformed outside. If windrows are moved outside, they are covered, for example with tarpaulin. Following turning, windrows are capped again with litter or other carbon source to a minimum depth of 10 cm. After an additional 2 to 3 weeks the compost can be applied to land with the approval of the relevant authorities.

5.1.1 Biosecurity of mass poultry mortality composting

The biosecurity of mass poultry mortality composting has been reviewed recently by Wilkinson (2007) and Berge et al. (2009). Although composting is a well-established pathogen reduction technology, process management and heterogenous pile conditions pose particular challenges for validating the microbiological safety of mortality composting. Biosecurity agencies in Australia, New Zealand, United States and Canada have recognised the potential benefits of using composting for both routine and emergency management of mortalities, and have identified it as a preferred method of carcass disposal (Department of

Agriculture, Fisheries & Forestry, 2005). However, the lack of a scientifically validated process is likely to be a major barrier to its widespread adoption in many countries (Wilkinson, 2007). Research projects are currently underway in the United States, Canada and Australia to bring scientific validation to a process that has been successfully used in a number of EAD outbreaks in North America (e.g. see Bendfeldt et al., 2005a,b; Malone et al., 2004; Spencer, 2005a,b). A growing body of studies published to date (e.g. Senne et al., 1994; Wilkinson et al., 2010; Xu et al., 2009; Xu et al., 2010) confirms that the process is a feasible and biosecure alternative to landfilling of EAD-affected poultry carcasses.

6. Conclusions

On-farm mortality composting is likely to play an increasing role in carcass disposal due to a general contraction in the availability of rendering services and tightening regulations governing on-farm burial. It is a relatively simple and effective process and, if done properly, it meets the biosecurity, environmental, and public health objectives of safe carcass disposal. It can be used successfully for the routine management of farm animal mortalities of all sizes. Mortality composting is particularly suited also to the broiler industry for management of mass mortalities in the event of an emergency disease outbreak.

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15

Recycling of Printed Circuit Boards

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1. Introduction

Printed circuit boards (PCBs) can be found in any piece of electrical or electronic equipment: nearly all electronic items, including calculators and remote control units, contain large circuit boards; an increasing number of white goods, as washing machines contains circuit boards for example in electronic timers. PCBs contain metals, polymers, ceramics and are manufactured by sophisticated technologies.
Wastes from electric and electronic equipments (WEEE) show an increasing upward tendency: a recent annual estimation for WEEE was almost 6.5 million tonnes, and it has been predicted that by 2015 the figure could be as high as 12 million tonnes (Barba-Gutiérrez et al., 2008). A significant proportion of WEEE is constituted by PCBs which represent about 8% by weight of WEEE collected from small appliances (Waste & Resources Action Programme Project, WRAP 2009) and 3% of the mass of global WEEE (Dalrymple et al., 2007).
However there is an increasing interest in the end-of-life management of polymers present in WEEE mainly due to high quotas of recycling and recovery set by legislation which can only be fulfilled by including the plastic fraction in recycling and recovery approaches. Furthermore, disposal of PCB in landfill is no longer accepted in developed countries because of environmental impact and loss of resources. So far recycling of waste PCBs is an important subject in terms of potential recovering of valuable products but several difficulties still exist due to environmental problems involved in end-of-life WEEE management. Due to its complex composition, PCBs recycling requires a multidisciplinary approach intended to valorise fibres, metals and plastic fractions and reduce environmental pollution, which are here reviewed in an attempt to offer a an overview of the latest results on recycling waste PCBs.

2. PCB composition

PCBs are platforms on which integrated circuits and other electronic devices and connections are installed. Typically PCBs contain 40% of metals, 30% of organics and 30% ceramics. Bare PCB platforms represent about 23% of the weight of whole PCBs (Duan et al., 2011). However there is a great variance in composition of PCB wastes coming from different appliances, from different manufacturers and of different age. As an example, after removing hazardous batteries and capacitors which, according to current legislation, must follow a separate recycling, the organic fraction resulted about 70% in PCBs from computers and TV set and 20% in those from mobile phones (William & Williams, 2007).

PCBs contain large amount of copper, solder and nickel along with iron and precious metals: approximately 90% of the intrinsic value of most scrap boards is in the gold and palladium content. However the board laminate mainly consists of a glass fibre reinforced thermosetting matrix which actual legislation imposes to be also conveniently recycled or recovered.

2.1 Polymer matrix and reinforcement

Platforms are usually thermoset composites, mainly epoxies, containing high amount of glass reinforcement; in multilayer boards multifunctional epoxies or cyanate resins are used; in TV and home electronics PCBs are often made with paper laminated phenolic resins. Biobased composites have been recently proposed as possible substitute of traditional resins used in PCBs (Zhan & Wool, 2010).
Due to the risk of ignition during soldering of the components on the platform or impact with electric current, the matrix is often a bromine-containing, fire retarded matrix likely to contain 15% of Br. Fire retardance can be attained either using additive or reactive fire retardants. The two primary families of brominated flame retardants are the polybrominated diphenyl ethers (PBDPE) and fire retardants based on tetrabromo-bisphenol A (TBBA). Despite PBDPE have now been restricted in electrical and electronic equipment they have been found above detection limits in some PCB wastes collected in 2006 in UK; as these results relate to equipment manufactured at least 15 years ago, these levels can be considered to be likely maximum levels. Future waste PCBs are expected to contain significantly lower amount (Department for Environment, Food and Rural Affairs [DEFRA], 2006).
One of the main reasons for the current concerns regarding the use of BFR is that nearly all of them generate polybrominated dibenzo-dioxins (PBBD) and polybrominated dibenzo-furans (PBDF) during the end of life processes involving even a moderate heating. Environmental impact of BFR has been considered (Heart, 2008; Schlummeret al., 2007) and several ecofriendly strategies of fire retardancy have been investigated particularly in Europe, United States and Japan, including incorporation of metal oxides, phosphorous (Pecht &. Deng 2006) and phosphorous-nitrogen compounds (El Gouri et al., 2009). However, these approaches still suffer for drawbacks and the market has not selected a standard replacement for bromine-based flame retardants yet. On the other hand in 2008, European Commission's Scientific Committee on Health and Environmental Risks concluded no risk for TBBA when used as a reactive fire retardant and does not foresee restrictions onTBBA marketing and use. (Kemmlein et al., 2009)
The majority of reinforcements in PCBs are woven glass fibres embedded in the thermoset matrix. However because of the crushing stage preliminary to most recycling technologies, they can be recovered as shorter fibres still possessing high length/density ratio, high elastic modulus and low elongation for being used in thermoplastic polymers.

2.2 Metals

Precious metals in electronic appliances serve as contact materials due to their high chemical stability and their good conducting properties. Platinum group metals are used among other things in switching contacts or as sensors. The typical Pb/Sn solder content in PCB scraps ranges between 4-6% of the weight of the original board. Copper-beryllium alloys are used in electronic connectors where a capability for repeated connection and disconnection is desired and such connectors are often gold plated. A second use of

beryllium in the electronics industry is as beryllium oxide which transmits heat very efficiently and is used in heat sinks.
Typically PCBs contain about 5%weight of Fe, 27% of Cu, 2% of Al and 0.5% of Ni, 2000 ppm of Ag 80 ppm of Au; however there is no average scrap composition and the values given as typical averages actually only represent scraps of a certain age and manufacturer. Additionally, non-ferrous metals and precious metals contents have gradually decreased in concentration in scraps due to the falling power consumption of modern switching circuits: in the ‘80s the contact layer was 1–2.5 μm thick, in modern appliances it is between 300 and 600 nm (Cui & Zhang, 2008).

3. WEEE legislation

Concern about environment prompts many governments to issue specific legislation on WEEE recycling: however with the notable exception of Europe, many countries seem to be slow in initiating and adopting WEEE regulations. In Europe the WEEE Directive (European Union 2003b) and its amendments as a first priority aims to prevent the generation of WEEE. Additionally, it aims to promote reuse, recycling and other forms of recovery of WEEE so as to reduce the disposal of wastes. In both developed and developing nations, the landfilling of WEEE is still a concern and accumulation of unwanted electrical and electronic products is still common. Handling of WEEE in developing countries show high rate of repair and reuse within a largely informal recycling sector (Ongondo et al., 2011).
The WEEE Directive requires the removal of PCBs of mobile phones generally, and of other devices if the surface of the PCBs is greater than 10 cm^2: To be properly recovered and handled waste PCBs have to be removed from the waste stream and separately recycled. Batteries and condensers also have to be removed from WEEE waste stream.
The RoHS Directive (European Union 2003a) names six substances of immediate concern: lead, mercury, cadmium, hexavalent chromium, polybrominated diphenyl ethers (Penta-BDE and Octa-BDE) and polybrominated biphenyls. The maximum concentration values for RoHS substances were established in an amendment to the Directive on 18 August 2005. The maximum tolerated value in homogenous materials for lead, mercury, hexavalent chromium, polybrominated diphenyl ethers and polybrominated biphenyls is 0.1% w/w and for cadmium 0.01% w/w.

4. Disassembling WEEE and PCBs

Nearly all of the current recycling technologies available for WEEE recycling include a sorting/disassembly stage. The reuse of components has first priority, dismantling the hazardous components is essential as well as it is also common to dismantle highly valuable components, PCBs, cables and engineering plastics plastics in order to simplify the subsequent recovery of materials. Moreover cell batteries and capacitors should be manually removed and separately disposed in an appropriate way. The PCBs can then be sent to a facility for further dismantling for reuse or reclamation of electric components.
Most of the recycle plants utilize manual dismantling. The most attractive research on disassembly process is the use of an image-processing and database to recognize reusable parts or toxic components. The automated disassembly of electronic equipment is well advanced but unfortunately its application in recycling of electronic equipment still face lot of frustration. In treatment facilities components containing hazardous substances are only

partly removed particularly in small WEEE. This implies that substantial quantities of hazardous substances are forwarded to subsequent mechanical crushing processes, causing significant dispersion of pollutants and possibly reduction of quantities of valuable recyclable materials (Salhofer & Tesar, 2011).
Electronic components have to be dismantled from PCB assembly as the most important step in their recycling chain, to help conservation of resources, reuse of components and elimination of hazardous materials from the environment. In semi-automatic approaches, electronic components are removed by a combination of heating and application of impact, shearing, vibration forces to open-soldered connections and heating temperature of 40-50 °C higher than the melting point of the solder is necessary for effective dismantling; pyrolysis probably occurs during the dismantling, which means there is a potential for dioxin formation when this scrap is heating (Duan et al., 2011).

5. Physical recycling

Thermosetting resins, glass fibres or cellulose paper, ceramics and residual metals can serve as good filler for different resin matrix composites. Physical recycling always involves a preliminary step were size reduction of the waste is performed followed by a step in which metallic and non-metallic fractions are separated and collected for further management.

5.1 Size reduction and separation

A crushing stage is necessary for an easier further easier management of PCB waste. The PCB are cut into pieces of approximately 1 -2 cm^2 usually with shredders or granulators giving the starting batch easily manageable for supplementary treatments (PCB scraps). Further particle size reduction to 5-10 mm can be carried out by means of cutting mills, centrifugal mills or rotating sample dividers equipped with a bottom sieve. The local temperature of PCB rapidly increases due to impacting and reaches over 250°C during crushing, so a pyrolytic cleavage of chemical bonds in the matrix produces brominated and not brominated phenols and aromatic/aliphatic ethers (Li et at., 2010)
Effective separation of these materials based on the differences on their physical characteristics is the key for developing a mechanical recycling system; size and shape of particles play crucial roles in mechanical recycling processes because the metal distribution is a function of size range: aluminum is mainly distributed in the coarse fractions (> 6.7 mm), but other metals are mainly distributed in the fine fractions (< 5 mm).
Almost all the mechanical recycling processes have a certain effective size range and mechanical separation processes is performed in a variety of technique. Shape separation by tilted plate and sieves is the most basic method that has been used in recycling industry. Magnetic separators, low-intensity drum separators are widely used for the recovery of ferromagnetic metals from non-ferrous metals and other non-magnetic wastes. The use of high-intensity separators makes it possible to separate copper alloys from the waste matrix. Electric conductivity-based separation such as Eddy current separation, corona electrostatic separation and triboelectric separation separates materials of different electric conductivity such non ferrous metals from inert materials. (Veit at al., 2005; Cui & Forssberg, 2003). Density-base separation of particles such as sink-float separation, jigging, upstream separation are also used to separate metal from non metal fractions in PCB scraps.

5.2 Applications in composites

Physical recycling for non metallic fraction sorting from separation stage has been recently reviewed by Guo (Guo et al. 2009). The thermal stability of the non metallic fraction of PCBs is very important for physical recycling methods which must be suitable for moulding processes. The thermosetting matrix more suitable for making composites with PCB scraps are phenolic resins, unsaturated polyester resins and epoxy resins. To ensure the surface smoothness, the size of non metallic fractions used was less than 0.15mm. The non metallic items so produced are used for trays, sewer grates, kitchen utensils, electronic switches etc. with properties comparable to that of composites with traditional filler. The 300-700 °C pyrolysis residues (75–80%) can be easily liberated for metal's recovery, and the glass-fibres can be re-compounded into new SMC and BMC structures as a filler replacement (Jie et al., 2008).

Nonmetals reclaimed from waste PCBs are used to replace wood flour in the production of wood plastic (polyethylene) composites (Guo et al., 2010). In analogy, addition of PCB non-metallic fraction as reinforcing fillers in polypropylene (PP) has proven to be an effective way to enhance strength and rigidity: particles 0.178-0.104 mm, modified by a silane coupling agent, could be successfully added in PP composites as a substitute of traditional fillers. Larger particles (> 0.178mm) are fibre-particulate bundles showing weakly bonded interface which make easier crazes initiation and particle detach from the polymer matrix. (Zheng et al., 2009a).

As one of the plastic wastes to a certain extent, the non-metallic fraction of PCB can also be used with some effectiveness as a partial replacement of inorganic aggregates in concrete applications to decrease the dead weight of structures. Lightweight concrete is extensively used for the construction of interior and exterior walls of buildings for the case where the walls are not designed for lateral loads (Niu & Li, 2007). The glass fibres and resins powder contained in the non-metallic fraction can also be used to strengthen the asphalt.

6. Chemical recycling

Chemical recycling refers to decomposition of the waste polymers into their monomers or some useful chemicals by means of chemical reactions. In this view, chemical recycling consists of pyrolysis process, depolymerization process by using supercritical fluids, hydrogenolytic degradation and gasification process. The refining of the products (gases and oils) is included in the chemical recycling process, and can be done with conventional refining methods in chemical plants. Metal fraction can be treated by pyrometallurgical and hydrometallurgical approaches, biotechnological processes being still in their infancy.

6.1 Pyrolysis

Pyrolysis of polymers leads to the formation of gases, oils, and chars which can be used as chemical feedstocks or fuels. Pyrolysis degrades the organic part of the PBC wastes, making the process of separating the organic, metallic and glass fibre fractions of PCBs much easier and recycling of each fraction more viable. Additionally, if the temperature is high enough, the pyrolysis process will melt the solder used to attach the electrical components to the PCBs. The combination of the removal and recovery of the organic fraction of PCBs and the removal of the solder aid the separation of the metal components.

The thermal behaviour of epoxy resins, the most common polymer matrix in PCB, has been widely investigated as a basis for pyrolytic recycling. In thermogravimetry brominated epoxy resins are less thermally stable than the corresponding unbrominated ones. They

exhibit a steep weight loss stage at 300-380°C depending on the hardener, those hardened by aromatic amines and anhydrides decomposing at higher temperature (Fig. 1).
Mostly brominated and unbrominated phenols and bisphenols are found in the pyrolysis oil however the balance phenols/bisphenols and brominated/unbrominated species depends on the temperature and residence time in the reactor; higher temperatures and longer times making debromination more extensive (Luda et al., 2007, 2010). The size of the PCB particles effects as well on the decomposition temperature: degradation is postponed when particles are larger than 1 cm^2 due to heat transfer limitation (Quan et al., 2009).

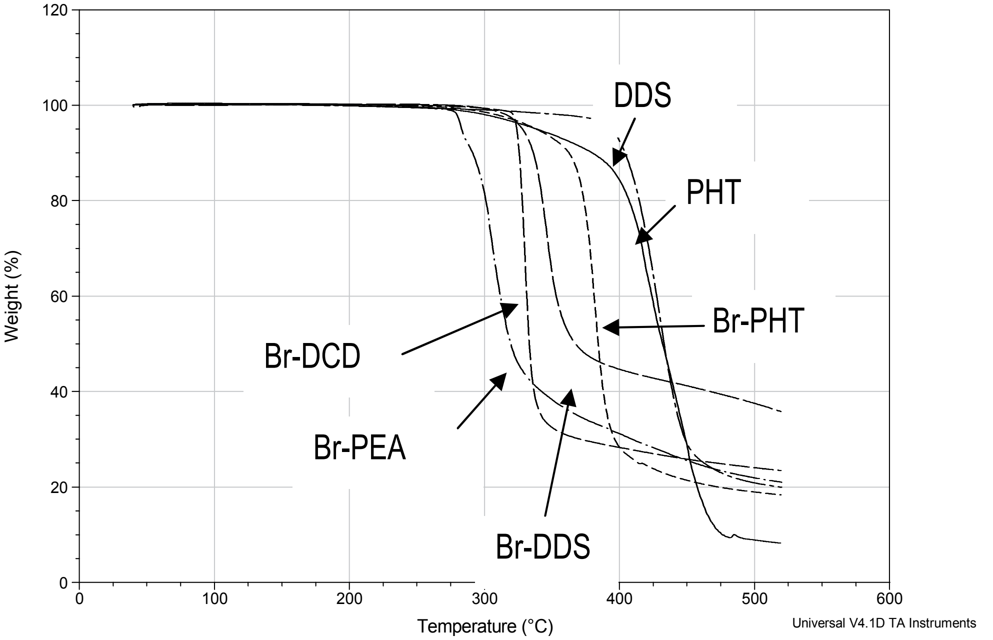

Fig. 1. TGA of epoxy-system based on Diglycidyl ether of bisphenol A or Diglycidyl ether of tetrabromobisphenol A (Br) crosslinked using different hardeners; DDS: Diaminodiphenylsulphone; DCD: Dicyandiamide; PEA: Polyethylene-polyamine, PHT: Phthalic anhydride. (20°C/min, Nitrogen)

When PCBs (4 cm^2) were pyrolyzed in a tubular type oven in the range 300 - 700 °C, no significant influence of temperature was observed over 500 °C both in gases and oil yields (9 and 78% respectively) as well as in the gross calorific value (30kJ/kg). However the oil resulted contaminated by polluting element and must be purged for further utilization. (Guan et al., 2008). The boards pyrolysed in a fixed bed reactor at 850°C were very friable and the different fractions could be easily separated (Hall & Williams, 2007) .

6.1.1 Vacuum pyrolysis

Recently studies on application of vacuum pyrolysis to PCBs have appeared in the literature. They were mostly aimed to recover solder and facilitate separation of metals and glass fibres from PCB scraps. Vacuum pyrolysis shorts organic vapour residence time in the reactor and lowers decomposition temperature, reducing the occurrence and intensity of secondary reactions.

The residue of vacuum pyrolysis at 550 °C of bare PCB scraps (25 cm^2) was crushed and size classified; about 99% of original copper was confined in particles > 0.4 mm, fibres remained in the smaller particles were recovered after calcinations. Pyrolysis oil and gases were collected from pyrolysis reactor for further refining (Long et al., 2010).

Two different arrangement for recycling disassembled PCBs (10-15 cm^2) were proposed: in the first centrifugal separation of solder (240°C) was followed by vacuum pyrolysis of the residue (600 °C); in the second vacuum pyrolysis (600 °C) was followed by centrifugal separation of the residue at 400°C in order to collect solder ready for reuse (Zhou & Quj, 2010; Zhou et al., 2010)

6.1.2 Dehalogenation

Contamination of oil by harmful compounds remains a severe issue with a strong impact on material and thermal recycling: bromine-containing phenols are potentially hazardous compounds emitted during heating of polymers flame retarded with TBBA based fire retardants. In effect brominates phenols likely form PBDD/PBDF through Ullmann condensation, contaminating pyrolysis products. So that reduction of the amount of brominated phenols in the pyrolysis oil in favour of less toxic substances is a way to add value to the whole PCB recycling process. Dehalogenation attempts have been carried out on model compounds, directly in the pyrolysis of PCB scraps or on refining the pyrolysis oil. Successful approach to debrominate PCB scraps was carried out by pyrolysis in the presence of NaOH or sodium-containing silicates resulting in an enhanced bromomethane evolution and depression of brominated phenol formation (Blazso et al., 2002). Various combination of cracking catalysts and absorbers for halogenated compounds ($CaCO_3$ and red mull) decreased as well the amount of all heteroatoms in pyrolysis oils of PCBs: after pyrolysis at the 300-540 °C the oils were passed into a secondary catalytic reactor (Vasile et al., 2008).

PBDD/PBDF formed during pyrolysis at 850 to 1200 °C of PCBs were destroyed under controlled combustion conditions (1200 °C): the total content decreased by approximately 50% increasing the pyrolysis temperature from 850 to 1200 °C. If CaO is added in the feeding, inhibition of 90% PBDD/PBDF occurs with prevention of evolution of HCl and HBr that corrode the equipment (Lai et al., 2007).

Liquid products obtained from pyrolysis of general WEEE, PCBs and their mixtures were upgraded by thermal and catalytic hydrogenation. The effect of thermal hydrogenation was improved by using catalysts such as commercial hydrogenation DHC-8 and metal loaded activated carbon. The upgraded degradation products were separated in residue, liquids and gases; liquids with high amount of aromatics were obtained but most of hazardous toxic compounds were eliminated after hydrogenation by converting them into gaseous HBr (Vasile et al., 2007).

Hydrodehalogenation with hydrogen-donating media is a promising option for the destruction of halogen-containing aromatics in the pyrolysis oil, converting them into non-halogenated aromatics and valuable hydrogen halide. It was found that PP was an effective and selective hydrodehalogenatig agent because only HBr was recovered at 290–350 °C from a mixture of chlorinated and brominated phenols PP was effective as well in upgrading pyrolysis oil (Hornung et al. 2003, Balabanovich et al., 2005). Recently other polymers have been tested for dehalogenation of a model brominated phenol. From pyrolysis of equimolecular mixture of various polymers with 2,4- dibromophenol (DBP) bromine was recovered as valuable HBr in gases, toxic brominated compounds in oil or confined in the charred residue.

Pyrolysis conditions		% of total Br in the pyrolysis fractions			
components	T (°C)	gases	oil	Residue	H_2O sol.
DBP	330	5	88	7	0
DBP+HDPE	330	77	0	23	0
DBP+LDPE	330	85	0	15	0
DBP+PBD	330	73	1	26	0
DBP + PS	330	49	51	0	0
DBP+PA-6	350	45	20	0	35
DBP+PA-6,6	350	59	4	12	26
DBP+PAN	330	35	23	15	27

Table 1. Percentage of the bromine resulting in the various fractions from pyrolysis of 2,4-dibromophenol (DBP) with low density polyethylenes (LDPE), High density polyethylene (HDPE), polystyrene (PS), polybutadiene (PBD) Polyamides (PA-6, PA-6,6), polyacrylonitrile (PAN).

LDPE was found nearly as effective as PP; PBD and HDPE were slightly less effective while activity of PS, polyamides and PAN was poor. Br was partially recovered in the water soluble fraction when polymers contained nitrogen (Tab. 1) (Luda & Balabanovich, 2011). Because these polymers are present in significant amount in the organic fraction of WEEE, or even in other solid wastes, their action can be considered as a viable and convenient route of recycling of PCBs.

6.1.3 Depolimerization in supercritical fluids

Supercritical methanol and water have been tested for depolymerization of thermoset resins in PCBs for recycling purposes: the lower critical temperature and pressure of methanol (Tc : 240 °C, Pc : 8.09 MPa) compared to those of water (Tc : 374 °C, Pc: 22.1 MPa) allow milder conditions.

At 350°C the oils of comminuted PCB (<1mm) treated with supercritical methanol included phenol with 58% purity, much higher than that produced by other conventional pyrolysis processes. The oils did not contain brominated compounds due to the complete decomposition and debromination during the process. Large amount of HBr existed in the gaseous products, which could be recovered effectively by simple distillation. Metallic elements in waste PCBs were concentrated effectively up to 62% in the solid residue. Longer reaction time and lower temperature was favorable for obtaining a higher oil yield (Xiu & Zhang, 2010).

6.2 Gasification and co-combustion

Gasification converts organic materials into carbon monoxide and hydrogen (syngas) by reacting the raw material at high temperatures with a controlled amount of oxygen and/or steam: syngas is itself a fuel or can be used as intermediates for producing chemicals or even combusted in gas turbines for electric power production. Staged-gasification of WEEE and PCB comprises pyrolysis (550°C) and high temperature gasification (>1230°C). Combustion or co-combustion competes with gasification producing electric power as well. A certain amount of bromine contained in the waste turns into ashes (co-combustion) or char

(gasification), while most turns into combustion gases or into syngas where: bromine can be recovered using suitable wet scrubbing systems.
A comparative environmental analysis of these two competing scenarios, intended for bromine recovery and electric power production, was carried out on recycling of the same mixed feeding PCB/green waste. While both processes resulted eco-efficient, staged-gasification was more efficient from an energy point of view, had a potentially smaller environmental impact than co-combustion and allowed a more efficient collection of bromine (Bientinesi & Petarca, 2009).

6.3 PCB recycling of the metal fraction

Despite the fluctuant average scrap composition amongst the various WEEE, cell phones, calculators and PCB scraps reveal that more than 70% of their value depends on their high content in metals. Metallurgical recovery of metals from WEEE is therefore a matter of relevance and has been recently reviewed by Cui (Cui & Zhang, 2008) underlining three possible approaches: pyrometallurgy, hydrometallurgy and biotechnology.

6.3.1 Pyrometallurgy

Some techniques used in mineral processing could provide alternatives for recovery of metals from electronic waste. Traditional, pyrometallurgical technology has been used for recovery of precious metals from WEEE to upgrade mechanical separation which cannot efficiently recover precious metals. In the processing the crushed scraps are burned in a furnace or in a molten bath to remove plastics, and the refractory oxides form a slag phase together with some metal oxides. Further, recovered materials are retreated or purified by using chemical processing. Energy cost is reduced by combustion of plastics and other flammable materials in the feeding. It should be stated, however, that applying results from the field of mineral processing to the treatment of electronic waste has limitations because the size of particles involved and material contents are quite different in the two systems.
Despite differences in the plants, general electronic scraps are treated together with other metal scraps by pyrometallurgical processes in the Noranda process at Quebec, Canada, at the Boliden Ltd. Rönnskår Smelter, Sweden (Association of Plastics Manufacturers in Europe [APME], 2000), at Umicore at Hoboken, Belgium (Hageluken, 2006). The used electronics recycled in the smelters represent 10-14% of total throughput, the balance being mostly mined copper concentrates at Noranda, lead concentrates at Boliden, various industrial wastes and by-products from other non-ferrous industries at Umicore.
Recently a modified pyrometallurgy to recover metals from PCBs has been proposed (Zhou et al., 2010) showing that addition of 12 wt.% NaOH as slag-formation material promotes the effective separation of metals from slag; the remaining slag in the blowing step was found to favour the separation of Cu from other metals and allow noble metals to enter the metal phase to the greatest extent. Additionally, the resulting slag was shown to be very effective in cleaning the pyrolysis gas. Eventually 68.4% Cu, 92.6% Ag and 85.5% Au recovery could be achieved in this process, confirming preliminarily the feasibility of modified pyrometallurgy in recovering metals from PCB.
However, pyrometallurgical processing of electronic waste suffers from some limits in particular the recover as metals of aluminum and iron transferred into the slag is difficult , the presence of brominated flame retardants in the smelter feed can lead to the formation of dioxins unless special installations and measures are present and precious metals are obtained at the very end of the process. Furthermore pyrometallurgy results in a limited

upgrading of the metal value and hydrometallurgical techniques and/or electrochemical processing are subsequently necessary to make refining.

6.3.2 Hydrometallurgy

Leaching is the process of extracting a soluble constituent from a solid by means of a solvent: for electronic wastes leaching involve acid and/or halide treatment due to the fact that acid leaching is a feasible approach for removing of base metals so as to free the surface of precious metals. The solutions are then subjected to separation and purification procedures such as precipitation of impurities, solvent extraction, adsorption and ion-exchange to isolate and concentrate the metals of interest. Consequently, the solutions are treated by electrorefining process, chemical reduction, or crystallization for metal recover.

A bench-scale extraction study was carried out on the applicability of hydrometallurgical processing routes to recover precious metals from PCBs in mobile phones (Quinet et al, 2005). An oxidative sulfuric acid leach dissolves copper and part of the silver; an oxidative chloride leach dissolves palladium and copper; and cyanidation recovers the gold, silver, palladium and a small amount of the copper. To recover the metals from each leaching solution, precipitation with NaCl was preferred to recuperate silver from the sulfate medium; palladium was extracted from the chloride solution by cementation on aluminum; and gold, silver and palladium were recovered from the cyanide solution by adsorption on activated carbon. The optimized flowsheet permitted the recovery of 93% of the silver, 95% of the gold and 99% of the palladium.

Recovery of Cu, Pb and Sn from PCB scraps equipment has been performed by a mechanical processing which concentrate metals. At the second stage, the concentrated fraction was dissolved with acids and treated in an electrochemical process in order to recover the metals separately (Veit et al. , 2006).

Recently a general approach for recycling of scrapped PBC by hydrometallurgy has been proposed. First the crushed PCB scraps were leached in the NH_3/NH_5CO_3 solution to dissolve copper. After the solution was distilled and the copper carbonate residue was converted to copper oxide by heating. The remaining solid residue after copper removal was then leached with hydrochloric acid to remove tin and lead. The last residue was used as a filler in PVC plastics which were found to have the same tensile strength as unfilled plastics, but had higher elastic modulus, higher abrasion resistance and were cheaper (Liu et al., 2009)

6.3.3 Biometallurgy

Biotechnology is one of the most promising technologies in metallurgical processing. Microbes have the ability to bind metal ions present in the external environment at the cell surface or to transport them into the cell for various intracellular functions. This interaction could promotes selective or non-selective in recovery of metals. Bioleaching and biosorption are the two main areas of biometallurgy for recovery of metals.

Bioleaching has been successfully applied for recovery of precious metals and copper from ores for many years. Despite, limited researches were carried out on the bioleaching of metals from electronic wastes but it has been demonstrated that using *C. violaceum*, gold can be microbially solubilized from PCB (Faramarzi et al., 2004) and using bacterial consortium enriched from natural acid mine drainage, copper could be efficiently solubilised from waste PCBs in about 5 days (Xiang et al., 2010). The extraction of copper was mainly accomplished indirectly through oxidation by ferric ions generated from ferrous ion

oxidation bacteria; a two-step process was necessary for bacterial growth and for obtaining an appropriate oxidation rate of ferrous ion.
Biosorption process is a passive physico-chemical interaction between the charged surface groups of micro-organisms and ions in solution. Biosorbents are prepared from the naturally abundant and/or waste biomass of algae, fungi or bacteria. Physico-chemical mechanisms such as ion-exchange, complexation, coordination and chelation between metal ions and ligands, depend on the specific properties of the biomass (alive, or dead, or as a derived product). Compared with the conventional methods, biosorption-based process offers a number of advantages including low operating costs, minimization of the volume of chemical/biological sludges to be handled and high efficiency in detoxifying. However further efforts are required because the adsorption capacities of precious metals on different types of biomass is greatly variable and much more work should be done to select a perfect biomass from the billions of microorganisms and their derivatives. Most of the researches on biosorption mainly focused on gold, more attentions should be taken into biosorption of silver from solutions and on recovery of precious metals from multi-elemental solutions.

7. Conclusion

A successful recycling approach of PCB should take into consideration the valorisation of the recycled items to compensate for recycling costs. Recycling of WEEE, and of PCB in particular, is still a challenging task due to complexity of these materials and possible evolution of toxic substances. Traditionally, recovering of valuable metals by waste PCBs was carried out on a large scale for a positive economic revenue. Legislation pushes now toward a more comprehensive processes which includes recovering and recycling of the ceramic and organic fractions in substitution to not-ecoefficient disposal in landfill.
A disassembly stage is always required to remove dangerous components such as batteries and condensers. Manual dismantling is still in operation despite the attempts to proceed by automatic procedures which however need more progress to be really effective. Crushing and separation are then key points for improving successful further treatments.
Physical recycling is a promising recycling method without environmental pollution and with reasonable equipment invests, low energy cost and diversified potential applications of products. However separation between the metallic and non metallic fraction from waste PCBs has to be enhanced.
Pyrolytic approach is attractive because it allows recovering of valuable products in gases, oils and residue. Evolution of toxics PBBD/PBDF can be controlled by appropriate treatments such as addition of suitable scavengers or dehydrohalogenation, which are still under development. New technologies are proposed such as vacuum pyrolysis or depolymerisation in supercritical methanol.
Metal recovery can be performed by traditional pyrometallurgical approaches on metal-concentrated PCB scraps fractions. Comparing with the pyrometallurgical processing, hydrometallurgical method is more exact, more predictable, and more easily controlled. New promising biological processes are now under development.
It should be kept in mind however that the chemical composition of e-waste changes with the development of new technologies and pressure from environmental organisations to find alternatives to environmentally damaging materials. A sound methodology must take in account the emerging technologies and new technical developments in electronics. Miniaturisation of electronic equipment in principle would reduces waste volume of PCBs

but make collection more difficult and repair more costly, so that a large amount of PCBs Is still expected in the e- waste in the future.

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Part 3

Industrial Solid Waste

16

Recycling of Waste Paper Sludge in Cements: Characterization and Behavior of New Eco-Efficient Matrices

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1. Introduction

The pulp and paper industry, in Europe, generates 11 million tons of solid waste each year (Monte et al., 2009). Paper waste covers a diverse range of non-hazardous waste streams, prominent among which are different types of sludge, boiler ash, combustion furnace ash and organic and inorganic rejects. Manufacturing processes to produce new paper from the deinking of recycled paper account for 70% of these waste products.

Following its reception, sorting and storage, the recycled paper is transformed into an aqueous suspension of fibers, while inappropriate materials are eliminated in different cleaning processes. Following this initial treatment, the resultant paper sludge is subjected to deinking in a froth flotation process, which produces waste known as de-inked sludge. This waste sludge is fundamentally composed of water, fiber, ink and a mineral load. In addition, various paper manufacturing processes have water treatment plants that generate sludges with high humidity contents.

The deinked paper sludge and the sludge from the water treatment process have a high humidity content (≈ 50%), and are roughly composed of organic material with their origin in paper fibers (≈ 25%) and mineral loads such as calcium carbonate, kaolin, talc and titanium oxide (≈ 25%). A similar composition highlights the wealth of energetic and mineral resources saturating the paper sludge. Thus, the most advanced techniques for the use of paper sludge are intended to take full advantage of the saturated biomass and the recovery of the mineral constituents in the inorganic fraction.

The most common options for the processing of paper industry sludge range from their exploitation for agricultural purposes, composting, or use as a primary material in the manufacture of ceramics and cement (Moo-Young & Zimmie, 1997; Ahmadi & Al-Khaja, 2001; Lima & Dal Molin, 2005; Conesa et al., 2008), to energy recovery in biomass boilers or fluidized bed systems. Thus, the Dutch CDEM process (International Patent, 2006) represents a pioneering recovery system, where the paper sludge is treated at temperatures of around 730ºC, in a fluidized bed combustion system, so as to activate the latent pozzolanic properties of its mineral content. The CDEM process was industrialized after the pioneering work of research groups led by Prof. Pera (Pera & Amrouz, 1998), which demonstrated that controlled calcination of the deinked sludge produces a highly reactive pozzolanic material, within a temperature range of between 700 and 750ºC.

On the basis of the scientific knowledge presented earlier, a team of Spanish researchers led by Dr. Frías, has been conducting in-depth research over the past decade into the scientific, technological and environmental aspects of obtaining active admixtures from the calcination of paper sludge and its behavior in cement and mortar.

2. Waste paper sludge and its activated products

2.1 Nature of the raw waste and activation process

The characteristic composition of this industrial waste is a mixture of organic material (non-recovered cellulose) and inorganic materials (principally, kaolinite and limestone), normally used as loadings in the manufacture of paper.

An example of the chemical and mineralogical composition of this type of waste is presented in Table 1. The characterization of this dry material is provided by the Spanish paper manufacturer Holmen Paper Madrid, S.L, which uses 100% recycled paper as the raw material. X-Ray Fluorescence (XRF) confirms that the principal oxides are CaO, SiO_2 and Al_2O_3, the sum of which exceeds 43% of the total mass. The high Loss on Ignition (LOI) in these waste products, at around 54%, should be underlined, due to the presence of organic material, kaolinite dehydroxylation and the decarbonation process of calcite. These values, for guidance only, vary in accordance with the type of paper, its origin, the percentage of recycled paper used as primary material, the loadings, and the type of process etc. With respect to its mineralogical composition, it is worth highlighting the presence of cellulose residue (about 32%, determined according to the results of XRF and XRD), as well as the presence of calcite and kaolinite content in a ratio of 3.3 (Frías et al., 2010). This value is above those in other research works that report ratios of under 2, even for samples from the same paper manufacturing process (Pera & Amrouz, 1998; Frías et al., 2008a). The variation in the composition of this industrial waste is therefore confirmed.

<table>
<tr><td colspan="11">Chemical composition by XRF (%)</td></tr>
<tr><td>CaO</td><td>SiO_2</td><td>Al_2O_3</td><td>MgO</td><td>Fe_2O_3</td><td>SO_3</td><td>TiO_2</td><td>Na_2O</td><td>K_2O</td><td>P_2O_3</td><td>LOI</td></tr>
<tr><td>25.43</td><td>10.79</td><td>6.82</td><td>0.86</td><td>0.46</td><td>0.33</td><td>0.28</td><td>0.13</td><td>0.24</td><td>0.13</td><td>54.34</td></tr>
<tr><td colspan="11">Mineralogical composition (%)</td></tr>
<tr><td colspan="2" rowspan="2">Organic material</td><td colspan="9">By XRD</td></tr>
<tr><td colspan="2">Calcite</td><td colspan="2">Kaolinite</td><td colspan="5">Phyllosilicates (talc, mica) and quartz</td></tr>
<tr><td colspan="2">32.34</td><td colspan="2">45.27</td><td colspan="2">13.67</td><td colspan="5">8.72</td></tr>
</table>

Table 1. Composition of the raw paper sludge

In the same way as with other clayey materials, this waste has to be subjected to a process of thermal activation to provide it with pozzolanic properties. As it is not a pure natural kaolinite, research into its thermal activation has centered on the range of temperatures between 500 and 800ºC, with retention times in the furnace of between 2 and 5 hours (Frías et al., 2008b; Rodríguez et al., 2009). All of this has the purpose of establishing optimal conditions that will guarantee total elimination of the organic material, an appropriate transformation of the kaolinite into MK, as well as a minimum content of free lime, which relates to aspects of volumetric instability.

2.2 Properties of the activated products

Knowledge of the chemical, physical, mineralogical and pozzolanic properties that determine the behavior of Portland cements prepared with activated wastes in the form of active additions represents a key point for the evaluation of their viability.

2.2.1 Physical properties

Laser diffraction granulometry confirms the presence of particle sizes of less than 90 micrometers. The distribution density curves show 2 maximums located at 40 and 4 micrometers. The BET surface area varies between 7 and 8 m^2/g, for original activated sludge, a much higher value than that found for a cement type I 42.5 R (<1 m^2/g), reaching values of around 12-13 m^2/g for activated paper sludge that is ground down to particle sizes of less than 45 micrometers (Ferreiro, 2010).
The different coloration between the raw paper sludge and the activated product is also worth mentioning (Fig.1). Whereas the former presents a grayish coloring due to the deinking process, the latter shows a white color.

Fig. 1. Appearance of the sludges before (left) and after calcination (right)

Determination of the colorimetric variables (Fig. 2) show values for the coordinate of luminosity (L^*)(or whiteness index) of between 75% and 94%. It may be seen that the luminosity value increases with the conditions of activation (higher temperature and a longer retention time). The increase in luminosity is directly related to different processes that take place in that temperature interval (presence of organic residues, inks, degree of

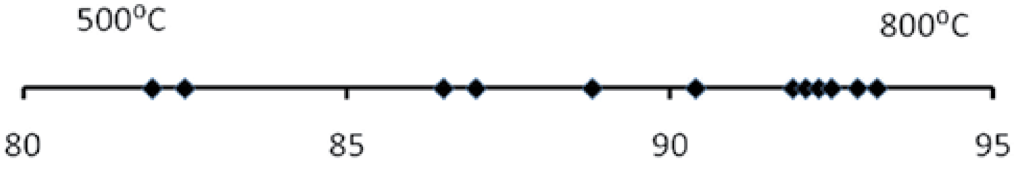

Fig. 2. Luminosity values (%) versus activation conditions

transformation of the kaolinite into metakaolinite and degree of calcite decarbonation). This parameter is of greater importance when using these activated products as pozzolans in the manufacture of commercial cements, especially in white cements, the minimum required value of which is 85% (RC-08).

2.2.2 Chemical composition

In a similar way to the processes described for basic paper sludges, the products yielded by thermal activation are formed principally of silica (20-30%), lime (34-45%), alumina (13-20%) and magnesia (2-3.5%). The remaining oxides are present in amounts of less than 1%. The chemical values increase with the intensity of the activation conditions, as a consequence of the reduction in loss on calcination. These results are in accordance with those obtained by Bai (Bai et al., 2003), but differ from those indicated by Toya (Toya et al. 2006).

2.2.3 Mineralogical and morphological composition

The mineralogical composition of the activated sludge from the most labile (500ºC for 2 hours) to the most drastic (800ºC for 2 hours) conditions reflects the changes undergone in the different minerals due to heating. The paper sludge calcined at 500ºC for 2 hours is composed of talc, kaolinite, illite, dolomite, calcite and quartz. As the temperature increases (550ºC/2 hours), the kaolinite is transformed into metakaolinite. This compound is detected by SEM, as it is not a crystalline material (Fig. 3). The talc and quartz remain unaltered in the range of temperatures under study. In contrast, the dolomite is transformed at 550ºC/2 hours and the calcite disappears at 800ºC/2 hours, as a result of the decarbonation of those minerals. The illite undergoes a transformation process at 800ºC/2 hours. The appearance of portlandite is notable at 650ºC/5 hours or more as a consequence of the exposure of the paper sludge to environmental humidity, while the formation of dicalcium silicate (bredigite) is detected at 800ºC or more.

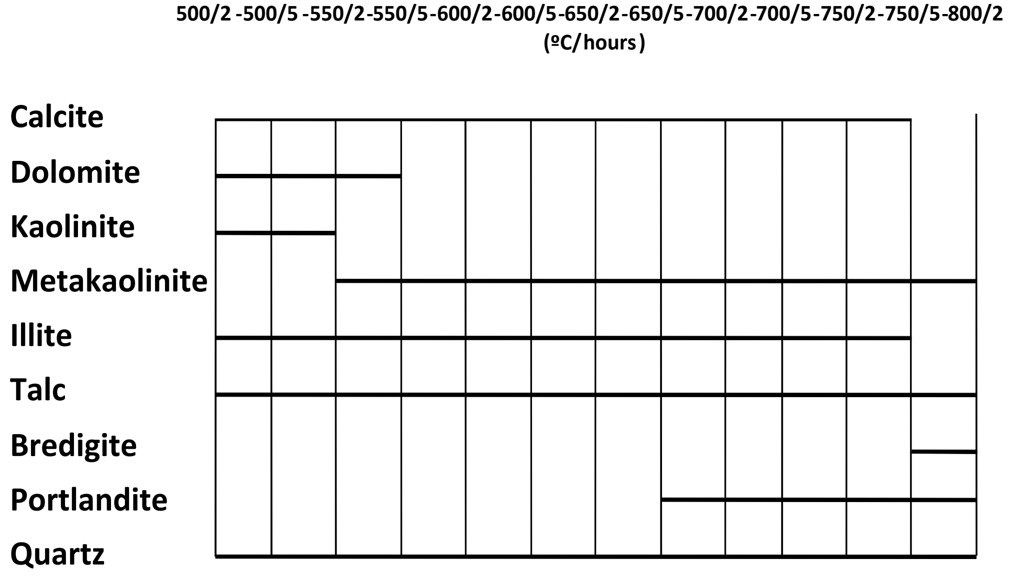

Fig. 3. Stability fields of the different materials identified in the interval 500ºC/2h and 800ºC/2h

Morphologically, the formation of aggregates takes place with the increase in heat through the coldest to the warmest stages, which entails an increase in the specific surface of the materials and means that they become absorbent. Fig. 4 (left) shows the situation of the crystals at 500ºC/2 hours, whereas Fig. 4 right illustrates the great number of aggregates present in the activated paper sludge, corresponding above all to metakaolinite and portlandite, at 800ºC, after 2 hours.

2.2.4 Pozzolanic properties

The fundamental property for a material or an industrial waste product to be used as an active admixture in the manufacture of commercial blended cements is its pozzolanic nature. A rapid method of supplying information in the short term is through the use of an accelerated chemical method in the pozzolan/lime system (Frías et al., 2008c).

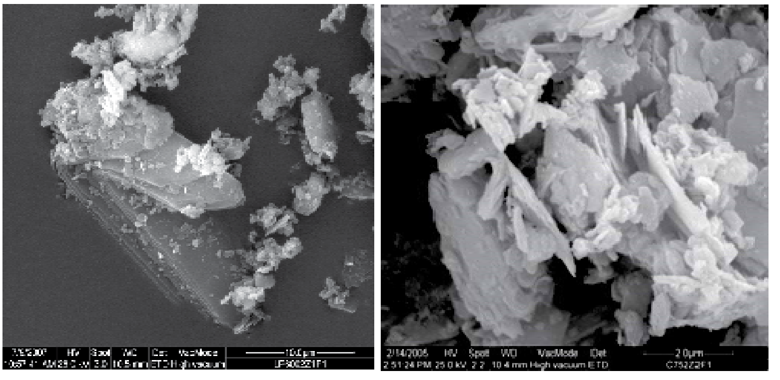

Fig. 4. Left) Large crystals of kaolinite, talc and calcite. Right) Aggregates of metakaolinite and portlandite

The results obtained for the different types of paper sludge, which are activated at temperatures of between 500 and 800ºC after two furnace retention times of 2 and 5 hours (Fig. 5), reveal good pozzolanic activity in all cases. No appreciable differences have been found between periods of 2 and 5 hours. With regard to the activation temperature, it is clearly observed that lime consumption drops at temperatures of 700ºC. This phenomenon may be attributed, on the one hand, to morphological changes in the metakaolinite in the form of more compact aggregates and less specific surface area and; on the other hand, to the initiation of the decarbonation process of the calcite that is present in the waste, liberating free CaO in dissolution, which overlaps with the pozzolanic reaction itself.
A comparative study of these results with pozzolans, normally included in the standards currently in force, shows that the activity of this activated waste is similar to that obtained for pure metakaolin (MK), and very close to silica fume (SF) (Frías et al., 2008d).
As a consequence of the above, together with the mineralogical and morphological observations, it is recommended that the activation of this type of paper sludge should be at temperatures of between 650 and 700ºC for 2 hours, so as to ensure high pozzolanic activity, to reduce energy costs and to minimize the generation of CO_2 associated with the calcite decarbonation process. Higher temperatures generate high contents of quicklime, whereas lower temperatures reveal the presence of kaolinite that is not transformed into metakaolinite.

At present, in view of the current global crisis, the preparation of commercial cements with more than one pozzolan (Types II/M, IV and V) acquires great importance from the economic and energetic point of view. For this reason, the pozzolanic behavior of this activated waste is analyzed when mixed with fly ash (1:1 by weight), as this is one of the most widely-used pozzolans in the world (Sanjuan, 2007).

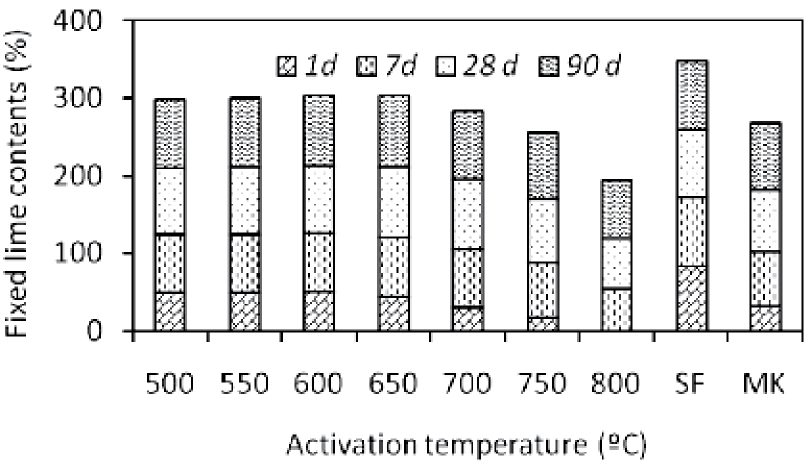

Fig. 5. Evolution of fixed lime (%) versus reaction time

Fig. 6 summarizes the evolution of pozzolanic activity for the activated sludge-fly ash systems $Ca(OH)_2$, for the first 90 days of the reaction. The figure shows an analysis of two waste paper sludges activated in different ways: a laboratory scale production (LPS) obtained under optimal conditions and secondly, an industrial scale production (IPS) at temperatures of over 700ºC, which is commercialized and patented (Patent, 1996).

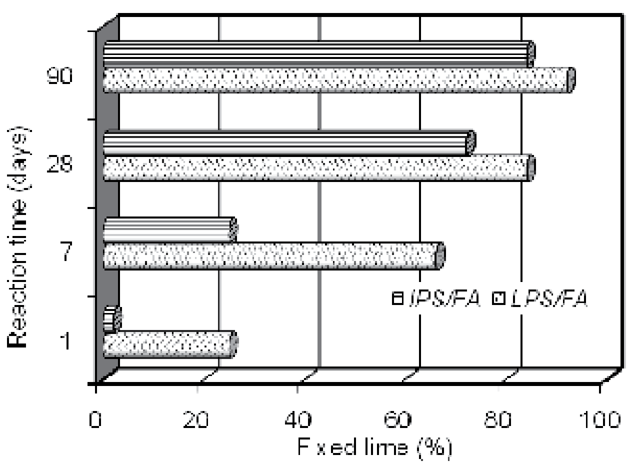

Fig. 6. Evolution of fixed lime in pozzolanic activated sludge mixtures /FA

The results show that the pozzolan mixtures under analysis behave in different ways. As the fly ash is the same in both cases, these differences in pozzolanic activity are directly related to the activation conditions of the paper sludge. This fact may be explained as the consequence of the sludge activation temperature that is higher at an industrial scale than it is at a laboratory scale. Moreover, other parameters may be involved such as the morphology of metakaolinite, the different origins of the paper sludge and the kaolinite/calcite ratio.

After a reaction time of 28 days, the pozzolanic behavior of both mixtures was very similar and evened out at reaction times of over 90 days, due to the slower speed of the pozzolanic reaction of the fly ash (Sánchez de Rojas et al., 1993 and 1996). It is worth highlighting that in the ISP/FA mixture, a significant jump in lime consumption takes place between day 7 and day 28 of the reaction time. This fact may be due to the fly ash acting as an activator of the pozzolanic reaction between the activated sludge and the surrounding lime, as additional quicklime is available from the industrial sludge.

3. The behavior of binary and ternary blended cement prepared with thermally activated paper sludge

3.1 Scientific aspects

3.1.1 Reaction kinetics in binary cements with the addition of 10% activated sludge

In general, the kinetics of pozzolanic reactions depends on various chemical, physical and mineralogical factors. In a study of the influence of the activation conditions on the hydrated phases, percentages of 10 and 20% cement were replaced in this study, which gave similar results. For example, the mineralogical behavior is described here over the reaction time in prismatic specimens (1x1x6 cm) of paste cement prepared with the addition of 10% paper sludge calcined at 700ºC/2h.

XRD and SEM/EDX techniques were used to perform the kinetic study of the reaction, so as to semi-quantify the formation of hydrated phases and the development of their morphologies with the reaction time. The XRD results are provided in Table 2, where the appearance of allite, portlandite, calcite, calcium aluminate hydrates (C_4AH_{13}), and LDH compounds (or compounds of double oxides, at times referred to as hydrotalcite-type compounds) were detected; the last three materials being the most stable over longer periods.

Cement with 10% activated sludge	1 day	7 days	28 days	180 days	360 days
Allite (%)	21	10	9	4	1
Portlandite (%)	38	37	41	32	27
C_4AH_{13} (%)	6	13	5	7	8
LDH compounds (%)	2	1	19	15	9
Calcite (%)	33	40	27	41	55

Table 2. Semiquantitative mineralogical composition by XRD of cement pastes with the addition of 10% activated sludge

Morphologically, layers of allite surrounded by CSH gels are much more easily identified by SEM/EDX (Fig. 7a), although they go undetected by XRD, given their amorphous nature. The CSH aggregates are arranged in bundles of short fibers, together with the LDH compounds (Fig. 7b) and the same situation reoccurs throughout the test period. Chemical composition by EDX analysis after curing for one year is shown in Table 3.

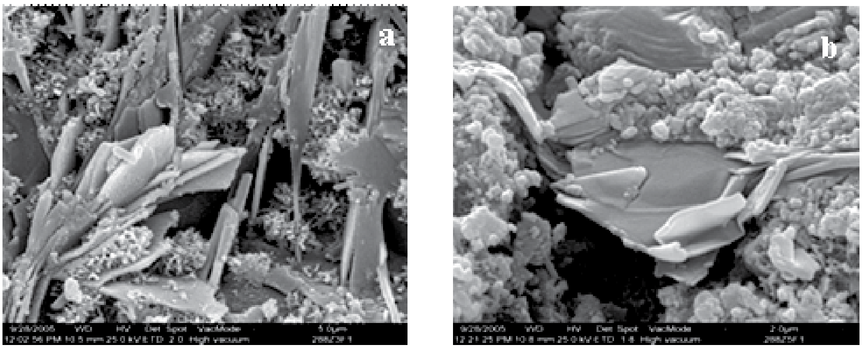

Fig. 7. a) Aggregates of gels and allite layers; b) CSH gels and LDH compounds

Oxides (%)	C-S-H Gel	Allite	Portlandite	LDH Compounds
Al_2O_3	6.27±0.38	2.87±0.36	-	7.70±0.58
SiO_2	29.45±1.52	29.87±1.46	-	22.66±1.04
CaO	64.28±0.95	67.26±0.74	100	69.64±1.36
CaO/SiO_2	2.18	2.25	-	3.07
CaO/Al_2O_3	10.25	23.44	-	9.04
SiO_2/Al_2O_3	4.70	10.41	-	2.94

Table 3. EDX chemical analysis in the cement with the addition of 10% calcined sludge.

3.1.2 Reaction kinetics in ternary cements prepared with 21% pozzolan mixture

In the case of paper sludge, the pioneering studies (Pera et al., 1998 and 2003) established that the formation of their hydrated phases depended on the relative quantities of metakaolinite and calcium carbonate present in the calcined sludge. Any variant that is introduced into the system will have a direct influence on the kinetic reaction. This is the case of the pozzolan mixtures where the influence of the calcined sludge in the reaction will be conditional upon the competitiveness of the other reaction with the surrounding lime. The absence of scientific works in this area means that these aspects are not extensively applied to the technical properties of cement matrices, principally with regard to their durability.

The study of these scientific aspects is based on ternary cements, prepared with the substitution of different percentages of Portland cement (6%, 21%, 35% and 50%), which gave similar results. The description therefore centered on the samples in which 21% of the Portland cement was replaced by a mixture of pozzolans, activated sludge and fly ash at a ratio of 1:1 by weight. The result of this system was the same for the OPC/activated sludge system, except for the presence of mullite from the fly ash and type II CSH gels, according to the Taylor classification (Taylor, 1997), with Ca/Si ratios of between 1.5 and 2.5 (Fig. 8).

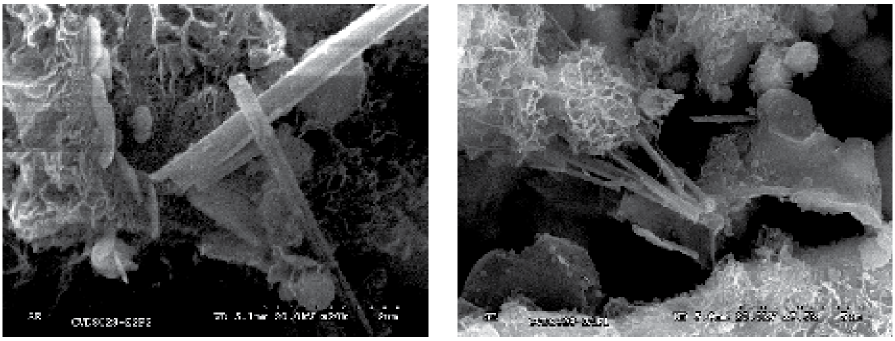

Fig. 8. Left) Formation of gels and layers on amorphous forms. Right) Bundles of CSH gel (II) fibers.

3.2 Technical aspects of blended cements

3.2.1 Properties of binary cements in fresh and hardened states prepared with thermally activated paper sludges

The fresh state of any base cement material may be defined as the period between the initial cement hydration process and its setting. During this period the mixtures show a plastic behavior. A study of a base cement mixture during its plastic state and its properties are of special interest, in order to ensure appropriate preparation and transport and the on-site laying of mortars and concretes. Once the cement has set, the material shows a certain capacity to withstand mechanical stress.

The binary mixtures were studied on the basis of the reference cement pastes and mortars prepared with proportions (0%, 10% and 20% of the Portland cement (CEM I 52,5N) replaced by paper sludge activated at 700ºC for 2 hours. The mortars were prepared at a water/binder ratio of 0.5 and at a binder/sand ratio equal to 1/3. Table 4 presents the main characteristics of the blended cements in their fresh state.

Percentage in weight of CEM I 52.5N Portland cement substituted by calcined paper sludge	Ratio of water consistency/ binder	Initial setting time (minutes)	Final setting time (minutes)	Expansion by Le Chatelier needles (mm)
100/0	0.29	145	255	< 0.5
90/10	0.32	120	170	< 0.5
80/20	0.37	60	130	< 0.5

Table 4. Fresh state properties of binary blended cements prepared with paper sludge calcined at 700ºC

The incorporation of thermally activated paper sludge under optimal conditions produces a parabolic increase in water demand for normal consistency. The greater specific surface of the thermally activated paper sludges, together with the distribution of finer sized particles, complicates the fluidity of the paste. Greater quantities of water are required with these additions to wet the cement surface.

These paper wastes accelerate setting times, especially when they replace percentages of over 10% of Portland cement (Vegas et al., 2006; Frías et al., 2008e). This phenomenon may be attributed to the joint presence of metakaolinite and calcium carbonate. Ambroise and colleagues (Ambroise et al., 1994) demonstrated that MK produces an accelerator effect on the hydration of C_3S when the ratio C_3S:MK is below 1.40; or in other words, when up to 30% of clinker is replaced by MK.

The expansion results reveal that the inclusion of activated paper sludge does not influence the variation in the volume of cement pastes. In fact, the values of the test are well below the limit of 10 mm established in the UNE-EN 197-1 for common cements.

Fig. 9 illustrates the evolution of relative compressive strength determined for standardized mortars with partial additions of 0%, 10% and 20% of thermally activated paper sludge. Up until 14 days of curing, an increase is observed in the relative compressive strength, as the incorporation of calcined paper sludge is increased. The acceleration of cement hydration and the pozzolanic reaction constitute the principal effects that explain the evolution of these strengths. The relative maximum is achieved after 7 days of curing. Likewise, replacement of 20% of the cement by calcined sludge provides greater relative compressive strength during the first fortnight of curing. This discussion coincides with the findings of other authors (Wild et al., 1996) when studying this mechanical property in cement mortars or concretes prepared with pure metakaoline. The lower the content of metakaolinite in the added sludge (10%), the further the values of relative compressive strength will fall for curing periods of over 14 days. The pioneering studies of Pera (Pera & Ambroise, 2003) demonstrated that the most influential parameter in pozzolanic activity at 28 days is the quantity of metakaolinite present in the sludges, regardless of other parameters, such as specific surface area, numbers of particles under 10 micrometers or the average diameter of the distribution of particle sizes.

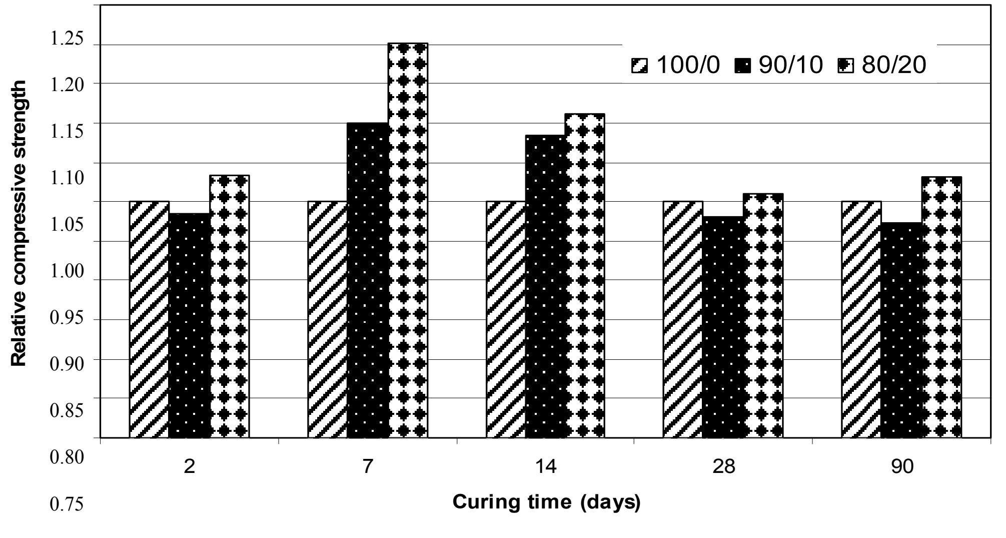

Fig. 9. Relative compressive strength of blended cements with paper sludge calcined at 700ºC

Table 5 presents other physico-mechanical properties of binary blended cements with paper sludge calcined at 700ºC.

Percentage in weight of CEM I 52.5N Portland Cement replaced by calcined paper sludge	Modulus of longitudinal deformation (GPa)	Total retraction at 28 days (%)	Creep deformation after one year of constant load (%)
100/0	34.8	0.04	0.114
90/10	33.0	0.09	0.120
80/20	34.9	0.12	0.093

Table 5. Modulus of longitudinal deformation, retraction and creep of binary cement mixtures prepared with paper sludge calcined at 700ºC

In general terms, it may be concluded that the inclusion of paper sludge calcined at 700ºC, up to a percentage of 20%, hardly modifies the value of the elastic modulus at 28 days of curing. There are few bibliographic references that cover the influence of pozzolanic additions on this mechanical parameter. Qian (Qian & Li., 2001) establishes that the partial replacement of cement by metakaolin, in percentages of up to 15%, produces an increase in the concrete's elastic modulus. These mineral additions show a certain refinement in the porous network of the base cement material; above all, for amounts replaced of 20%. This greater densification means that the fines contribute to a greater extent to the modulus of deformation.

Drying shrinkage increases with the percentage inclusion of paper sludge calcined at 700ºC. After 28 days of drying, cement shrinkage with 20% thermally activated paper sludge triples that shown in the reference cement sample. Greater contraction shown by those mortars that incorporate thermally activated paper sludge may be explained on the basis of phenomenon such as:

- Nucleation of hydration products on the particles of this mineral additions, accelerating the hydration of cement, and therefore, increasing the drying of the product.
- Pozzolanic reaction between the metakaolinite and the calcium hydroxide, either from the calcined sludge, or from hydration of the cement clinker. This reaction requires greater water consumption, accelerating drying of the mixture.
- Increase in capillary pressure, as a consequence of a greater refinement of the distribution of pore size. The greatest relative refinement is observed at 14 days of curing.

The inclusion of 20% thermally activated paper sludge reduces creep deformation by approximately 20% of the deformation observed in the reference mortar sample, after one year subject to a pressure state of 40% of the respective compressive strengths. In a similar way to the explanations of other mechanical characteristics, this reduction may be attributed to a denser pore structure, a stronger cement matrix, and greater adherence between the cement paste and the fines (Brooks & Megat, 2001). As a more refined porous network is created, the movement of free water is prevented, which is responsible for the initial creep. Likewise, the pozzolanic activity contributes to the consumption of water, and therefore, to reductions in early creep.

3.2.2 Properties of ternary blended cements prepared with thermally activated paper sludge and fly ash

The characteristics of the ternary mixtures were determined in standardized pastes and mortars prepared with Portland cement (CEM I 52.5N), thermally activated paper sludge calcined at 700ºC and fly ash. Table 6 presents the percentage mixture of each agglomerate.

Percentages in weight OPC replaced by calcined paper sludge	CEM I 52.5N (% in weight)	Paper sludge calcined at 700ºC (% in weight)	Fly ash (% in weight)
100/0	100	0	0
94/6	94	3	3
79/21	79	10.5	10.5
65/35	65	17.5	17.5
50/50	50	25	25

Table 6. Proportions of ternary cement mixtures with activated paper sludge

Table 7 presents the principal characteristics of the ternary cement mixtures under study in their fresh state.

Percentage in weight of OPC replaced by calcined paper sludge and fly ash	Ratio water consistency /binder	Initial setting time (minutes)	Final setting time (minutes)	Expansion Le Chatelier needles (mm)
100/0	0.28	155	270	0.7
94/6	0.29	140	225	0.3
79/21	0.31	105	165	0.5
65/35	0.34	90	165	0.4
50/50	0.41	35	70	0.2

Table 7. Fresh state properties of ternary cement mixtures prepared with paper sludge calcined at 700ºC and fly ash

In a similar way to the description of the study of binary mixtures, the thermally activated paper sludge calcined at 700ºC appears to control water demand for water consistency, although this result is less apparent in binary mixtures due to the presence of fly ash. This latter mineral addition requires less water content as a consequence of its spherical morphology, thereby minimizing the surface/volume ratio of the particle (Li & Wu, 2005). Likewise, the joint presence of paper sludge calcined at 700ºC and fly ash accelerates the setting times, though there is no evidence of a significant effect on the expansion of cement pastes.

Fig. 10 illustrates the evolution of relative compressive strength determined from standardized cement mortars with partial additions of 0%, 6%, 21%, 35% and 50% of the mineral additions under study. The ternary cements 79/21, 65/35 and 50/50, with a thermally activated paper sludge content of over 10% in weight, display lower mechanical strength than the reference cement sample, although the decrease in their strength is lower than the total percentage of cement that is replaced. At 90 days, a recovery of mechanical resistance is observed in the ternary cements as a consequence of the activity developed by the fly ash.

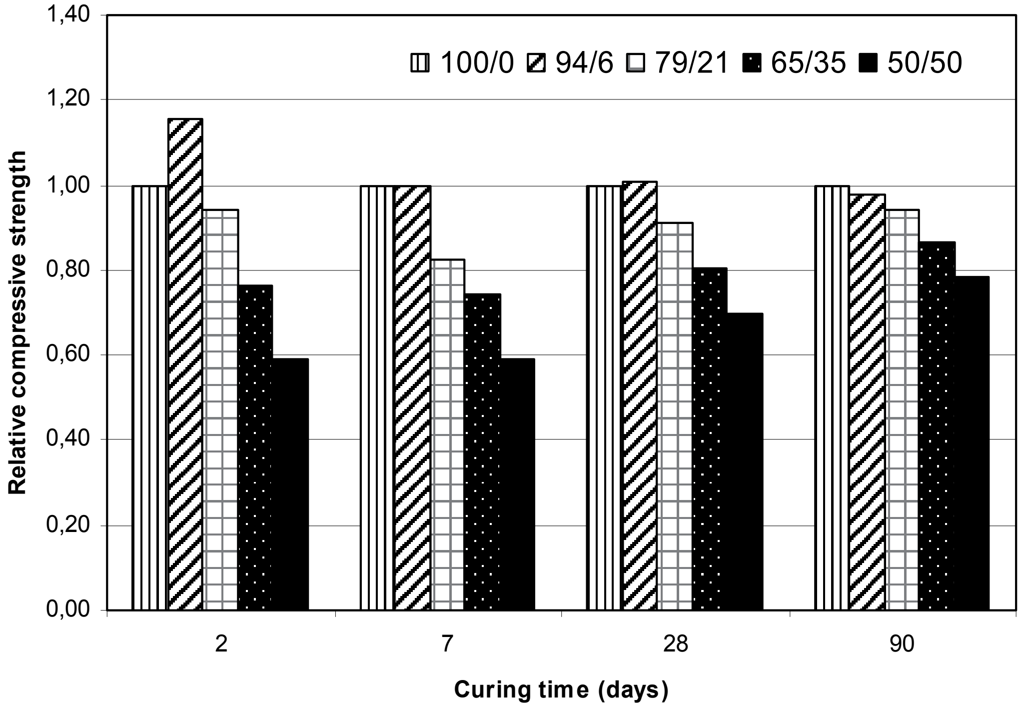

Fig. 10. Relative compressive strength in relation to ternary cement mixtures with paper sludge calcined at 700ºC and fly ash

3.2.3 Durability aspects

Durability is understood as a capacity that maintains a structure or element safely in service for at least a specific period of time, which is referred to as its useful life, in the environment where it will be sited, even when the surrounding conditions (physical, chemical and biological) are unfavorable. In short, the condition demanded from the construction materials and components is that they should perform the function for which they were intended, throughout a certain period of time.

This section discusses the behavior of binary mixtures prepared with thermally activated paper sludge when exposed to weathering action. The durability of the ternary mixtures is at present under study, for which reason it can not be included in this chapter. Among the various degradation mechanisms, two types of aggressive attack are covered: one of a physical nature where extreme temperatures and water intervene, the second of a chemical type in the presence of sulfates.

3.2.3.1 Behavior in the face of freezing/thawing cycles

Binary cement mortars that include 10% and 20% thermally activated paper sludge present, respectively, two and three times more strength faced with freezing/thawing actions than the standard reference mortar (Fig. 11). As the exposure cycles progress, the increase in total porosity is less for those cements that incorporate thermally activated paper sludge. The higher the percentage substitution of cement by calcined paper sludge, the denser the mortar microstructure throughout a higher number of freezing/thawing cycles. Moreover, the greater the replacement percentage of thermally activated paper sludge, the slower the loss of compressive strength in the mortars exposed to freezing/thawing cycles (Vegas et al., 2009).

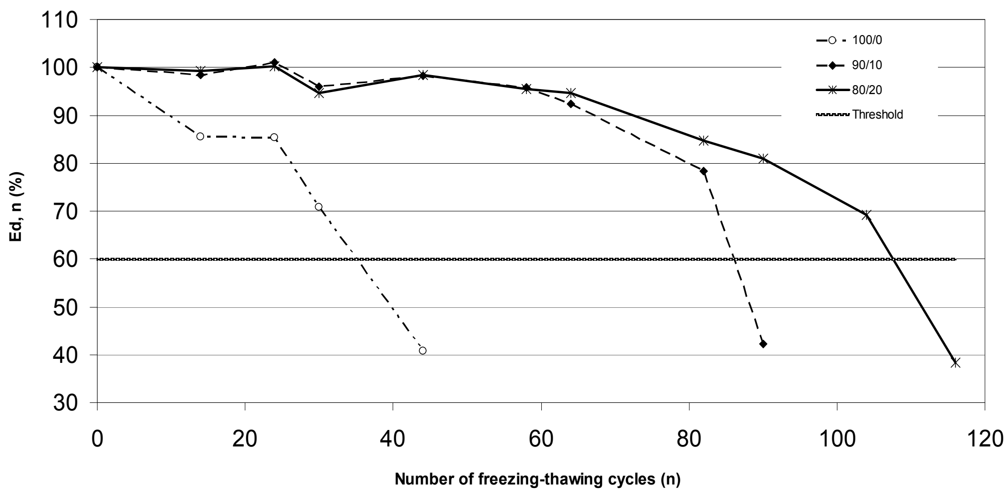

Fig. 11. Evolution of the dynamic modulus of binary cement mixtures with paper sludge activated at 700ºC subjected to freezing/thawing cycles

3.2.3.2 Resistance to sulfates

It is well known that sulfates constitute one of the most aggressive agents against cement based materials, and cause different deterioration mechanisms as a consequence of the direct reaction between sulfate ions and the alumina phases in the cement, giving rise to ettringite, a highly expansive compound. The cements prepared with pozzolans of a siliceous-aluminous nature (fly ash and metakaolinite) can be more susceptible to sulfate attacks, owing to the incorporation of the reactive alumina of the pozzolan (Taylor, 1997; Siddique, 2008). The bibliographic data found on the behavior of normal Portland cements prepared with calcined paper sludge highlights the lower strength in the face of sulfate attacks (external and internal source) with respect to the reference cement sample. Thus, in accordance with the research into cement/calcined sludge/gypsum mortars by Vegas (Vegas, 2009) that is in agreement with the American standard (ASTM C 452-95), the following considerations are proposed:

- The reference cement (CEM I 52.5N) may be categorized by a high resistance to sulfates, given that $\Delta L_{28\ days} \leq 0.054\%$ and $\Delta L_{14\ days} \leq 0.040\%$.
- Binary mixtures with percentages of thermally activated paper sludge above 10% may be classified as having low resistance to sulfates presenting a $\Delta L_{28days} \geq 0.073\%$.
- Observing the increase in length at 7 days, and in accordance with the physical requirements of the ASTM C 845-04 standard, binary cements with 10% and 20% in

volume of activated paper sludge may be classified as hydraulic cements, given that the values ΔL_{7days} are greater than 0.04% and less than 0.10%.

5. Conclusions

The paper industry that uses 100% recycled paper as a primary material generates waste paper sludge which, by its nature, constitutes an inestimable source of kaolin, with the subsequent environmental benefits.

Controlled calcination of waste (500-800ºC) supplies an alternative approach to obtain recycled metakaolin, a highly pozzolanic material for the manufacture of commercial cements.

The products obtained in this way present a high pozzolanic behavior, comparable to a natural metakaolin, which is very close to silica fume; temperatures of between 650-700ºC and 2 hours of retention time in the furnace are established as the most efficient laboratory conditions to obtain these pozzolans. It is likewise worth highlighting their high pozzolanic compatibility with fly ash.

The cement pastes prepared with 10% sludge calcined at 700ºC/2h generate LDH compounds and CSH gels as stable products. The incorporation of a second pozzolan (fly ash) into the blended cement system does not modify the reaction kinetics, for which reason it is worth highlighting the compatibility between both pozzolans.

In the manufacture of binary cements, and in a similar way to the regulations for silica fume, it is recommended that the percentage should be limited to around 10% clinker for paper sludge calcined at 700ºC. A compromise has to be reached between the positive effect on the mechanical properties and the determining factors associated with the reduction in setting times, loss of workability and excessive total drying shrinkage.

In the manufacture of ternary cements that contain sludge calcined at 700ºC and fly ash, the percentage of clinker replaced by the addition of these minerals should not exceed 21%, in order to guarantee the maximum pozzolanic effect (synergy between the two industrial by-products), while ensuring that the workability of the mixture is not adversely affected.

The results of this research have clearly shown the scientific and technical viability of including thermally activated waste paper sludges as active admixtures in the manufacture of binary and ternary cements.

6. Acknowledgements

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17

Agroindustrial Wastes as Substrates for Microbial Enzymes Production and Source of Sugar for Bioethanol Production

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1. Introduction

Environmental issues and concerns aimed at reducing the ambient pollution have boosted the search for "clean Technologies" to be used in the production of commodities of importance to chemical, energy and food industries. This practice makes use of alternative materials, requires less energy, and diminishes pollutants in industrial effluents, as well as being more economically advantageous due to its reduced costs. Considering this scenario, the use of residues from agroindustrial, forestry and urban sources in bioprocesses has aroused the interest of the scientific community lately. The utilization of such materials as substrates for microbial cultivation intended to produce cellular proteins, organic acids, mushrooms, biologically important secondary metabolites, enzymes, prebiotic oligosaccharides, and as sources of fermentable sugars in the second generation ethanol production has been reported (Sánchez, 2009). Notably, the microbial enzymes can be the products themselves as well as tools in these bioprocesses. Agroindustrial wastes are valuable sources of lignocellulosic materials. The lignocellulose is the main structural constituent of plants and represents the primary source of renewable organic matter on earth. It can be found at the cellular wall, and is composed of cellulose, hemicellulose and lignin, plus organic acids, salts and minerals (Pandey et al., 2000; Hamelinck et al., 2005). Therefore, such residues are superior substrates for the growth of filamentous fungi, which produce cellulolytic, hemicellulolytic and ligninolytic enzymes by solid state fermentation (SSF). These fungi are considered the better adapted organisms for SSF, since their hyphae can grow on the surface of particles and are also able to penetrate through the inter particle spaces, and then, to colonize it (Santos et al., 2004). Filamentous fungi are the most distinguished producers of enzymes involved in the degradation of lignocellulosic material, and the search for new strains displaying high potential of enzyme production is of great biotechnological importance. Several agroindustrial wastes are commonly used for this purpose, such as sugarcane bagasse, wheat bran, corn cob and straw, rice straw and husk, soy bran, barley and coffee husk (Sanchéz, 2009). Microbial cellulases, xylanases and ligninases are enzymes with potential application in several biotechnology processes. For decades, such enzymes have been used in the textile, detergent, pulp and paper, food for animals and humans (Bocchini et al., 2003; Kumar et al, 2004;Maicas & Mateo, 2005; Graminha et al., 2008; Hebeish et al., 2009). Recently, research has been focused on the potential use of these enzymes for the degradation of lignocellulosic materials, aiming at the releasing of fermentable sugars that can be converted to second generation ethanol by the action of fermentative microorganisms (Buaban et al., 2010; Talebnia et al, 2010). Among the enzymatically saccharified lignocellulosic wastes intended for the production of ethanol one can cite rice straw, wheat bran, wheat straw, sawdust, rice husk, corn straw and sugarcane bagasse, being the later greatly abundant in Brazil (Binod et al., 2010; Martín et al., 2007). Bearing this in mind, research has been focused on the development of new technologies capable of making the sugar available from bagasse, in order to supply the internal market and also to be exported (Cerqueira Leite et al., 2009). The intimate chemical and physical association between lignin and polysaccharides from the plant cell wall makes the enzymatic degradation of the carbohydrate portion difficult, and consequently the extraction of fermentable sugars, since this phenylpropanoid polymer is not easily degraded biologically. Furthermore, the crystalline structure of cellulose prevents the action of microbial enzymes (Gould, 1984). In order to facilitate the enzymes access to the polysaccharides, especially the cellulose, several pretreatments of the lignocellulosic materials have been proposed, with the intention of disorganizing the plant cell wall structure and lignin removal (Krishna, Chowdary, 2005). In this chapter, we will approach the application of lignocellulosic wastes as substrates for the growth of microorganisms able to produce enzymes such as cellulases, hemicellulases and ligninases, and as sources of fermentable sugars in the production of second generation ethanol, via enzymatic hydrolysis. It will be emphasized the composition of the main residues, the prominent microorganisms, their enzymes and mechanisms of action involved with lignocellulose degradation, SSF characteristics, pretreatment methods and enzymatic hydrolysis of lignocellulosic material, as well as the strategies that have been explored for second generation ethanol production.

2. Lignocellulose

Lignocellulose is the name given to the material present in the cell wall of higher terrestrial plants, made up of microfibriles of cellulose embebed in an amorphous matrix of hemicellulose and lignin (Fig. 1) (Martínez et al., 2009).

These three types of polymers are strongly bonded to one another and represent more than 90% of the vegetable cell's dry weight. The quantity of each polymer varies according to the species, harvest season and, also, throughout different parts of the same plant. In general, softwoods (gimnosperms such as pine and cedrus) have higher lignin content than hardwoods (angiosperm such as eucalyptus and oak). Hemicellulose content, however, is higher in gramineous plants. In average, lignocellulose consists of 45% of cellulose, 30% of hemicellulose and 25% of lignin (Glazer & Nikaido, 2007). Lignocellulosic materials also include agricultural residues (straws, stover, stalks, cobs, bagasses, shells), industrial residues (sawdust and paper mill discards-, food industry residues), urban solid wastes e domestic wastes (garbage and sewage) (Mtui, 2009).

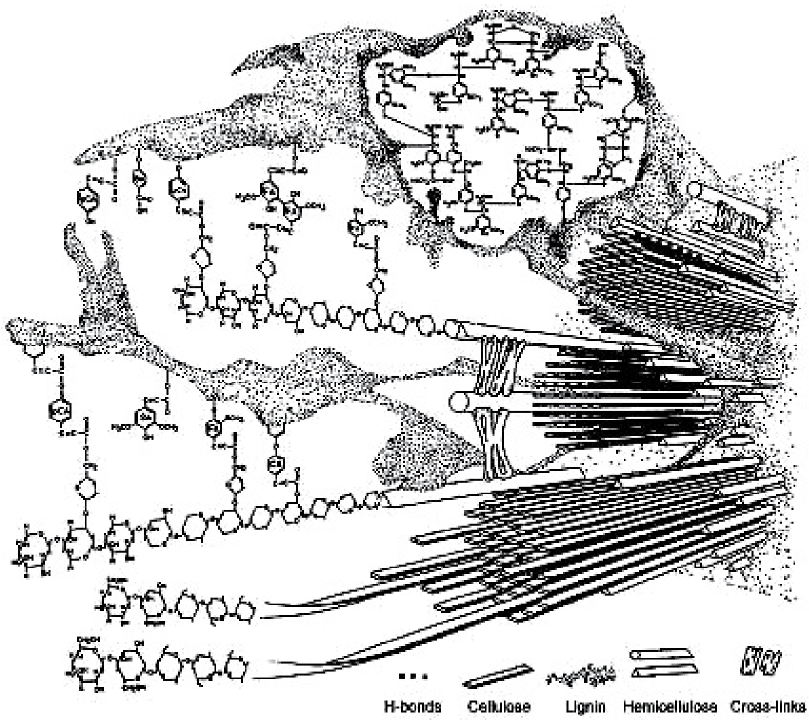


Fig. 1. Scheme of secondary plant cell wall. CA: *p*-coumaric acid; FA: ferulic acid; SA: sinapic acid. Source: Bidlack et al, 1992.

Lignocellulose is the world's main source of renewable organic matter and the chemical properties of its components make it a material of great biotechnological value. Therefore, a few years ago, the concept of lignocellulose biorefinery emerged, which has received growing attention due to the potential of conversion of this material into many high added value products such as chemical compounds, fermentation substrates, feedstock and biofuels (Ragauskas et al., 2006; Demirbas, 2008).

The accentuated growth of the world's consumption of energy originating from fossil resources has aggravated the problem of atmospheric pollution by the release of gases related to the greenhouse effect. For this reason, besides the high cost of petroleum and the eminent depletion of these resources in a few decades from now, the obtainment of fuels from renewable sources, such as lignocellulosic biomass, has aroused great interest in the last years. Currently, it is believed that ethanol, as the main form of bioenergy, is the best alternative to the use of fossil fuels (Wang et al., 2011).

Inside this context, new technologies have been developed for the efficient obtainment of fuels from lignocellulosic bimass and developed and developing Countries have been focusing efforts in researches aimed at obtaining the so called biofuels, such as bioethanol and biodiesel, as well as their introduction and prevalence in the market (Hamelinck et al., 2005; Prasad et al., 2007).

2.1 Cellulose

Cellulose is the most abundant organic compound on Earth and the main constituent of plant cell walls. It consists of linear chains of aproximately 8,000 to 12,000 residues of D-glucose linked by β-1,4 bonds (Timell, 1967; Aro et al., 2005). Cellulose chains exhibit a flat structure, stabilized by internal hydrogen bonds (Fig. 2). All alternate glucose residues in the same cellulose chain are rotated 180°. One glucose residue is the monomeric unit of cellulose and the dimer, cellobiose, is the chain's repetitive structural unit (Brown Jr et al., 1996). Cellulose chain is polarized, once there is a nonreducing group at one of its end and, at the opposite end there is a reducing group. New glucose residues, originating from UDP-glucose, are added to the nonreducing end during polymer synthesis (Koyama et al., 1997). Parallel cellulose chains interact, through hydrogen bonds and van der Waals forces, resulting in microfibriles, which are very extensive and crystaline aggregates (Glazer & Nikaido, 2007; Somerville et al., 2004).

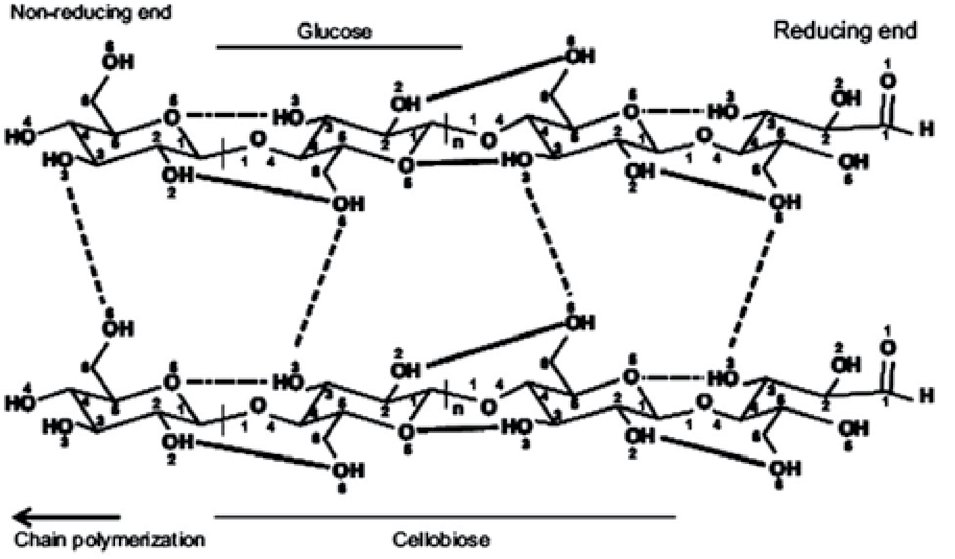


Fig. 2. Representative structure of cellulose chains. Dotted and solid lines: inter and intra chain hydrogen bonds, respectively. Source: Festucci-Buselli et al., 2007.

The microfibriles are made up of aproximately 30-36 glucan chains, exhibit a 2-10 nm diameter and are cross-linked by other components of the cell wall, such as the xiloglucans. (Arantes & Saddler, 2010). The cellulose microfibriles networks are called macrofibrils, which are organized in lamellas to form the fibrous structure of the many layers of plant cell wall (Fig. 3) (Glazer & Nikaido, 2007).

In cellulose fibers, crystaline and amorphous regions alternate. The crystaline regions are very cohesive, with rigid structure, formed by the parallel configuration of linear chains, which results in the formation of intermolecular hydrogen bonds, contributing to cellulose's insolubility and low reactivity, at the same time making it more resistant to acid hydrolysis, making water entrance difficult and modifying fiber elasticity. The amorphous regions are formed by cellulose chains with weaker organization, being more accessible to enzymes and susceptible to hydrolysis (Bobbio & Bobbio, 2003; Nelson & Cox, 2006).

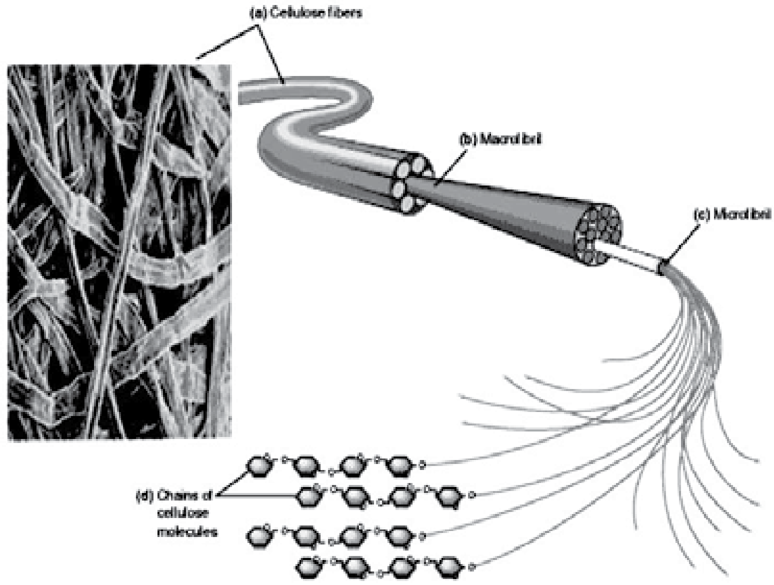


Fig. 3. Representative scheme of cellulose fiber Available from http://nutrition.jbpub.com/resources/chemistryreview9.cfm.

2.2 Hemicellulose

Hemicellulose is the second group of most abundant polyssacharide in plant cell wall and, differently from cellulose, it is made up of non crystaline heteropolyssacharides (Aspinall, 1959).

Schulze (1891) initialy classified hemicellulose as the polyssacharide fraction of plant cell wall easily hydrolized, an imprecise classification which was used for a long time. The group also defined as polyssacharides present in plant cell wall and intercellularly (in the middle lamella), extracted from higher terrestrial plant tissues through alkaline treatment or, yet, as certain carbohydrates of cereal endosperms, which are non starch polyssacharides that are described as cereal gum or pentosans (Timell, 1965; Wilkie, 1979).

Afterwards, hemicellulose classification was redefined, based on the chemical properties of its components, including only cell wall polyssacharides non covalently bonded to cellulose made up by β-(1,4)-linked pyranosyl residues that have the *O*-4 in the equatorial position. Such characteristics result in a conformation that is very similar to cellulose (cellulose-like conformation) and cause a tendency to hydrogen-bond to cellulose chains (O'Neill & York, 2003).

In a general way, the hemicellulose fraction makes up 15 to 35% of plant biomass, representing a great renewable source of biopolymers which may contain pentoses (β-D-xylose, α-L-arabinose), hexoses (β-D-mannose, β-D-glucose, α-D-galactose) and/or uronic acids (α-D-glucuronic, α-D-4-O-methylgalacturonic and α-D-galacturonic acids). Other sugars such as α-L-rhamnose and α-L-fucose may also be present in small amounts and the hydroxyl groups of sugars can be partially substituted with acetyl groups (Gírio et al., 2010). Therefore, hemicllulose classification depends on the type of monomer constituent and these may be called xyloglucans, xylans (xyloglycans), mannans (mannoglycans) and β-(1→3,1→4)-glucans (mixed-linkage β-glucans). Galactans, arabinans and arabinogalactans are also many times

included in the hemicellulose group, however do not share the equatorial β-(1→4)-linked backbone structure (Scheller & Ulvskov, 2010). Its form and structure depend on where they are present, being in woods or fruits cell walls (Dey & Brinson, 1984).
Most of the hemicelluloses are relatively small molecules, containing 70 to 200 residues of monossacharides, being hardwood hemicellulose the largest molecules with 150 to 200 units (Coughlan, 1992).
Hemicelluloses associate to cellulose by physical intermixing and hydrogen bonds, and to lignina e pectin, by covalent bonds (Freudenberg, 1965).

2.2.1 Xylans

Xylans, the most relevant components of hemicellulose, constitute the secong group of polyssacharides most abundant in nature, contributing to aproximately one third of all the renewable organic carbon on Earth (Prade, 1995). Xylans constitute about 20–30% of the biomass of hardwoods and herbaceous plants. In some tissues of grasses and cereals xylans can account up to 50% (Ebringerová et al., 2005).
Xylans are mainly situated in the secondary cell wall of plants (Timell, 1965), in close contact to cellulose through strong interactions established by hydrogen bonds and van der walls forces. They may also be present in the primary cell wall, in particular in monocotyledoneans (Wong et al., 1988). Their molecules are oriented parallel to cellulose chains, in the cell wall matrix, and are located between cellulose microfibriles (Northcote, 1972). Xylan, covalently bonded to lignin and non covalentely bonded to cellulose, exhibits important role in maintaining cellulose integrity in situ, protecting the fibers against the action of cellulases (Beg et al., 2001; Uffen, 1997).
Xylans of all higher plants are heteropolysaccharides with a backbone of β-(1→4) linked xylopryranose units, usually substituted with sugar units and O-acetyl groups (Stephen, 1983). Exception to this pattern have been isolated from the seeds of *Plantago* species (Sandhu et al., 1981; Samuelsen et al., 1999) The occurrence of homoxylans in higher plants is rather rare, being isolated from esparto grass (Chanda et al., 1950), tabaco caule (Eda et al., 1976), guar seeds (Montgomery et al., 1956) and from some marine algea (Barry & Dillon, 1940; Nunn et al., 1973).
Methylglucuronoxylans (MXG) (*O*-acetil-4-*O*-metilglucuronoxylans) are dominating in the secondary walls of hardwoods (dicots such as eucalipto e carvalho) having single side chains of α-D-glucuronic acid (GA) and/or its 4-O-methyl derivative (MeGA) attached at position 2 of the xylopyranose monomer units (Ebringerová & Heinze, 2005). The content of acetyl groups varies in the range 3-13% and is responsible for xylan's partial solubility in water (Sunna & Antranikian, 1997).
Arabinoglucuronoxylan (AGX) (arabino-4-O- methylglucuronoxylans) are the major components of non-woody materials (e.g., agricultural crops, grasses) and a minor component (5–10% of dry mass) for softwoods (ex gimnosperms such as pine and cedrus). They contain single side chains of 2-Olinked α-D-glucopyranosyl uronic acid unit (GA) and/or its 4-O-methyl derivative (MeGA) and 3-linked α-L-arabinofuranosyl unit (Timell, 1965; Sunna & Antranikian, 1997).
Into the cell walls polysaccharides of forage and grasses, ferulic acid residues are introduced via an ester linkage between their carboxylic acid group and the primary alcohol on the C5 carbon of the arabinose side chain of arabinoxylans (Hartley, 1973) (Fig. 4), but can also be covalently linked to lignin monomers via an ether linkage (Kondo et al., 1990). So, ferulic acid participates with lignin monomers in oxidative coupling pathways to generate ferulate–

polysaccharide–lignin complexes that cross-link the cell wall (Buanafina, 2009). Esters of p-coumaric acid are also abundant in grass cell walls, but it is not clear if they can be attached directly to the xylans, and they may be primarily associated with lignin (Hatfield et al., 2008).

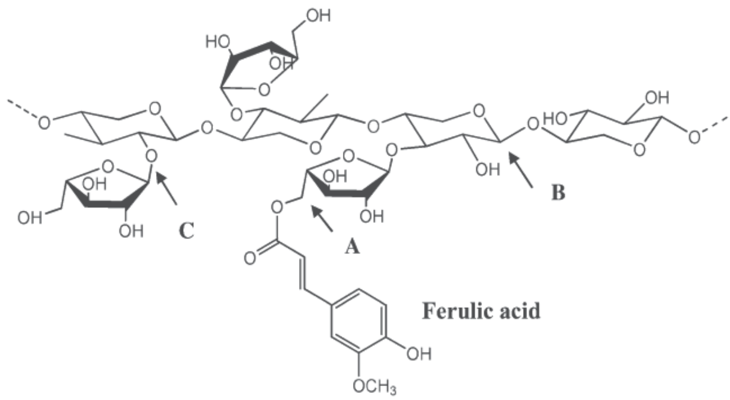
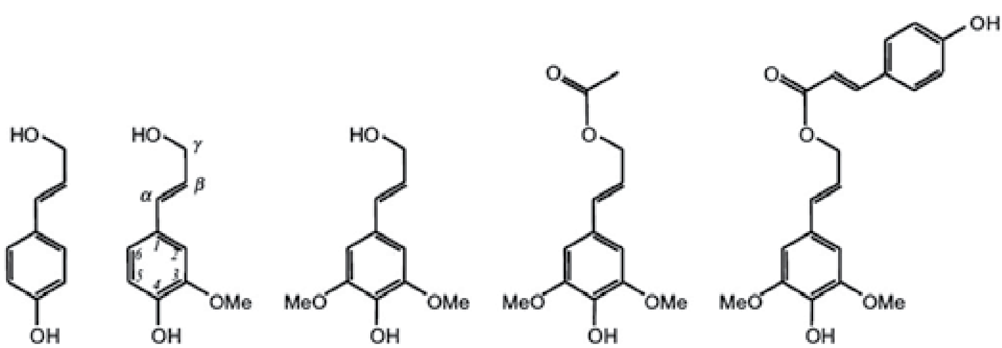


Fig. 4. Structure of ferulic acid esterified to arabinose units of arabinoxylan. A: ferulic acid linked to O-5 of arabinose chain of arabinoxylan; B: β-1,4-linked xylan backbone; C: α-1,2-linked L-arabinose. Source: Buanafina, 2009.The occurrence and structural characteristics of other types of xylan have been reported in detail in previous review papers (Ebringerová & Heinze, 2005; Ebringerová et al., 2005).

2.3 Lignin

Lignin is the second is the second most abundant polymer in the cell wall of vascular plants and in nature, after cellulose. Aproximately 20% of the total carbon fixed by photosynthesis in land ecosystems is incorporated into lignin. It is a complex and recalcitrant aromatic polymer, without defined repetitive units (Hammel & Cullen, 2008), its precursors are three *p*-hydroxycinnamyl alcohols or monolignols (*p*-coumaryl, coniferyl and sinapyl) and their recently reported acylated forms (Ralph et al., 2004; Martínez et al., 2008) (Fig. 5).



1 2 3 4 5

Fig. 5. Lignin precursors or monolignols. Classical: *p*-coumaryl (1), coniferyl (2), sinapyl (3). Acylated: derived from sinapyl alcohol γ-esterified with acetic (4) and *p*-coumaric acid (5). Source: Ruiz-Dueñas & Martínez, 2009 – partially reproduced

Although these precursors are phenolic compounds, the polymer is basically non-phenolic (Fig. 6), due to the high frequency of ether linkages between the phenolic position (C4) and a side-chain (or aromatic ring) carbon of the *p*-hydroxyphenylpropenoid precursors (Fig. 6, substructures A, B and D), strongly predominant in the growing polymer. Unlike cellulose and hemicelluloses, the lignin polymerization mechanism (based on resonant radical coupling) results in a complex three-dimensional network. During the polymer synthesis, a variety of ether and carbon–carbon inter-unit linkages are formed, resulting in many substructures, such as those shown in Figure 6 (Ruiz-Dueñas & Martínez, 2009).

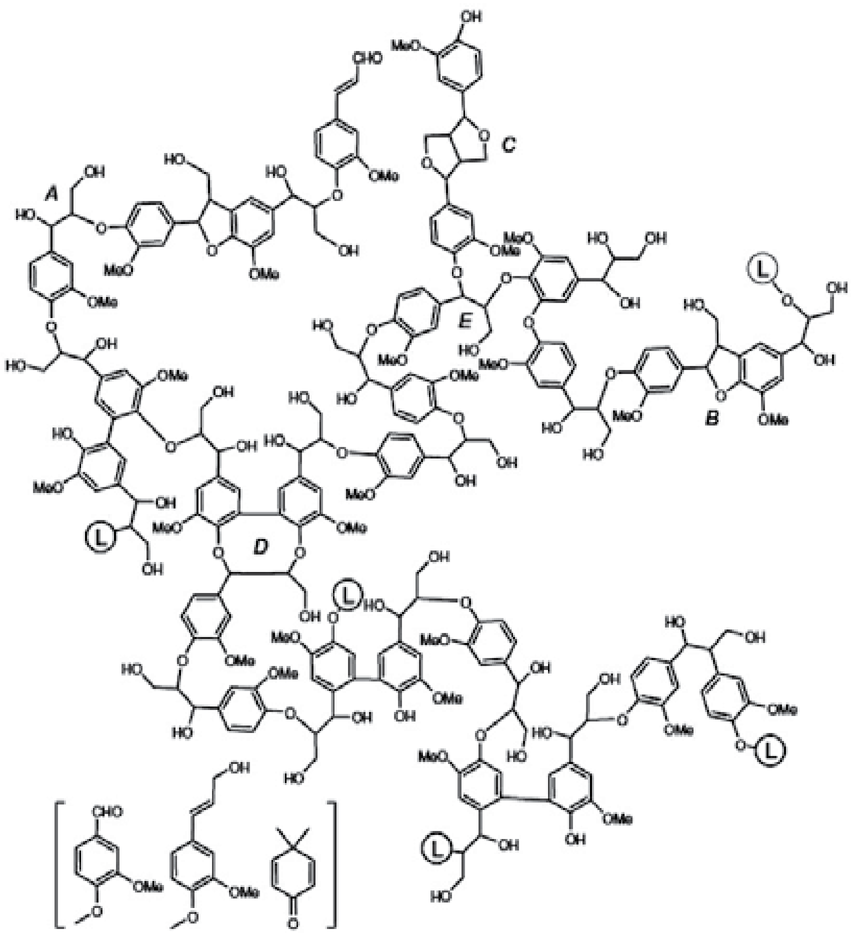


Fig. 6. Lignin structure. Substructures: β-*O*-4′(A); phenylcoumaran (B); pinoresinol (C) and dibenzodioxocin (D). L-containing circles indicate linkages to additional lignin chains. Brackets indicate other minor structures, such as vanillin, coniferyl alcohol and dimethylcyclohexadienone-type units, the latter in new spirodienone substructures. Source: Ruiz-Dueñas & Martínez, 2009 – partially reproduced.

The association of lignin and hemicelluloses occurs through covalent linkages (such as benzyl ester bonds with the carboxyl group of 4-*O*-methyl-d-glucuronic acid in xylan and more stable ether bonds between lignin and arabinose or galactose side groups in xylans and mannans) (Kuhad et al., 1997) and also through noncovalent linkages. These interactions form a dense and organized network that surrounds the cellulose through extensive hydrogen bonding (Fig. 1) (Westbye t al., 2007). This network protects the cellulose and is one of the reasons for biomass recalcitrance. These lignin-carbohydrate complexes (LCC) represent an obstacle for lignocellulose bioconversion processes because lignin hinders enzyme-mediated hydrolysis of carbohydrates, since it acts as a physical barrier, restricting enzyme access to carbohydrates (Mooney et al., 1998). Lignin may also interact with enzymes possibly through hydrophobic interactions resulting in nonproductive binding (Sutcliffe & Saddler, 1986).

3. Lignocellulose-degradating enzymes

Lignocellulosic materials represent an important source of added-value chemicals, such as reducing sugars, furfural, ethanol and other products that can be obtained by enzymatic or chemical hydrolysis. Enzyme-catalyzed hydrolysis of lignocellulosic biomass provides better yields without the generation of side products (Demirbas, 2008).

Enzymatic degradation of vegetable biomass by microorganisms is carried out by a complex mixture of enzyme, amongst which cellulases and hemicellulases stand out, whose action results in free carbohydrates that may be hydrolyzed to soluble saccharides that can be further metabolized, besides ligninases, which promote lignin depolymerization. Most carbohydrate hydrolases are modular proteins that, besides the catalytic site, have a carbohydrate-binding module (CBM) (Jørgensen et al., 2007). CBMs were first discovered on cellulases but it is now evident that many carbohydrate hydrolases acting on insoluble but also soluble polysaccharides have CBMs, which function is to bring the catalytic site in close contact with the substrate and ensure correct orientation. Furthermore, for some CBMs a disruptive effect on the cellulose fibers has also been shown (Boraston et al., 2004).

3.1 Cellulases

The production of cellulases by microrganisms occurs, mainly, by bacteria and filamentous fungi, with few reports of production by yeasts. Ascomycetes and imperfect fungi have great importance for degradating cellulose and decomposing soil vegetable residues, being known as brown rot fungi (Sandgren & Hiberg, 2005).

A cellulolytic system based on 'free' enzymes, that act synergistically to complete cellulose degradation, is typically produced by aerobic fungi and bacteria. This enzyme system includse three types of cellulases (Fig. 7):

i. *Endoglucanases* (EG, endo-1,4-β-D-glucan 4-glucanohydrolase, EC 3.2.1.4): hydrolyses, at random, β-1,4 glucosidic bonds at internal amorphous sites in the cellulose chains, providing more ends for the cellobiohydrolases to act upon;
ii. *Exoglucanases* or *cellobiohydrolases* (CBH, 1,4-β-D-glucan cellobiodehydrolase, EC 3.2.1.91): act on the reducing (CBH I) or nonreducing (CBH II) ends of cellulose chains, liberating cellobiose;
iii. *β-glucosidases* (β-glucoside glycosyl hydrolase or cellobiase, EC 3.2.1.21): hydrolyze cellobiose or cello-oligosaccharides to glucose and are also involved in transglycosylation reactions of β-glucosidic linkages of glucose conjugates (Coughlan & Ljungdahl, 1988).

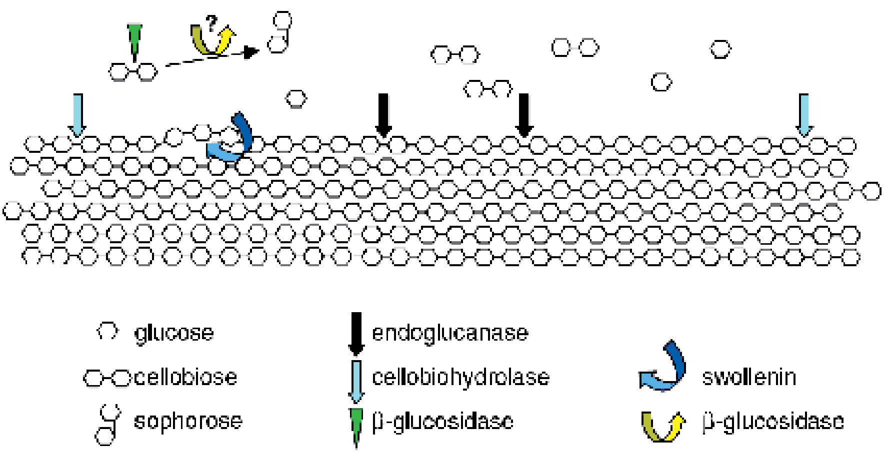


Fig. 7. Schematic representation of the cellulolytic system. Sites of intense cellulolytic enzyme activity are shown, besides an alternative path for the formation of sophorose by β-glicosidase's tranglycosylating activity. Source: Aro et al, 2005.

In addition to the classical cellulases, Figure 7 also shows the action of swollenins, proteins with amino acid similarity to plant expansins that regulate cell wall enlargement in growing cells. Expansins were firstly isolated from *Trichoderma reesei* in 1992 and are thought to disrupt hydrogen bonding between cellulose microfibrils or between cellulose and other cell wall polysaccharides, without hydrolyzing them, causing sliding of cellulose fibers or expansion of the cell wall (Whitney et al., 2000). It has been reported that swollenin action helps enzymatic cellulose degradation since it causes a partial damage and loose structure on cellulose, similar to that caused by ultrasound treatment, without releasing reducing sugar (Saloheimo, et al. 2002)

The enzymes of the cellulolytic complex may be subject to catabolic repression by the final product of hydrolysis. For preventing accumulation of celobiose, β-glicosidase is responsible for controling the overall speed of the cellulolytic hydrolysis reaction, exhibiting a crucial effect on the polymer's enzymatic degradation (Leite et al., 2008).

Cellulases synthesized by anaerobes, particularly clostridia and rumen microorganisms, frequently assemble into a large multienzyme complex (molecular weight >3 MDa) termed cellulosome and first identified in 1983 from the thermophilic and spore-forming *Clostridium thermocellum* (Lamed et al., 1983). This bacterial cellulosome shows very high activity on crystalline cellulose ("true cellulase activity" or Avicelase) which is not commonly observed among fungal cellulases (Johnson et al., 1981).

In *C. thermocellum*, cellulolytic enzymes are typically distributed both in the liquid phase and on the surface of the cells. However, several anaerobic species that degrade cellulose do not release measurable amounts of extracellular cellulase, and instead have localized their complexed cellulases directly on the surface of the cell or the cell-glycocalyx matrix (Lynd et al., 2002).

Besides *Clostridium* and other anaerobic bacteria, evidences suggest the presence of cellulosome in at least one aerobic bacterium and a few anaerobic fungi such as *Neocallimastix*, *Piromyces* and *Orpinomyces* (Fanutti et al., 1995; Li et al., 1997).

In addition to cellulases, cellulosomes include xylanases, mannanases, arabionfuranosidases, lichenases, and pectin lyases (Bayer et al., 2004).

The structure and function of bacterial cellulosomes have been reviewed several times elsewhere (Bayer et al., 1998; Nordon et al., 2009). All cellulosome described share some characteristics (Fig. 8): their enzymes are linked to noncatalytic modules, called dockerins, by carbohydrate-binding modules (CBMs). Dockerins bind, by calcium-dependent interactions, to the cohesin modules, located in a large noncatalytic protein that acts as a scaffoldin. In general, the scaffoldin, a large and distinct protein, allows binding of the whole complex to the plant cell wall, via a cellulose-specific family 3 CBM (CBM3a), and to the bacterial cell via a C-terminal divergent dockerin (Fontes & Gilbert, 2010).
Since the recalcitrance and chemical complexity of some polymers represent an obstacle to the enzymatic degradation of lignocellulose, more efficient enzyme systems are required. Cellulosomes highlight as one of nature's most elaborate nanomachines and the arrangement of plant cell wall degrading enzymes into this complex has advantages over free enzyme systems. Less total protein may be required to solubilize cellulose, including crystalline cellulose, which suggests that specific activity of the cellulosome for such substrates is higher than that of free enzyme systems (Johnson et al., 1982; Boisset et al, 1999). We could say that cellulosome enzymes are "concentrated" and positioned in a suitable orientation both with respect to each other and to the cellulosic substrate, thereby facilitating stronger synergism among the catalytic units. Due to this optimum spacing of the components, working in a synergistic manner, non-productive adsorption is avoided. Since cellulosome is close to the microorganism cell surface, hydrolysis inhibitory products would not accumulate, but would be maintained at appropriate concentrations for most efficient use by the cell (Shoham et al, 1999).

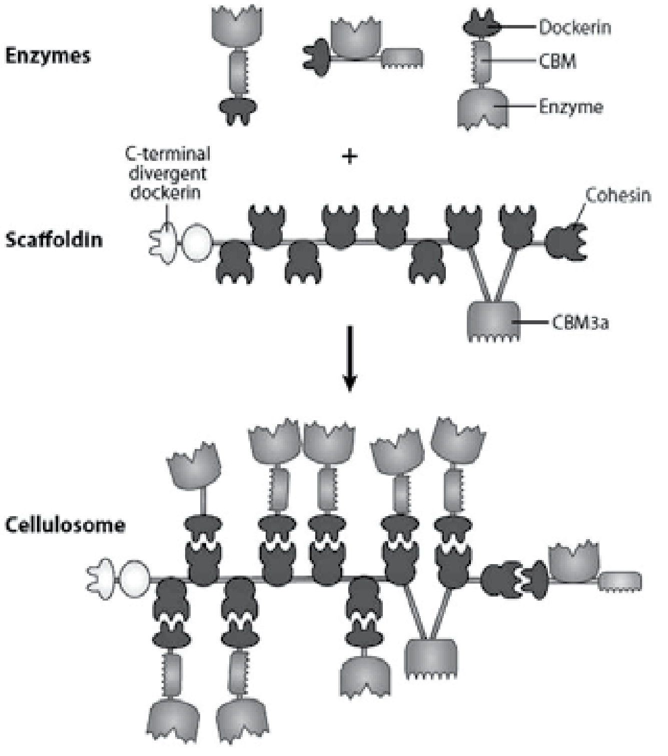


Fig. 8. Mechanism of cellulosome assembly. Source: Fontes & Gilbert, 2010.

3.2 Xylanases

Due to xylans heterogeneity and complexity, the complete hydrolysis of this polysaccharide requires the action of an enzyme system with different specificities and ways of action (Fig. 9). The microbial systems are made up by (Wong et al., 1988):

i. *Endo-1,4-β-D-xylanases* (EC 3.2.1.8): hydrolyze β-1,4 glycosidic linkages between xylose residues in the backbone of xylans;
ii. *1,4-β-D-xylosidases* (EC 3.2.1.37): release β-D-xylopyranosyl residues from the non-reducing terminus of xylobiose and some small 4-β-D-xylooligosaccharides;
iii. *α-L-arabinofuranosidases* (EC 3.2.1.55): removes L-arabinose side chains from the xylose backbone of arabinoglucuronoxylan;
iv. *α-glucoronidases* (EC 3.2.1.1): hydrolyzes the α-1,2 glycosidic bonds between the glucuronic acid residues and β-D-xylopyranosyl backbone units found in glucuronoxylan;
v. *acetyl xylan esterases* (EC 3.1.1.72): removes the *O*-acetyl groups from positions 2 and/or 3 on the β-D-xylopyranosyl residues of acetyl xylan
vi. *p-coumric* and ferulic acid *esterases* (EC 3.1.1.1): cleave ester bonds on xylan, between arabinose and erulic acid sidegroups and between arabinose and *p*-coumaric acid, respectively (Christov & Prior 1993).

The *feruloyl esterases* exhibit a key role providing an increase on hydrolytic enzyme's accesability on hemicellulose fibers due to the removal of ferulic acid from the side chains and cross links (Wong, 2006).

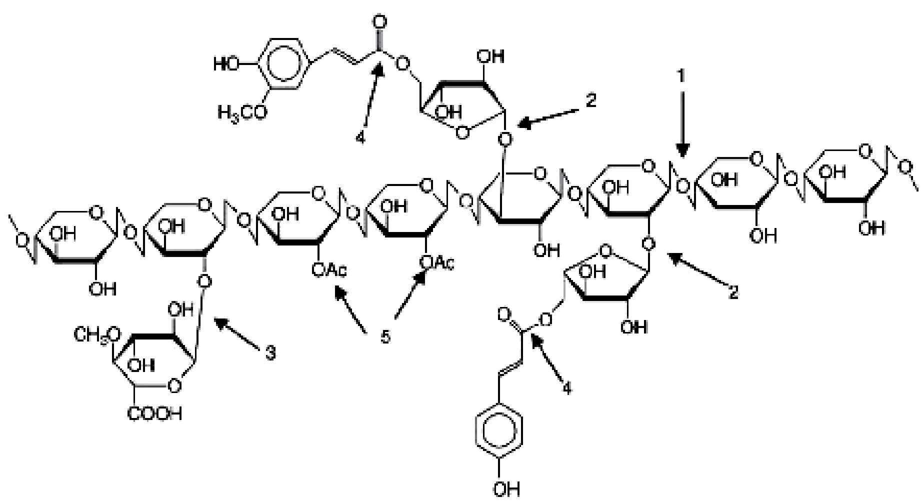


Fig. 9. Schematic representation of the action of the xylanolytic enzyme system. 1 – endoxylanases; 2 - α-L-arabionofuranosidases; 3 – glucuronidases; 4 – feruloyl and coumaroyl esterases; 5 – acetyl xylan esterases; 6-β-xylosidases. Source: Chávez et al., 2006

All xylanolytic enzymes act in a cooperative way to convert xylan into its monomers (Wong et al., 1988). Such multifunctional xylanolytic system may be found in fungi and bacteria (Sunna &Antranikian, 1997), including the actinomycete (Elegir et al., 1995). Some of the most important microrganisms that produce xylanolytic enzymes belong to the genera

Aspergillus, *Trichoderma*, *Streptomyces*, *Phanerochaete*, *Clotridium* e *Bacillus* (Collins et al., 2005; Subramaniyan & Prema, 2002).

3.3 Ligninases

Lignin decomposition is indispensable for carbon recycling since it is the most abundant renewable source of aromatic polymer in nature. Due to its complex and heterogeneous structure, lignin is chemically recalcitrant to breakdown by most organisms. Only the basidiomycetous, called whiterot fungi, are able to degrade lignin efficiently by producing an array of extracellular oxidative enzymes that act synergistically. Among these enzymes, the major groups include lignin peroxidases, manganese peroxidases, and laccases (Wong, 2009).

3.3.1 Laccases

Laccase (E.C. 1.10.3.2) is a multicopper protein belonging to the family of the blue oxidase enzymes. This enzyme generally contains four copper ions, grouped in three groups: T1, formed by one ion and is responsible for substrate oxidation and for electron transfer; T2, also formed by one ion and, together with the T3 group, which contains two ions, constitute the copper trinuclear center, involved in oxygen reduction and water release (Torres et al., 2003). Copper located at the T1 site is the one responsible for the strong absorption of the enzyme at the 600 nm range; however there has been reports of laccases with copper deficiency at T1, called white laccases, due to the absence of the characteristic absorbance at the blue range (Baldrian, 2006).

Laccase is classified as a phenol oxidase and catalyzes the oxidation of various aromatic and inorganic compounds (phenols in particular) and at the same time reduces oxygen to water. Phenolic dyes, phenols, chlorophenols, some diphenylmethanes and benzopirenes are amongst the substrates oxidized by this enzyme (Durán & Esposito, 2000). Laccase can degrade lignin even in the absence of other ligninases, such as manganese peroxidase and lignin peroxidase (Mayer & Staples, 2002).

The catalytic mechanism is based on the reduction of molecular oxygen forming water, at the expense of consecutive monoeletronic oxidation of the substrate.

Laccases may directly interact with phenolic substrates and oxidize them, however they are not capable of directly acting on aromatic non-phenolic molecules, thus requiring a mediator molecule for the degradation of such compounds. In this laccase-mediator mechanism, the mediators oxidize substrates of high molecular mass (Torres et al., 2003). Generally, the mediators are low molecular weight substances, secreted by the fungi itself, which when oxidized by laccases, are capable of oxidizing compounds that would not be the enzyme's direct target. The mediation phenomenon increases the range of substrates of these enzymes (Leonowicz et al., 1999)

3.3.2 Lignin peroxidases (LiP)

Lignin peroxidase (LiP) (E. C. 1.11.1.14) is a glycoprotein that has a prosthetic group made up of iron protoporphirin IX, with catalytic activity dependent on H_2O_2 (Rodriguez & Durán, 1988). The H_2O_2 required for LiP acitivy is originated from different biochemical pathways, expressed according to the nutritional factors and growth conditions of the microrgansim (Pointing, 2001).

LiP is an enzyme capable of oxidizing various non-phenolic aromatic compounds such as benzyl alcohol, cleaving side chains of these compounds, catalyzing aromatic ring opening reactions, demetoxilation and oxidative dechlorination (Conessa et al., 2002).

Veratrilic alcohol may induce enzyme's action, protect it against activation by high levels of H_2O_2, besides acting as co-substrate, carrying out oxidation of the non-phenolic aromatic compounds (Mester et al., 1995; Pointing, 2001).

3.3.3 Manganese peroxidases (MnP)

Manganese peroxidase (MnP, E. C. 1.11.1.13) is a glycosylated extracellular enzyme that has an heme prosthetic group, found only in basidiomycetes. In general, it does not unleash direct transformations on its substrates. The electron transfers involving manganese happen in the presence of dicarboxylic radicals, such as oxalate, malate, fumarate and malonate (Leonowicz et al., 1999). These radicals are Mn^{3+} chelating agents. When complexed, apparently, they act as low molecular weight mediators which will effectively oxidize lignin's phenolic compounds (Higuchi, 2004). MnP's reaction mechanism is similar to that of other peroxidases, such as lignin peroxidase but, in this case, compounds I and II of MnP oxidize Mn^{2+} (Conessa et al., 2002).

4. Solid state fermentation (SSF) and its application on the cultivation of microrganisms that produce lignocellulolitic enzymes

SSF can be defined as the cultivation of microrganisms in a solid insolubel matrix, in the presence of enough water to only support microbial growth (held in the particles). The solid matrix may be used by the microrganism as substrate, or may just serve as as innert support for growth (in this last case, it should be soaked in a nutrient solution) (Pandey et al., 2000). The selection of an appropriate substrate is a key factor for the success of SSF. Besides having the adequate compostition to induce the desired product, the particles size should also be considered, since this is a factor that greatly influences SSF. Small particles offer more contact surface, allowing more access of the microorganism to the nutrients; however, depending on the type of substrate and on the moisture level, it can get compacted, impairing aeration and oxygen availability, as well as heat dissipation, limiting microbial growth. Big particles, on the other hand, facilitate aeration, however may hinder microbial access, limiting substrate contact surface and making heat transfer difficult (PALMA, 2003). Other parameters should also be evaluated and optimized for higher process efficiency, such as initial moisture and pH, incubation temperature, aeration, inoculum size, nutrient supplementation, extraction and purification of the final product (Pandey, 2003).

Solid state fermentation (SSF) is an interesting process for lignocellulose-degrading enzymes production at low cost, due to the possibility of using agricultural and agroindustrial residues as substrate for microbial growth. Filamentous fungi are considered to be the most adapted microrgansims for SSF, since their hyphae may grow on the particles surface and penetrate the intra-particle spaces, and thus colonize solid substrates (Santos et al, 2004). In a general way, the demands of high water activity for the development of bacteria make them not adaptable to SSF processes. However, scientific reports have shown that some bacterial cultures may be adapted to this type of process (Nampoothiri & Pandey, 1996; Orozco et al., 2008).

Fig. 10 is a schematic representation of some biological and physical processes that take place during SSF. During a process of solid-state fermentation, fungi grow by forming a mycelial mat on the surface of the particles that comprise the solid substrate. Aerial hyphae protrude into the gaseous space and penetrative hyphae grow into liquid-filled pores. Unless the moisture is above the appropriate level, the empty spaces between the aerial hyphae are filled with gas (g), whereas the empty spaces within the mycelial mat and within the substrate are filled with liquid (l). Near the substrate surface and within the pores are regions of most metabolic activities. However, the regions of aerial hyphae also show metabolism and there can be a transport of substances from the penetrative to the aerial hyphae. By producing hydrolytic enzymes that diffuse into the solid matrix, fungi can degrade macromolecules into smaller units which are taken up by mycelium to serve as nutrients. During fermentation, there is consumption of O_2 and production of CO_2, H_2O, other biochemical products and heat. Gradients develop within the biofilm that, for instance, force O_2 to diffuse from the gaseous phase into deeper regions of the biofilm and CO_2 to diffuse from these regions to the gaseous phase. Heat is removed from the substrate via conduction and also by evaporation, which is part of the complex balance of water in the system. The release of organic acids and ammonia might change the local pH (Hölker & Lenz, 2005).

The use of SSF for fungi cultivation presents many advantages, such as: simulation of their natural environment, which results in better adaptation to the medium and higher enzymatic productions; reduction of bacterial contamination, due to absence of free water; obtainment of more concentrated enzymes, once the enzymatic extracts may be extracted with small amounts of water (Bianchi et al, 2001). Other advantages of fungal enzyme production by SSF can also be mentioned when comparing it to submerged fermentation, such as use of lesser amount of water and consequently a decrease in effluent generation, lower requirement of space and energy, more stability of the obtained products, more biomass and enzyme production, less catabolic repression of enzymes and little protein degradation (Pandey et al., 2000; Viniegra-González et al., 2003).

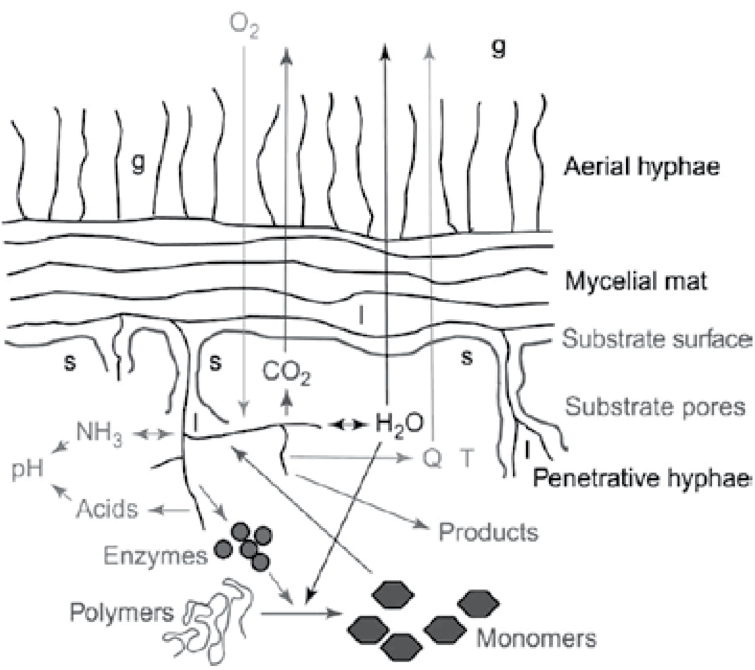


Fig. 10. A scheme of some microscale processes that occur during solid-state fermentation (SSF). Source: Holker & Lenz, 2005.

The main disadvantages of using SSF are the difficulty of heat dissipation. Temperature is directly related to water activity (*aw*) and aeration. The low thermal conductivity of the solid medium limits the removal of excess heat generated by microbial metabolism in SSF. In practice, SSF requires aeration mostly to dissipate heat rather than to provide oxygen (Viesturs et al., 1981). The temperature increase in bioreactors may cause protein denaturation, especially of the thermolabile ones (Santos et al., 2004). In this matter, thermophilic fungi encourage special interest for enzyme production processes by SSF. By exhibiting optimum growth at 45 and 55°C, such microorganisms better endure the possible temperature raise due to the difficulty of heat dissipation in SSF, especially in large-scale bioreactors, as well as secrete enzymes that are more active and stable at higher temperatures, which present many important properties from the industrial application point of view (Maheshwari et al., 2000).

Scale-up represents another obstacle to be overcome in SSF processes, once different gradients (moisture, temperature, substrate concentration and others) form along the bioreactor, which may negatively influence the process, especially the static bed ones. Heat dissipation, mass transfer and control of the fermentative parameters are the main challenges to be overcome in this sense (Hölker & Lenz, 2005; Khanahmadia et al., 2006). It is worth mentioning that, in the last decades, researches on design, modeling, operation and scaling up have brought progress to the knowledge of SSF processes involving bioreactors (Mitchell et al., 2000).

5. Lignocellulosic residues: Substrates for the production of microbial enzymes and source of fermentable sugars for second generation ethanol production

Agricultural, industrial, forestry and urban solid residues are abundant sources of lignocellulose available worldwide that, in the past, were treated as waste in many countries (Dashtban et al., 2009). In some countries, these materials are still treated as wastes, disposed in the environment, many times without an adequate treatment, raising environmental damage. In other countries, they are also used to generate thermal energy by traditional (firing for cooking and heating) or modern uses (producing electricity and steam, and liquid biofuels) (Dawson & Boopathy, 2007; Kim & Dale, 2004).

With the increasing expansion of agro-industrial activities, large quantities of lignocellulosic residues are generated annually worldwide (Sanchéz, 2009). In Brazil, according to the data cited by Ferreira-Leitão et al. (2010), the agroindustry of corn (13767400 ha), sugarcane (7080920 ha), rice (2890930 ha), cassava (1894460 ha), wheat (1853220 ha), citrus (930591 ha), coconut (283205 ha), and grass (140000 ha) collectively occupies an area of 28840726 ha and generates 597 million tons of residue per year.

Biotechnological innovations bring many significant and successful efforts to convert lignocellulosic residues into valuable products including enzymes for biotechnological and industrial applications (Leite et al., 2008; Tengerdy & Szakacs, 2003). However, the cost to produce these enzymes is still an obstacle that needs to be overcome. Therefore, the use of agroindustrial residues as carbon source for fungal growth and ligno/hemi/cellulolitic enzyme production through solid state fermentation (SSF) has been reported in many studies as a way of significantly reducing process cost. Many agricultural and agroindustrial by-products are used in SSF processes for the production of cellulases, xylanases and ligninases, among other enzymes (Table 1) (Gao et al., 2008; Kang et al., 2004).

As previously mentioned fungi are better adapted to SSF and are able to produce copious amounts of cellulases and hemicellulases which are secreted to the medium for easy extraction and purification. Their enzymes are often less complex than bacterial glycoside hydrolases and can therefore be more readily cloned and produced via recombination in a rapidly growing bacterial host, such as *Escherichia coli* (Maki et al., 2009). Besides, there are the white-rot fungi that significantly contribute to lignocellulose conversion since they are the best producers of the lignin degrading enzymes. Table 1 shows a few examples of lignocellulose-degrading enzymes produced by fungi in SSF.

Lignocellulosic residues can also be used to obtain biofuels like bioethanol, named second generation or cellulosic ethanol. New technologies promote the hydrolysis of the carbohydrate fraction of these residues into fermentable sugars that are converted into ethanol by fermenting microorganisms. This technology contrasts with the conventional first generation ethanol production, where carbohydrates of the edible portion (starch, sugar, oils, etc.) of food-based crops are used to be converted into ethanol (Balat et al., 2008).

Among lignocellulosic materials, crop residues are the major potential feedstock for second generation bioethanol production. Table 2 shows the composition of some crops and residues in terms of lignin and carbohydrates, besides the potential yield of second generation ethanol (Kim & Dale, 2004).

With the increase of world population and of the use of fossil fuels, together with constant climate changes, the search for sources of renewable energy has become more necessary, through the coordinated and sustainable actions, in environmental, social and economical aspects. So far, many countries have invested in research to introduce and ensure the prevalence of bioethanol on the market. In this context, Brazil stands out for having an energetic matrix with 46% of renewable sources, more than half of them represented by agroenergy, in a world where the use of these sources is of only approximately 15% It is the first country to regulate the mandatory addition of a certain percentage of ethanol into motor fuel. In 1975, the Brazilian government implemented the *Proálcool* (Alcohol National Program) for the large scale substitution of petroleum originated vehicle fuels, due to the petroleum crises in 1973 and 1979. This program greatly contributed to the country's alcohol and sugar sector development. Currently, sugarcane and its by-products are the second major source of primary energy of the national energetic matrix and the use of ethanol is already superior to that of gasoline. This experience has won over the world and Brazil has become reference in this area (Statistical Yearbook AgriEnergy, 2009).

In Brazil, first generation ethanol is produced from sugarcane broth, a readily fermentable material, once the substrate (sucrose) is directly used by the fermentative agent, *Saccharomyces cerevisiae*. In the United States alcohol fuel is produced from corn starch, meanwhile in Europe (except for France, which uses beet sugar) wheat and barley starch are mainly used. In both cases, enzymatic hydrolysis of starch is employed, and these countires have, therefore, industrial experience in the use of enzymes for bioethanol production (Bon et al., 2008).

Presently, Brazil ranks No. 1 in using first generation ethanol, obtained from sugarcane broth, with the highest rate in the world (Wang et al., 2011). The country's interest in developing biofuels has driven researches and investments for the diversification of the national energetic matrix. Therefore, the use of sugarcane bagasse (the fibers which are left after milling), a lignocellulosic residue widely available in the country, stands out as a new technology to be explored for the production of second generation ethanol.

Enzymes	Fungi	Substrates	Reference
Endoglucanase	*Trichoderma reesei* QM9414	Rice bran	Rocky-Salimi & Hamidi-Esfahani, 2009
	Aspergillus terreus M11	Wheat bran and straw, corn cob, reed straw, sugarcane bagasse	Gao et. al, 2008
	Myceliophthora sp. IMI 387099	Rice straw	Badhan et. al, 2007
	Fusarium oxisporum F3	Corn Cob	Panagiotou et al., 2003
	Aspergillus Níger 38	Wheat straw and bran	Jecu, 2000
β-glucosidase	*Aspergillus sydowii BTMFS 55*	Wheat bran	Madhu et al., 2009
	Thermoascus aurantiacus CBMAI 456 and *Aureobasidium pullulans* ER-16	Wheat bran, soy bran, soy peel, corncob and corn straw	Leite et al., 2008
Xylanase	*Neosartorya spinosa* D19	Wheat bran	Alves-Prado et al., 2010
	Aspergillus terreus	Palm waste	Lakshmia et al, 2009
	Coprinellus disseminatus	Wheat bran	Singh et al., 2009
	Aspergillus carneus M34	Soybean Shell, Rice bran	Fang et al., 2009
	Fomes sclerodemeus	Wheat bran	Papinutti & Forchiassin, 2007
Laccase	*Coriolopsis byrsina*	Wheat bran	Gomes et al., 2009
	Lentinus tigrinus BAFC 197	Wheat straw	Lechner & Papinutti, 2006
	Pleutorus sp	Coconut coir	Bhattacharya et al., 2011
	Pleurotus ostreatus MTCC1804 and *Anacystis nidulans* IU625	Peanut shell	Mishra & Kumar, 2006
	Trametes versicolor	Horticultural wastes	Xin & Geng, 2001
	Trametes versicolor	Sugarcane bagasse	Pal et al., 1995
Manganese peroxidase	*Lentinus* sp	Wheat bran	Gomes et al., 2009
	Fomes sclerodemeus		Papinutti & Forchiassin, 2007
	Pycnoporus Cinnabarinus	Wood	Alves et al., 2004
	Ceriporiopsis subvermispora	Wood waste	Ferraz et al., 2003
	P. chrysosporium	Wheat straw	Fujian et al., 2001
Lignin peroxidase	*Lentinus* sp	Wheat bran	Gomes et al., 2009
	Poliporus sp, *Hexagonia* sp	Wheat bran	Abrahão et al., 2008
	Pleurotus sajor-caju	Banana waste	Reddy et al, 2003
	Pleurotus ostreatus, Phanerochaete chrysosporium	sugarcane bagasse	Pradeep & Datta, 2002
	P. radiate	corn straw	Vares et al., 1995

Table 1. Production of enzymes related to the degradation of vegetable material by fungi cultivation in SSF, using agricultural residues as substrate.

Biomass	Residue/crop ratio	Dry matter[1]	Lignin[1]	Carbohydrates[1]	Ethanol yield[2]
Barley	1.2	88.7	2.90	67.10	0.41
Barley Straw		81.0	9.00	70.00	0.31
Corn	1.0	86.2	0.60	73.70	0.46
Corn stover		78.5	18.69	58.29	0,29
Oat	1.3	89.1	4.00	65.60	0.41
Oat straw		90.1	13.75	59.10	0.26
Rice	1.4	88.6		87.50	0.48
Rice Straw		88.0	7.13	49.33	0.28
Sorghum	1.3	89.0	1.40	71.60	0.44
Sorghum straw		88.0	15.0	61.00	0.27
Wheat	1.3	89.1		35.85	0.40
Wheat straw		90.1	16.0	54.00	0.29
Sugarcane		20.0		67.00	0.50
Sugarcane bagasse	0.6[3]	71.0	14.50	67.15	0.28

[1]Data in percentage; [2]Data in L/ Kg of dry biomass; [3]Kg of bagasse/Kg of dry sugarcane. (Kim & Dale, 2004).

Table 2. Composition of some lignocellulosic biomass (based on dry biomass) and potential ethanol yield.

The Brazilian production of sugarcane for sugar and alcohol obtainment was of 563,638,524 milled tons in the 2008/2009 harvest (approximately 78 ton/ha). Ethanol yield per sugarcane ton in this harvest season was of approximately 49 L or 3,528 L/ha (Statistical Yearbook AgriEnergy, 2009).

In gerenal, the processing of one sugarcane ton generates approximately 280 kg of bagasse. According to Macedo (2005), about 92% of sugarcane bagasse is used for heat process. If the remaining 8% were to be converted to ethanol, then one could expect an additional 2200 L of ethanol per hectare, bringing the ethanol yield per hectare to 5.728 L, reducing the land use requirements by 29% (Soccol et al., 2010).

Sugarcane bagasse represents the highest residue percentage of the Brazilian agroindusty, with an average of 75 million tons/year (Teixeira et al., 2007). Although a great part is used for thermal energy generation at the manufacturing plant, the excess bagasse represents a great opportunity for the national agro industry. Therefore, there is broadly recognized that the annual sugarcane bagasse production reaches considerable numbers and that the use of this residue is a national need, with wide space for the development of more noble activities than direct energy generation through combustion. According to a few authors, the use of the excess bagasse makes the investments needed to adapt the sugar and alcohol manufacturing plants economically feasible, solving the supplying problem of the sugar and alcohol industry, offering social and environmental advantages and increasing yield of the economical process (Gamez et al., 2006).

Sugarcane composition varies according to many factors, such as the extraction process, sugarcane variety and soil composition. Generally, it is made up of approximately 32-44% cellulose, 27-32% hemicellulose and 19-24% lignin (Malherbe & Cloete, 2002; Howard et al., 2003). This residue presents small amounts of other compounds such as pectin, protein, minerals and low molecular weight compounds, besides having lower ash content (approximately 2.4%), which represents a great advantage for its bioconversion by microorganisms when compared to other residues such as rice and wheat straw, which contain approximately 17.5 and 11% ash, respectively. In addition, when compared to other agro industrial residues, bagasse can be considered a rich reservoir of energy due to its high yield (approximately 80 t/ha, when compared to 1, 2 and 20 t/ha for wheat, grasses and trees, respectively) and annual regeneration capacity (Pandey et al., 2000).

In Brazil, the South-Central region is responsible for approximately 90% of the sugarcane production and designated an area of 7.01 million hectares for its plantation in 2008. In the São Paulo State alone, 3.82 million hectares (54.5%) were occupied with this crop. The sugar and alcohol sector in Brazil has been keeping a continuous expansion along the past years, which can be noted by the enlargement of sugarcane planted area, construction of new manufacturing plants throughout the national territory and by the production of ethanol in the country (Teixeira et al., 2007). Often, this expansion raises a number of questions regarding its competitiveness with food crops and the possibility of destruction or damage of high biodiversity areas, deforestation and degradation or damaging of soils through the use of chemicals. Studies have demonstrated that such concerns have been grossly exaggerated (Cerqueira Leite et al., 2009; Goldemberg & Guardabassi, 2010). Goldenberg et al (2008) reported that, according Instituto de Economia Agrícola – IEA, a large portion of Brazil has conditions to economically support agricultural production while preserving vast forest areas with different biomass. Most expansion on existing sugarcane crops is taking place on degraded and pasture lands (Lora et al., 2006), areas are far from important biomes like Amazon Rain Forest, Cerrado, Atlantic Forest and Pantanal. Goldenberg et al. (2008) also presented in their paper that sugarcane growth does not seem to have an impact in food areas, since the area used for food crops has not decreased between 1990 and 2006, according data from IBGE and IEA. Besides, the amount of fertilizer used is also small compared to sugarcane crops in other countries (48% more is used in Australia). Authors also consider that there is an increase on ethanol production in Brazil, especially in Sao Paulo State, due to the growth of overall productivity related to the development of new species and to the improvement of agricultural practices.

6. Pretreatments and enzymatic saccharification of lignocellulosic biomass aiming ethanol production

6.1 Types and effects of pretreatments

One of the major obstacles towards saccharification of the lignocellulosic material is its recalcitrant nature. Heterogenous characteristic of biomass particles, surface area and the presence of hemicellulose-lignin complexes covering cellulose contribute to the resistence of lignocellulosic biomass towards hydrolysis (Chang & Holtzapple, 2000).

Structural and constitutive factors reflect the complexity of the lignocellulosic material. The variability of these characteristics contributes to the differences of recalcitrance among lignocellulose sources (Mosier et al., 2005). The removal of lignin, the reduction of cellulose cristalinity and the increase of fiber porosity as a means to facilitate enzyme access may significantly improve saccharification of the lignocellulosic material (McMillan, 1994). This way, the bioconversion of lignocellulosic material into ethanol involves many stages which include: a) pretreatment (physical, chemical and/or biological); b) lignin depolymerization

and carbohydrate hydrolysis; c) conversion of fermentable sugars liberated through the hydrolysis in ethanol and d) ethanol separation and purification.
The objective of any technology related to the pretreatment of lignocellulosic material is to minimize or remove the constraints of hydrolysis, improving enzymatic hydrolysis rate, thus increasing the yield of fermentative sugars. Gererally, pretreatments alter the structure of lignocellulosic biomass, making it more accessible to the action of depolymerizing enzymes.
The success of pretreatments is based on a few factors such as: obtainment of fermentable sugars or of a facilitated subsequent obtainment of these sugars through enzymatic hydrolysis; no formation or low formation of products that may inhibit both enzymatic hydrolysis and the action of fermtative microorganisms; be a low cost technology (Sun & Cheng, 2002). Studies have shown that the pretreatment step defines at which extension and cost the carbohydrates will be converted into ethanol (Chandel et al., 2007). It should also be mentioned that lignin removal, during the pretreatment, is important due to the possibilities of its industrial use (Gopalakrishnan et al., 2010).
Table 3 shows a few examples of lignocellulosic biomass pretreatments. It should be mentioned that not all of these methods have been completely established in such a way as

	Processes	**Mechamism of changes on biomass**
Physical	Ball-milling; Two-roll milling; Hammer milling; Colloid milling; Vibro energy milling ; Hydrothermal; High pressure steaming; Extrusion; Expansion; Pyrolysis; Gamma-ray irradiation; Electron-beam irradiation; Microwave irradiation.	Increase in accessible surface area and pores size; decrease in cellulose crystallinity and polymerization degree; partial hydrolysis of hemicellulose and lignin depolimerization
Physicochemical and chemical	EXPLOSION: Steam explosion; Ammonia fiber explosion (AFEX); CO_2 explosion; SO_2 explosion. ALKALI: Sodium hydroxide; Ammonia; Ammonium sulfite. GAS: Chlorine dioxide; Nitrogen dioxide ACID: Sulfuric acid; Hydrochloric acid; Phosphoric acid; Sulfur dioxide. OXIDING AGENTS: Hydrogen peroxide; Wet oxidation; Ozone. CELLULOSE SOLVENTS: Cadoxen; CMCS; SOLVENT EXTRACTION OF LIGNIN: Ethanol-water extraction; Benzene- water extraction; Ethylene-water extraction; Butanol-water extraction; Swelling agents.	Delignification; decrease of cellulose crystallinity and polymerization degree; partial or complete hemicellulose hydrolysis
Biological	Actinomycetes and Fungi	Delignification; reduction of cellulose and hemicellulose polymerization degree

Source: Taherzadeh & Karimi, 2007.

Table 3. Pretreatment methods of lignocellulosic materials for enzymatic hydrolysis.

to be technically and economically feasible for large scale processes. In some cases, a method is initially employed to increase the efficiency of another method to be used afterwards. Besides, it should be taken in consideration that the choice of pretreatment has to be compatible to the subsequent hydrolysis step. Hydrolysis with diluted acids (gentle acid treatments) is probably the most widely used amongst the chemical pretreatments, and it can be used for direct obtainment of fermentable sugars, originated from hemicellulose fraction or as precedent of enzymatic hydrolysis (Taherzadeh & Karimi, 2007). One of the main advantages of this type of pretreatment is the high xylose yield, from xylan degradation, representing an important economy in the ethanol production process as a whole, given that approximately 90% of sugars from hemicellulose are this way recovered (Sun & Cheng, 2002). However, *S. cerevisiae,* the most used microorganism for industrial ethanol fermentation processes, and the majority of bacteria and other yeasts do not utilize D-Xylose. Thus, there has been a great emphasis in the last two decades on developing an efficient organism for xylose fermentation through metabolic engineering (Jeffries, 2006; Martín et al., 2007a).

In the pretreatment with diluted acids, cellulose and lignin fractions remain unaltered. On the other hand, some disadvantages are the need of pH neutralization for the following enzymatic hydrolysis process, high maintenance cost due to corrosion problems and possible generation of cellulase activity inhibitors, furfural and other condensation products generated from the rapid xylose degradation, which should be removed through biomass washing prior to enzymatic hydrolysis (Sun & Cheng, 2002). Cellulose, on the other hand, needs more severe hydrolysis conditions, considering its crystalline nature. Acid-catalyzed cellulose hydrolysis is a complex and heterogeneous reaction, involving physical and chemical factors following the mechanism of β-1,4 glycosydic bond cleavage (Xiang et al., 2003).

Pretreatment with diluted sulphuric acid generaly reaches high reaction rates and significantly provides celulose hydrolysis (Esteghlalian et al., 1997). In a general way, acid hydrolysis reactions may be represented by:

1st hydrolyze step

$$\underset{\text{Pentosan}}{(C_5H_8O_4)_n} + n(H_2O) \rightarrow \underset{\text{Pentose}}{(C_5H_{10}O_5)_n}$$

$$\underset{\text{Pentose}}{C_5H_{10}O_5} \rightarrow \underset{\text{Furfural}}{C_5H_4O_2} + 3(H_2O)$$

2nd hydrolyze step

$$\underset{\text{Pentose}}{(C_6H_{10}O_5)_n} + n(H_2O) \rightarrow \underset{\text{Hexose}}{n(C_6H_{12}O_6)}$$

The explosion, with or without the presence of vapor, is one of the most cited pretreatment methods for lignocellulosic biomass. Vapor explosion removes most of the hemicellulose, promotes lignin depolymerization, facilitating the access of cellulases to their substrate (Moniruzzaman, 1996) decreasing the amount of lignin available for non specific and unproductive bonding to cellulases and hemicellulases. In this method, biomass is treated with saturated vapor (160-260 °C) for a few seconds to minutes, under high pressure (0.69-4.83 MPa), which is abruptly reduced to atmospheric pressure, making the material go through an explosive decompression (Varga et al., 2004). Adding H_2SO_4 (or SO_2) or CO_2 to the vapor explosion process may significantly improve enzymatic hydrolysis, decrease the formation of inhibiting compounds and promote a more complete hemicellulose removal (Morjanoff & Gray, 1987). The advantages of vapor explosion pretreatment include the low energetic requirement, when compared to mechanical fragmentation, and the absence of environmental or recycling costs. This type of pretreatment is amongst the ones with lower cost for hardwood and agricultural residues (Clark & Mackie, 1987). Disadvantages of this method include incomplete dissociation of the lignin-carbohydrates matrix and formation of compounds that may have inhibitory effect on microorganisms used during the downstream processes (microbial growth, enzymatic hydrolysis and/or fermentation) (Mackie et al., 1985), which leads to the need of washing the pretreated biomass (McMillan, 1994). Washing decreases the overall yield of the saccharification process, once it removes part of the soluble sugars generated by hemicelluloses' hydrolysis (Mes-Hartree et al., 1988).

6.2 Enzymatic saccharification

The enzymatic hydrolysis of lignocellulosic biomass is advantageous when compared to the chemical conversion, once there is no substrate loss due to chemical modifications and moderate and non corrosive operational conditions may be used. In addition, it provides better yields since it does not generate collateral products. During lignocellulose saccharification, the use of auxiliary enzymes such as feruloyl esterases and ligninases is interesting, given that it facilitates the access of cellulases and hemicellulases to their substrates (Sandgren et al, 2005; Selig et al., 2008). Therefore, complex enzymatic preparations are more efficient for the degradation of lignocellulosic material than those that do not have the concerted action of these enzymes (Yu et al., 2005).

The different biomass pretreatment strategies and the optimization of the enzymes production processes have resulted in several-fold reduction of enzyme loading for lignocellulose hydrolysis (Merino & Cherry, 2007) but the efficient enzymatic conversion of cellulose and hemicellulose polymers is still the major bottleneck in the utilization of lignocellulosic residues for bioethanol production. Thus, the need of obtaining efficient enzyme mixtures, with the minimal number of key enzymatic activities, in an optimal combination is clear. Besides, the optimal reaction conditions and minimal dosage requirements must be known (Meyer et al., 2009).

With the cost reduction for enzyme production, caused by the use of alternative and inexpensive substrates such as the lignocellulosic residues or by the use of modern technologies for production optimization such as cloning and heterologous superexpression, enzymatic hydrolysis cost tends to become relatively low when compared to acid hydrolysis. Moreover, there is a possibility of recover and recycle of the enzymes bound to the residual substrate as well as the enzymes from reaction suspension.

One of the great disadvantages of the enzymatic hydrolysis of lignocellulosic material is related to the inhibition by the products of carbohydrate hydrolysis, especially when concerning enzymes from the cellulolytic complex, especially β-glucosidases, key enzymes for the complete saccharification of cellulose into glucose. The low activity of this enzyme or its inhibition by glucose (K_i of most β-glucosidases from typical cellulase-producing microorganisms is 1–14 mmol L^{-1} glucose) (Yun et al., 2001; Decker et al., 2000) results in celobiose accumulation, a potent inhibitor of cellobiohydrolases and endoglucanases, decreasing hydrolysis rates (Holtzapple et al., 1990; Kovács et al., 2009). Inhibition of the cellulases by hemicellulose-derived sugars has also been shown (Xiao et al., 2004).

The addition of a surplus of β-glucosidase activity is an alternative to overcome the competitive product inhibition of the β-glucosidases. However, the use of high concentrations of this enzyme to reduce inhibition problems does not seem advantageous, once it can significantly increase process cost. Another strategy is to screen for β-glucosidases with high glucose tolerance. β-glucosidases with K_i up to 1400 mmol L^{-1} have been reported (Decker et al., 2000) and these could be cloned into the cellulase-producing microorganisms to produce a more efficient enzyme mixture. The removal of sugars during hydrolysis by ultrafiltration or by employing the simultaneous saccharification and fermentation process (SSF), where the sugars produced during enzymatic hydrolysis are simultaneously fermented to ethanol, have also been reported as alternatives to overcome the problem of enzyme inhibition by the final products of carbohydrate degradation (Sun & Cheng, 2002; Jeffries & Jin, 2000).

It has been shown that ethanol also inhibits cellulases, although less intensely when compared to glucose (Holtzapple et al., 1990; Chen & Jin, 2006). Cellulases inhibition by ethanol follows a noncompetitive inhibition pattern for ethanol concentrations less than 4 M and, when the ethanol concentration is increased, the enzyme is denatured. Ethanol also interferes with enzyme (manly cellobiohydrolases) adsorption to cellulose and modifies the cooperative effect between cellobiohydrolases and endoglucanases (Ooshima et al., 1985; Holtzapple et al., 1990).

During the pretreatment, the lignocellulose degradation products and hemicellulose-derived monomeric sugars formed and released into the liquid fraction (prehydrolysate) have also been shown to inhibit enzymes activities, since the remaining solid fraction (amorphous cellulose and lignin) is absorbed with this liquid up to 60–90% of its total weight. Among these degradation products we can mention organic acids (acetic acid, formic acid, and levulinic acid), sugar degradation products (furfural from xylose and 5-hydroxymethylfurfural – HMF – from hexoses at high temperature and pressure) and lignin degradation products (vanillin, syringaldehyde, and 4-hydroxybenzalde-hyde) (Palmqvist et al., 1999a; Cantarella et al., 2004). However, the inhibition of enzymatic hydrolysis by these products has not been clearly elucidated (Jorgensen et al., 2007). It has been shown that washing the pretreated material results in faster conversion of cellulose due to removal of inhibitors (Tengborg et al., 2001). Ultrafiltration has also been used to remove sugars and other small compounds that may inhibit the action of the enzymes.

Another obstacle to enzymatic hydrolysis of lignocellulose carbohydrates is the possibility of unspecific adsorption of enzymes (both cellulases and hemicellulases) onto lignin particles or surfaces, mainly due to hydrophobic interaction and, possibly, due to ionic-type lignin–enzyme interaction. Actually, after almost complete hydrolysis of the cellulose fraction in lignocellulosic material, up to 60–70% of the total enzyme added can be bound to lignin (Lu et al., 2002). Therefore, cellulases with lower affinity for lignin could be explored in the development of new enzymatic complexes preparations (Berlin et al., 2005; Palonen et al., 2004).

The trouble of unspecific enzymes adsorption to lignin could be overcome by the addition of non-ionic surfactants like Tween 20 or Tween 80. It could also improve the hydrolysis rate so that the same degree of conversion can be obtained at lower enzyme loadings.106. Ethylene oxide polymers such as poly(ethylene glycol) (PEG) show a similar effect and it is advantageous due to its low cost (Kristensen et al., 2007). Besides the use of surfactants, other methods for desorbing enzymes have been developed, such as use of alkali, urea and buffers of varying pH (Otter et al., 1989).

As previously mentioned, recycling of the enzymes from the reaction suspension as well as from the residual substrates is an attractive way of reducing costs for enzymatic hydrolysis (Qi et al., 2011). The addition of fresh substrate could recover free cellulases in bulk solution by adsorption, due to the high affinity of these enzymes for cellulose (Castanon & Wilke, 1980). The new material retaining up to 85% of the enzyme activity free solution could then be separated and hydrolyzed in fresh media eventually with supplementation of more enzyme (Tu et al., 2007). Since β-glucosidase does not typically bind to the cellulosic substrate it cannot be reused and supplementation with this enzyme is required at the beginning of each round of hydrolysis in order to avoid the buildup of cellobiose and the subsequent end-product inhibition of cellulase (Lee etal., 1995; Tu et al., 2007).

Ultrafiltration has been cited as viable process capable of recovering all of enzyme components (endoglucanase, exoglucanase and β-glucosidase) after complete hydrolysis of the cellulose (Mores et al., 2001; Qi et al., 2011). Depending on the lignin content of the substrate, only up to 50% of the cellulases can be recycled using this approach. The saving is therefore low, taking into account recovery costs (Singh et al., 1991; Lee et al., 1995).

The denaturation or loss of enzyme activity due to mechanical shear, proteolytic activity or low thermostability should also be considered as limiting factors for hydrolysis. Besides, due to cellobiohydrolases processivity and strong binding to cellulose chain (by the catalytic site) obstacles can make the enzymes halt and become unproductively bound. Summarizes the factors that limit efficient cellulose hydrolysis (Jorgensen et al., 2007).

The range of toxic compounds generated during some types of pretreatment and hydrolysis of lignocellulosic materials, mainly with high temperature and pressure, under acidic conditions, can limit the rapid and efficient fermentation of the hydrolysates by the fermenting microorganisms, such as *Saccharomyces cerevisiae*. The inhibiting compounds are divided in three main groups based on origin: weak acids, furan derivatives and phenolic compounds. As mentioned above, furfural and HMF are formed from xylose and hexoses respectively and when they are broken down, they generate formic acid. HMF degradation also yields levulinic acid. Besides, the partial lignin breakdown generates phenolic compounds (Palmqvist & Han-Hagerdal, 2000).

Undissociated weak acids inhibit cell growth since they are liposoluble when undissociated and can diffuse across the plasma membrane. In the cytosol, dissociation of the acid occurs due to the neutral intracellular pH, thus decreasing the cytosolic pH (Pampulha &Loureiro-Dias, 1989) and cell viability. According to Verduyn et al. (1990), when fermentation pH is low, cell proliferation and viability are inhibited also in the absence of weak acids, due to the increased proton gradient across the plasma membrane, resulting in an increase in the passive proton uptake rate.

Studies have been reported that furfural is metabolized by *S. cerevisiae* under aerobic, oxygen-limiting and anaerobic conditions (Taherzadeh et al., 1998; Navarro, 1994; Palmqvist et al., 1999b). Furfural is reduced to furfuryl alcohol during fermentation, with high yields, and the reduction increases with the increasing of inoculum size and of specific growth rate in chemostat (Fireoved & Mutharasan, 1986) and batch cultures (Taherzadeh et al., 1998). At high furfural concentrations (above 84 $mmol.g^{-1}$) the reduction rate decreases in anaerobic batch fermentation, probably due to cell death (Palmqvist et al., 1999b). Aerobic growth is less sensible to inhibition by furfuryl alcohol in *S. cerevisiae* than in *Pichia stipitis* (Weigert et al., 1988; Palmqvist et al., 1999b). According to Palmqvist et al. (1999a) growth is more sensitive to furfural than is ethanol production. Indeed, at low concentrations of furfural (approximately 29 mmol/L) there is an invrease in ethanol yield. The authors reported that this probably occurs because the reduction of furfural to furfuryl alcohol, by NADH-dependent yeast dehydrogenases (which regenerates NAD^+) has a higher priority than the reduction of dihydroxyacetone phosphate to glycerol (which regenerates NADH). Thus the lower carbon consumption for glycerol production leads to an increase in ethanol yield.

Cell integrity is harmed by phenolic compounds, especially those of low molecular weight, since they affect the membrane ability to act as selective barrier and enzyme matrice (Heipieper et al., 1994). These compounds have a considerable inhibitory effect during the fermentation of lignocellulosic hydrolysates, by a not elucidated mechanism (Delgenes et al., 1996).

Another obstacle for the efficient enzymatic saccharification of lignocellulosic material is related to the cellulase recycling (turnover), since the absorption characteristics of these enzymes on lignocellulosic substrates have not yet been completely understood. The enzymatic degradation of cellulose is a complex process that occurs at the limit of solid/liquid phases, where the enzymes are the mobile components. When the cellulases act in vitro on the insoluble substrate, three processes occur simultaneously: (a) physical and chemical changes of cellulase at the solid phase (still not solubilized); (b) primary hydrolysis, involving the liberation of soluble intermediates from the surface of cellulose molecules that are in reaction and (c) secondary hydrolysis, involving the hydrolysis of soluble intermediates into others of low molecular weight and, finally, into glucose (MOISER; LADISCH; LADISCH, 2002).

In a general way, enzymatic hydrolysis rate of the lignocellulosic material rapidly decreases, with cellulose enzymatic degradation being characterized by a fast initial phase, followed by a slow secondary phase, which can last until all the substrate is degradated. This has been frequently explained by the rapid hydrolysis of the easily accessible cellulosic fraction, by strong enzyme inhibition, especially β-glucosidases, by the product and the low inactivation of absorbed enzyme molecules (Balat et al., 2008).

Cellulose is an insoluble substrate; the adsorption of the cellulases onto the cellulose surface is the first step in the initiation of hydrolysis. Therefore, the presence of CBMs is essential for fast and correct docking of the cellulases on the cellulose. Removal of CBMs significantly lowers the hydrolysis rate on cellulose (Suurnäkki et al., 2000).

7. Stratagies for second generation ethanol production

Saccharification of lignollulosic material and the conversion of sugars into ethanol may employ different strategies, carried out simultaneously or sequentially. In all cases, the pretreatment stage is of crucial importance to increase enzymatic conversion efficiency.

When enzymatic hydrolysis and alcoholic fermentation are carried out separately, the process is known as Separate (or Sequencial) Hydrolysis and Fermentation (SHF). In this case, the enzymatic hydrolysis of the carbohydrates and the subsequent fermentation of hexoses and pentoses are carried out in distinct reactors and they can be performed under their optimum conditions, which is an advantage of this strategy. However, SHF leads to the accumulation of the glucose derived from the hydrolysis of cellulose that can inhibit cellulases, affecting the reaction rates and yields. Besides, part of glucose is adsorbed in the solid residual material, lowering the sugar conversion (Soccol et al., 2010; Olofsson et al., 2008).

Enzymatic hydrolysis and sugar fermentation can run together, in a same reactor, as Simultaneous Saccharification and Fermentation (SSF), is faster and presents a low cost process since only one reactor is necessary and the glucose formed is simultaneously fermented to ethanol, which also avoid the problem of product inhibition associated with enzymes. The risk of contamination is lower due to the presence of ethanol, the anaerobic conditions and the continuous withdrawal of glucose. Pentoses fermentation can be performed in a separate reactor. One disadvantage of this strategy is relates to the different optimum temperature for enzymatic hydrolysis (45–50 °C) and alcoholic fermentation (28–35 °C) (Soccol et al., 2010).

The process called Simultaneous Saccharification and Co Fermentation SSCF, pentoses and hexoses conversion are carried out in the same reactor (Castro; Pereira Jr, 2010). Finally, in the Consolidated BioProcessing (CBP) a single microbial community produced all the required enzymes and converts sugars into ethanol in a single reactor (Lynd, 1996), lowering overall costs. Studies suggest that CBP may be feasible and the researches have focused on the development of new microorganisms adapted to this process, which has been a key challenge (Lynd et al., 2002).

8. Conclusions

The search for "clean technologies", using alternative feedstocks, in order to obtain products of industrial interest, save energy and reduce effluent production is economically advantageous and has been encouraged by environmental issues during the last years. Researches dealing with the use of lignocellulosic wastes in bioprocesses, specially for microorganisms cultivation and cellulases, xylanases, ligninases and other enzymes production, stand out. These enzymes have potential for various biotechnological applications and in recent years special attention has been given to the destructuring, hydrolysis and saccharification of lignocellulosic material in order to obtain fermentable sugars that can be converted into second generation ethanol by fermenting microorganisms. However, for an efficient conversion of lignocellulosic materials into products of industrial interest, some bottlenecks must be overcome. The search for microbial strains suitable for cultivation in large scale, producing enzymes with characteristics appropriate to the biotechnological processes to which they are intended is of great importance.

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18

Are WEEE in Control? Rethinking Strategies for Managing Waste Electrical and Electronic Equipment

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1. Introduction

Electrical and electronic equipment (EEE) that has come to its end-of-life (EoL) either by ceasing to function or ceasing to be of any value to its owners is commonly referred to as e-waste (Widmer et al., 2005). In the European Union (EU), these wastes are referred to as waste electrical and electronic equipment (WEEE). This chapter discusses two key themes critical to understanding and tackling the challenge posed by WEEE, namely: (i) four key issues that make WEEE a priority waste stream; and (ii) WEEE management practices in various countries and regions. Drawing on a comprehensive literature review and four case studies, we critically analyse and discuss the factors that influence the generation, collection and disposal of WEEE, specifically addressing the spatial and temporal interactions of these factors before an alternative approach to conceptualising and managing WEEE is proposed.

2. Importance of WEEE

Four key global issues make WEEE a priority waste stream, specifically: global quantities of WEEE; resource impacts; potential health and environmental impacts; and ethical concerns.

2.1 Global quantities of WEEE

The rate of discarded EEE is growing at an alarming rate, especially in OECD countries where markets are inundated with huge quantities of new electronic goods. As one of the fastest growing waste streams around the world (Dalrymple et al., 2007; Darby & Obara, 2005; Davis & Herat, 2008), a phenomenal growth in the amounts of discarded WEEE has been observed in various regions of the world (Ketai He, 2008; Nnorom & Osibanjo, 2008), attracting the attention of various governments, environmental organisations (Greenpeace, n.d.) and the scientific community. Increasingly short product lifecycles and rapidly advancing technology have led to huge volumes of relatively new electronic goods being discarded (Goosey, 2004). Although there is a paucity of reliable data, estimates place the amount of WEEE generated globally between 20-50 million tonnes annually (Greenpeace, n.d.; Ketai et al., 2008) although a recent estimate suggests ~40 million tonnes of WEEE are generated annually (Schluep et al., 2009). However, we believe this figure is highly unlikely (see Ongondo et al., 2011a) and almost certainly too low. Such large quantities of WEEE have focused attention not only on how WEEE is handled but also on why so much of it is generated and ways in which it can be prevented.

2.2 Resource impacts

WEEE has an enormous resource impact (Meskers & Hagelüken, 2009). Access to and availability of a number of raw materials key to the production of EEE is increasingly becoming important with world reserves of metals such as gold and palladium in fast decline and becoming more expensive (See EurActiv, 2009; Meskers & Hagelüken, 2009). Consisting of a mixture of various materials, WEEE can be regarded as a resource of valuable metals, such as copper, aluminium and gold. When these materials are not recovered, raw materials have to be extracted and processed afresh to make new products, resulting in significant loss of resources (Cui & Forssberg, 2003). Insufficient EEE is collected, part of which is exported to developing countries where it is largely not entering official recycling systems (Meskers & Hagelüken, 2009). When WEEE is not recycled, raw materials have to be processed to make new products resulting in a significant loss of resources (Bains et al., 2006; Bohr, 2007). In addition to the resources that are lost when WEEE is discarded without some form of materials recovery, a phenomenon known as stockpiling traps resources and prevents them from re-entering the materials/resource stream. Stockpiling, a practice especially common in the USA and various other countries, refers to the storing/hoarding of EoL EEE by consumers despite such devices being of little or no use to them (Li et al., 2006; Lombard & Widmer, 2005; Wagner, 2009).

2.3 Potential health and environmental impacts

When WEEE is disposed of or recycled without any controls, there are potentially negative impacts on human health. Containing more than 1000 different substances, many of which are highly toxic (such as lead, mercury, arsenic and cadmium), there are potentially serious health impacts if WEEE is not disposed of properly (Widmer et al. 2005). The open burning of plastics, widespread general dumping, exposure to toxic solders and other malpractices associated with improper dismantling and treatment of WEEE as observed in various developing countries, can result in serious health consequences (Mureithi & Waema, 2008; Natural Edge Project, 2006; Puckett et al., 2003; Widmer et al., 2005). Hence, serious concerns have been raised with regard to the export of WEEE from developed countries for treatment in Asian countries such as China and India, where the waste treatment operations utilized have in some cases lead to adverse health and environmental consequences. The heavy metals found in WEEE (such as lead) can contaminate drinking water by leaching into groundwater from sources such as landfills (Fishbein, 2002). It is estimated that about 70% of the heavy metals in US landfills come from WEEE (Puckett et al., 2003). The extraction of raw materials, and the goods made from them, may also entail environmental damage through mining, manufacturing, transport and energy use (Bains et al., 2006). Although effective recycling has a much lower environmental footprint than primary production, it is reported that the amount of WEEE recycled today is still low (Meskers & Hagelüken, 2009).

2.4 Ethical concerns

Two issues highlight the ethical concerns associated with WEEE. The first is the reported incidences of child labour in informal WEEE industries/handling, especially in some parts of Asia (Puckett et al., 2003; Shinkuma & Huong, 2009) and Africa. Secondly, the illegal shipments of WEEE from affluent countries to poorer developing countries that lack the facilities to properly treat such wastes is becoming more prevalent (Nnorom & Osibanjo, 2008; Puckett et al., 2003). The evidence suggests a close link between ethical malpractices in the handling of WEEE and the potential environmental and health impacts; it has been observed that WEEE collected from illegal shipments is often handled informally with very little regard to safety standards. Hence, prevention of illegal WEEE shipments could alleviate (but not necessarily eradicate) negative environmental and health impacts.

3. Brief overview of WEEE management strategies in selected countries

Various strategies and practices have been adopted by a few countries and regions to handle, regulate and prevent WEEE as a response to the above challenges posed by this waste stream. Most of these have been enacted via legislation specific to WEEE. These are briefly summarised below for selected countries.

3.1 Europe

In response to the large amounts of WEEE disposed within its borders every year, (~6.5 million tonnes), the EU enacted the so called WEEE Directive (Directive 2002/96/EC) which its Member States (MS) were to transpose as legislation in their respective countries. The extended producer responsibility (EPR)-based Directive obliges manufacturers to finance the takeback of WEEE classified in 10 categories from consumers and ensure their safe disposal. The legislation promotes individual producer responsibility (IPR), reuse, recycling and other forms of recovery in order to reduce the disposal of WEEE. In addition, it sets various annual targets for the collection, reuse and recycling of WEEE. Currently, MS are required to annually separately collect at least 4kg of household WEEE per person. Despite these efforts, the European Commission (EC) reports that only one-third of generated WEEE is collected and treated according to the stipulated procedures with prevalent exports to developing countries (Commission of the European Communities [CEC], 2008; Dalrymple et al., 2007; European Union, 2003).

3.2 Asia

Rapid economic growth in Asia has led to an increase in the quantities of WEEE generated in the region. Most of the WEEE generated from other parts of the world end up in Asian countries, especially in China (receives ~90%). There is no commonly agreed political strategy for managing WEEE in the region. However, various countries have or are in the process of ratifying WEEE specific legislation. To cope with the alarmingly large quantities of EoL products it receives and the attendant spontaneous illegal/informal and in some cases (potentially) harmful handling and treatment of WEEE within the country, China has recently legislated measures to cope with WEEE. Stockpiling of WEEE also occurs since people rarely dispose of their used EEE due to the perception that goods retain a residual value which might have future uses (Ketai et al., 2008; Li et al., 2006; Terazono, Murakami, et al., 2006; Y. Wang et al., 2009; Xinhua News Agency, 2010). Japan has legislation designed to tackle their 5 largest sources of WEEE: Televisions (TV); refrigerators; washing machines; clothes dryers; and air conditioning units. Specific recovery targets for reuse and recycling are stipulated by the legislation referred to as the home appliance recycling law (HARL). In addition, the law requires consumers to pay a recycling fee at the time of disposal (Aizawa et al., 2008; Zhang & Kimura, 2006).

3.3 Africa

African countries still lag behind when it comes to enacting legislation to deal with WEEE. This is despite well documented evidence showing that certain African countries have been the recipients of WEEE illegally exported from various affluent nations. It has been observed that informal collection, dismantling and recycling of WEEE is beginning to take shape in several countries such as Nigeria, Ghana and Kenya. However, the absence of infrastructure and appropriate collection and recycling services for WEEE is still a major challenge in addition to scarcity of data on amounts of WEEE generated. In South Africa, there is both informal and formal WEEE recycling with noticeable levels of recycling taking place (BAN, 2005; Dittke et al., 2008; Lombard & Widmer, 2005; Nnorom & Osibanjo, 2008; Rochat & Laissaoui, 2008).

3.4 North America

Both the USA and Canada lack WEEE specific federal legislation. However, a number of states in the USA have established some form of EPR regulations and takeback programmes to deal with WEEE including Maine, the first state to mandate producer responsibility. WEEE in the USA is mainly managed via municipal waste management services. As previously mentioned, a lot of WEEE is stockpiled rather than returned for reuse/recycling with ~24 million EoL computers and TVs destined for storage each year. In Canada, a national scheme for the collection of mobile phones, smart phones and similar devices exists although quantities of returned phones are still low (Canadian Wireless Telecommunications Association, 2009; Kahhat et al., 2008; Wagner, 2009).

3.5 Latin-South America

It is reported that penetration of EEE in a number of Latin-South American countries is reaching commensurate levels in industrialised countries. Formal recycling in some countries is still at its infancy although many others lack any such facilities. There is lack of political structure and logistical infrastructure to adequately handle WEEE. However, Brazil is currently the frontrunner in attempts to formulate policy on WEEE with Costa Rica the only country with specific WEEE legislation as of 2008. In Argentina, similar to countries in other developing economies, stockpiling of obsolete and broken products is common (Horne & Gertsakis, 2006; Silva et al., 2008).

3.6 Australia

Most of the WEEE generated in Australia is sent to landfills. In 2008, ~180 million WEEE items were destined for landfills. Until recently, the country lacked a national policy for dealing with WEEE. The end of 2009 saw the establishment of the National Waste Policy, a 10-year vision for resource recovery and waste management including a voluntary industry-led (but Government-supported) scheme for recycling TVs and computers. The scheme was scheduled to start operations in 2011, allowing householders to freely dispose of their EoL products. (Davis & Herat, 2008; Garrett, 2009; TEC, 2008).

Table 1 summarises WEEE generation and management practices in selected countries. For a thorough discussion on WEEE management practices in various countries see Ongondo et al. (2011a).

It is clear from the preceding discussions that strategies to effectively deal with WEEE have still not been perfected. Despite the efforts by various countries to deal with the challenge of

Country	Generation (tonnes/year)	Reported discarded items	Collection & treatment routes
Germany	1.1 million (2005)	Domestic WEEE	Designated collection points, retailers takeback
UK	940K (2003)	Domestic WEEE	Designated collection points, retailers takeback
Switzerland	66,042 (2003)	Diverse range of WEEE	National takeback programmes
China	2.21 million (2007)	Computers, printers, refrigerators, mobile phones, TVs	Mostly informal collection and recycling
India	439K (2007)	Computers, printers, refrigerators, mobile phones, TVs	Informal and formal
Japan	860K (2005)	TVs, air conditioners, washing machines, refrigerators	Collection via retailers
Nigeria	12.5K (2001-06)	Mobile phones chargers & batteries	Informal
Kenya	7,350 (2007)	Computers, printers, refrigerators, mobile phones, TVs	Informal
South Africa	59.6K (2007)	Computers, printers, refrigerators, mobile phones, TVs	Informal and formal
Argentina	100K	Excludes white goods, TVs and some consumer electronics	Small number of takeback schemes, municipal waste services
Brazil	679K	Mobile and fixed phones, TVs, PCs, radios, washing machines, refrigerators and freezers	Municipalities, recyclable waste collectors
USA	2.25 million (2007)	TVs, mobile phones, computer products	Municipal waste services; a number of voluntary schemes
Canada	86K (2002)	Consumer equipment, kitchen and household appliances	A number of voluntary schemes
Australia	-	Computers, TVs, mobile phones and fluorescent lamps	Proposed national recycling scheme from 2011; voluntary takeback

Table 1. WEEE generation and management in selected countries (compiled from Ongondo et al., 2011a)

WEEE, a lot still needs to be done to promote, in the first instance, prevention of WEEE, as well as reuse, recycling and safe treatment options (see Ongondo et al., 2011a). This situation calls for a global rethink in how WEEE is managed. A number of alternative approaches to managing WEEE have been proposed including the recast of the WEEE Directive which would require a stricter collection target of 65% of the average weight of products placed in the respective markets of EU countries in the two preceding years (European Union, 2008). Huisman & Stevels (2006) proposed a shift away from weight-based approaches to takeback and recycling targets (such as the WEEE Directive household collection target). In their view, targeting specific important materials with high environmental and economic values makes much more sense. For instance, although ~50% of a mobile phone is composed of plastics, the recovery of the embedded precious metals such as gold, copper and palladium should be prioritised and hence reflected in policy/legislation recovery targets.

In an effort to contribute to the debate on how the management of WEEE can be rethought, we propose an alternative but complementary approach to conceptualising and managing WEEE with regard to consumers' decisions about their EoL products. The strategy, discussed below, is complementary to existing WEEE policies and regulations and would be more useful at the operational level of managing WEEE. Hence, it could find application in a waste practitioner's policies and strategies for managing WEEE.

4. Rethinking strategies for managing WEEE

At the heart of this proposed approach to managing WEEE is a critical understanding of the interaction of the factors that affect the generation, collection and disposal of WEEE in space and time. Knowledge about these interactions can help policy makers in their decision making regarding takeback and disposal for specific products as opposed to applying a one-size-fits-all approach to managing all WEEE. This essentially means using different models to manage WEEE depending on the context (the nature of the interactions). The ultimate aim of understanding these interactions is to facilitate a closed-loop system for resources/raw materials use as opposed to a linear flow of resources (see Figure 1). This calls for the design of strategies and systems to manage WEEE that would maximise the recovery of resources with minimal ethical malpractices, health and environmental impacts. A review of the literature reveals a number of factors that determine whether and how a product becomes WEEE, if, how and when it is collected and its eventual disposal or lack of it. However, these factors are mostly discussed in isolation of each other. What is lacking is an analysis and discussion on the interaction of the factors, i.e. to what extent do they interact to affect decisions (from consumers, policy makers and other stakeholders' point of view) about EoL appliances? Secondly, what is the nature of those interactions? Are they similar in space and time? Which factors are more important than others and why? Similarly, little has been discussed about the effect of the factors on each other. For instance, does recyclability of a product dictate its takeback or do takeback options affect recycling options?

To illustrate the proposed alternative approach to conceptualising and managing WEEE, using case studies, the following section first discusses factors that influence the generation, collection and disposal of WEEE. Secondly, it groups and discusses those factors into similar variable groups. Finally, it critically discusses the interaction of some of the identified factors in relation to their influence on product EoL decisions.

4.1 Factors affecting the generation, collection and disposal of WEEE: Examples from the UK

In the following sections, four case studies highlighting the key factors that influence the generation, collection and disposal of WEEE are presented (subsequently known as W-KFs).

Although all the case studies are from the UK, the literature review reveals that the identified factors are individually generic to various other countries.

4.1.1 Case study 1: UK household WEEE collection network

The aim of this study, carried out in 2010, was to assess and evaluate the UK household WEEE collection network. The study utilised both primary and secondary data. The latter was sourced from online databases of the environment agency (EA) as well as Valpak, UK's only distributor takeback scheme (DTS). The aim of the DTS is to assist EEE retailers meet their compliance obligations as stipulated by the UK WEEE Regulations (Ongondo et al., 2011a). Primary data was collected from a broad national survey of UK's designated collection facilities (DCF) for WEEE. A total of 393 DCFs were invited to participate in the questionnaire of which 168 completed the survey. The results show that in the UK, there is both an enabling infrastructure for the collection of WEEE as well as a matching service provision.

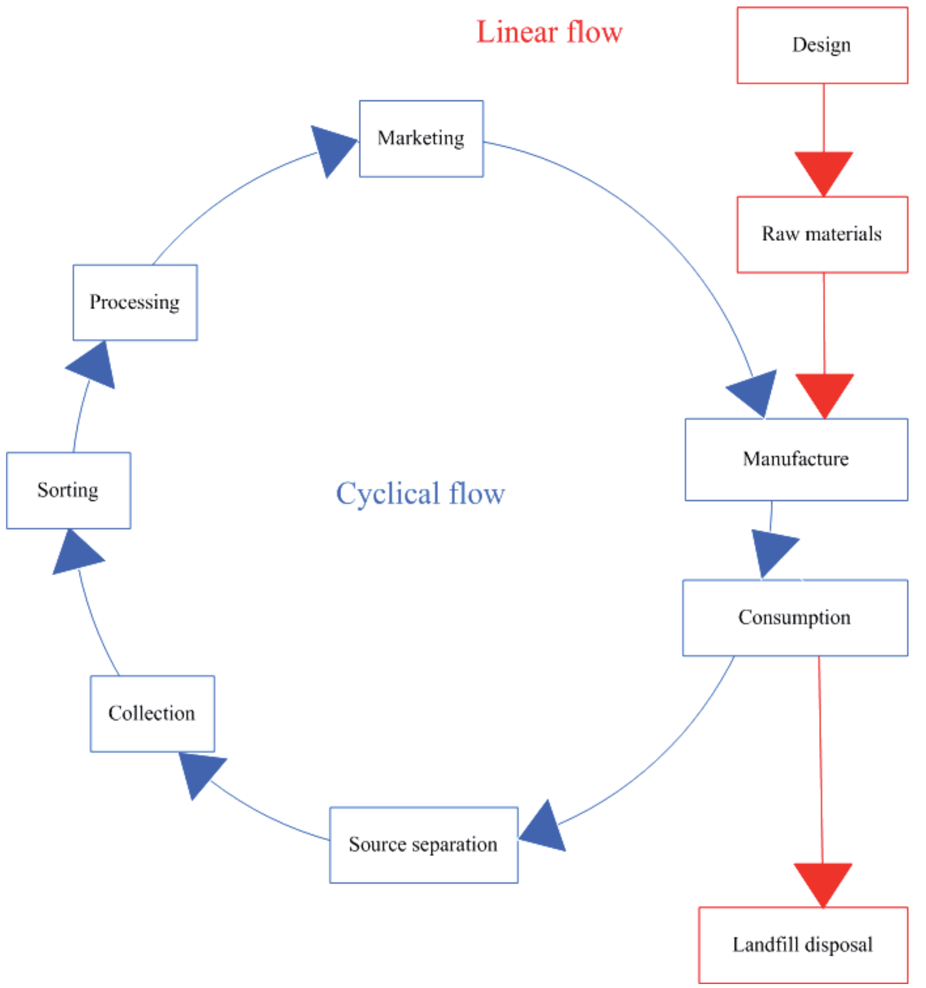

Fig. 1. Linear and cyclical resource flows (ZeroWIN, 2010)

The study revealed that in most urban areas in the UK there were adequate facilities for consumers to deposit their unwanted EEE. In general, the network is capable of collecting WEEE from 5 different streams (see Ongondo et al. 2011a) as stipulated by the UK WEEE Regulations. However, 26% of the DCFs did not have the capacity to collect all WEEE types. In addition, ~40% of the DCFs lacked sufficient storage space for collected WEEE (large appliances taking up a lot of space) whilst almost a quarter reported lack of awareness by the public to correctly separate WEEE at the DCFs.

Summary of the W-KFs identified from Case Study 1:

- Specific legal framework for handling WEEE;
- Infrastructure for collecting WEEE;
- Service for collecting and disposing of WEEE;
- Product size (problem with storing large appliances); and
- Public awareness about disposal service.

4.1.2 Case study 2: Mobile phone collection in the UK

An online survey to assess and evaluate the operations of UK voluntary mobile phone takeback services was carried out in Autumn-Winter 2008. Over 100 voluntary schemes operated by various organisations were identified. These promote their activities in various ways including newspapers, online and in-store. They also offer various incentives to encourage consumers to return their unwanted handsets to the takeback services for either reuse or recycling. The most common incentives are free collection of and monetary payments for returned devices. The takeback services offer various levels of convenience to enable the consumer to return their handsets in a hassle-free process. These include (in some cases) an easy to use online returns service, courier pick-up in cases involving 15+ phones as well as provision of appropriate envelopes/bags for returning unwanted phones. Most services use the available postal services infrastructure to collect phones from consumers. Central to the business model of these voluntary services are the residual reuse and recycling values of mobile phones.

Summary of the W-KFs identified from Case Study 2:

- Infrastructure for collecting WEEE (virtual [Internet] and physical);
- Service for collecting and disposing of WEEE;
- Product size (mobile phones are generally small and easy to transport);
- Product reusability and recyclability;
- Incentives to encourage takeback;
- Convenient takeback services; and
- Public awareness about collection service via promotion and advertising.

4.1.3 Case study 3: Consumer attitudes and behaviour toward use of mobile phones

This study involved a large scale online survey of 2287 students at 5 universities in South-East England carried out between November 2008 and August 2009. The aim of the research was to assess the behaviour of university students with regard to their use and disposal of mobile phones. The findings indicate that many students replace their phones at least once a year with male students replacing phones more often than females. The most common reasons advanced for changing mobile phones were replacement of broken phones (~58%) followed by upgrade phones offered by mobile phone network providers (~40%). Other reasons given were a desire to have a phone with a longer battery life (~18%) and fashion trends (~16%). Despite most students' awareness of takeback services for mobile phones (69%), in total, most of the phones replaced by students (~60%) are not sent to reuse/recycling services but are stockpiled. This equates to almost 3.7 million phones stockpiled by students in UK higher education (comparatively, 29.3 and 28.1 million stockpiled respectively for Europe and USA) (see Ongondo & Williams, 2011a). The most cited reason for stockpiling phones was "keep as back-up phone" (~78%) followed by "I don't know what to do with the phone" (~30%). Monetary incentives were found to elicit the greatest influence on students' willingness to recycle their unwanted mobile phones, followed by convenience of the takeback system and its ease of use.

Summary of the W-KFs identified from Case Study 3:

- Use and disposal attitudes and behaviour;
- Influence of gender on use and disposal of a small EEE;
- Service for collecting and disposing of WEEE;
- Product size (mobile phones are generally small and easy to store);
- Product durability (lifespan);
- Product reusability and recyclability;
- Incentives to encourage takeback;
- Promotion by retailers;
- Fashion;
- Convenient and easy to use takeback services; and
- Awareness about takeback services.

4.1.4 Case study 4: WEEE arising from one-off large scale events

Similar to countries in the EU and the USA, the UK enacted a policy to switch to digital-only TV by 2012. The policy, referred to as the digital switchover policy (DSO), would see all TV regions in the UK switch off their terrestrial analogue TV signals in favour of solely digital ones. In relation to this, the aim of this case study was to assess the potential logistical, financial, ethical and environmental impacts of the DSO on UK's network for collection of household WEEE. The Hampshire County Council in the South-East of England was used as a case study. Two public surveys (postal and structured street interviews) of Hampshire residents were carried out (319 respondents) in 2009. It was found that majority of residents (~98%) were aware of the DSO although only a moderately lower number (~67%) were aware of when the event would actually take place. The findings also showed that people on low-incomes and the unemployed were more aware about the date of the event in comparison to those on higher incomes and in employment respectively. The results showed that the DSO had the potential to generate large quantities of TV and related equipment WEEE (see Ongondo et al., 2011b). Whereas residents indicated their intention to dispose of their unwanted TVs via the established networks, smaller TV-related items such as remote controls and aerials would not warrant a trip to the household waste recycling centres (HWRCs); these were most likely disposed in the general waste. Residents also indicated they would keep their video cassette recorders (VCRs) despite their technological limitations once the switchover took place (see Ongondo, et al., 2011b). In addition, more males than females were aware that the capabilities of their VCRs would be affected by the switchover.

Summary of the W-KFs identified from Case Study 4:

- Impact of policy and technological changes;
- Use and disposal attitudes and behaviour;

- Influence of gender on use of EEE (VCR limitation issue);
- Service for collecting and disposing of WEEE;
- Product size (smaller items disposed of in general waste; larger ones taken to recycling centres);
- Effect of economic status on awareness regarding important events affecting EEE; and
- Public awareness about takeback services.

4.2 Factor groups

The evidence summarised in Section 4.1 indicates that factors that affect the generation, collection and disposal of WEEE can be grouped into at least 3 broad categories:

- Consumer variables: *Attitudes, behaviour, perception, values, awareness levels, age, gender, employment status, storage space, etc.;*
- Takeback system variables: *Infrastructure, service provision, convenience and ease of use of takeback system, incentives/disincentives, promotion and advertising of takeback options (awareness);* and
- Product variables: *Product type, size, quality (condition), quantity, material composition, reusability and recyclability.*

In addition, a number of factors external to the consumers' immediate decision making scope may influence the generation, collection and disposal of WEEE. These include:

- Policy: *Regulations, legislation, guidelines;*
- Technological change: *Emergence of new technologies such as digital TVs;*
- Market forces: *Fashion, retailer promotions, etc.;*
- Costs: *Cheaper/affordable products, etc.;*
- Product EoL; and
- Social need/pressure: *Peer influence, etc.*

These factor groups are individually discussed in the succeeding sections followed by an analysis of how they interact with each other and the likely outcomes of those interactions.

4.2.1 Consumer variables

Consumer variables such as age, gender, culture, perceptions and attitudes affect both the generation and disposal of WEEE, although globally there are disparities between countries regarding the effect of these variables. Whereas in some parts of Asia and Africa EoL products are rarely thrown out, in some parts of Europe the opposite is true. In the former, perception of what is waste is very different from attitudes in Europe. Possibly, this is largely shaped by the differences in affluence in these regions. In Asia, South America and Africa, it is not uncommon for EoL products to find secondary uses. For instance, a broken refrigerator would find use as a cupboard. Hence, stockpiling of WEEE is generally common in Asia compared to Europe. Although the same phenomenon occurs in the USA, the reasons behind the practice are different. The key consumer variable in the USA is the availability of space to store unused and EoL EEE as well as the lack of (affordable) takeback services for the devices.

4.2.2 Takeback system variables

Takeback services are an integral part of the management of WEEE. However, the existence of such services is neither a guarantee that WEEE will be collected nor disposed of properly. In addition, the nature of the takeback services may influence the type and amounts of

WEEE collected. Important variables in takeback services include infrastructure, formal or informal operations, incentives/disincentives offered by the services, awareness about the services and the number and types of products collected. This means that takeback services range from the "fairly straightforward" (e.g. takeback of mobile phones) to complex systems such as the takeback of different WEEE within the EU. Due to the many variations in the interplay of these variables, the logistics of designing and implementing a takeback system for WEEE are complex.

Perhaps the basis of any takeback system is an enabling infrastructure (see Timlettt & Williams, 2011). As the evidence in Case Study 2 suggests, this does not have to be specific infrastructure established for takeback of WEEE since piggy-backing on existing infrastructure may be a viable option. Lack of infrastructure is a primary limiting factor to the takeback of WEEE, as highlighted in Sections 3.3 and 3.5. Equally important in a takeback system is the provision of a service to collect WEEE for reuse or recycling from consumers. As demonstrated in Australia, although there is infrastructure in place to collect WEEE, the deficiency of a matching reuse and recycling service and policy has resulted in vast amounts of WEEE deposited in landfills. An additional issue in WEEE takeback is competition among takeback services. In China, competition between informal and formal takeback and recycling of WEEE curtails the operations of the latter. In some cases, this has lead to serious health and environmental consequences resulting from informal WEEE recycling activities (see Ongondo et al., 2011a).

A common thread in takeback systems is the influence of incentives on consumer willingness to return unwanted products. Although there is very strong evidence (see Ongondo & Williams, 2011a) suggesting incentives, especially monetary ones, positively influence consumers to return their WEEE, offer of incentives varies by region and by product. Monetary incentives are generally offered for products with a residual reuse value such as mobile phones (see Ongondo & Williams, 2011a; 2011b). On the contrary, in some countries such as Japan, despite the existence of takeback/recycling fees, many consumers still return their unwanted products using the official takeback schemes. This is conceivably related to the culture/attitudes of the people in that country since in the USA the suspicion is that recycling fees encourage stockpiling (see Ongondo et al., 2011a). In Case Study 4, it was established that one of the reasons given by UK consumers for throwing WEEE in the general waste is that the size of the WEEE does not warrant a trip to the HWRC. This raises an important issue about the levels of convenience and accessibility that takeback systems should offer balanced against the level of responsible behaviour that consumers should display.

4.2.3 Product variables

The influence of product variables on the generation, collection and disposal of WEEE cannot be underestimated. At the basic level, attributes such as size, type and quantity have a bearing on these 3 aspects of WEEE. Case Studies 1 and 4 illustrated that sizes and quantities of WEEE also have a bearing on how the waste is handled. For instance, the potentially large quantities of TV WEEE generated by the DSO would necessitate careful and strategic planning to ensure the takeback system would effectively handle the waste arising. As exemplified in Section 3 as well as Case Studies 2 and 3, product reusability dictates what happens to WEEE at its EoL. On average, devices with higher reusability value such as mobile phones and computers will rarely be thrown away compared to other WEEE with lower reusability value such as TV remote controls, toasters and hairdryers.

4.2.4 External factors

Policy decisions have the potential to trigger large scale generation of WEEE. For instance, the DSO policy in the UK and other countries will lead to the generation of substantial amounts of waste TVs and related equipment. Without adequate plans in place, such decisions may lead to a strain on existing infrastructure to cope with the sudden influx of WEEE. This was the case in 2001 in the UK when the Regulation on Ozone Depleting Substances was enacted. The ban on the export of refrigerators containing such substances led to the build up of thousands of refrigerators (so-called "fridge mountain") at local councils' civic amenity sites (Florence & Price, 2005). This highlights the importance of policy on management of WEEE and available infrastructure being in tandem.

Policies such as the WEEE Directive can also lead to undesirable negative effects such as the illegal export of WEEE to countries without the capacity to properly handle such wastes (see CEC, 2008).

In the case of the DSO, changes in policy were necessitated by advances in technology. However, as a separate entity, changes in technology or new technology can in themselves lead to the generation of WEEE. Some of the effects can be gradual, for instance, the replacement of old technology, such as the adoption of digital TVs and the subsequent replacement of analogue ones, or exponential, for instance, the rapid uptake of mobile phones especially in developing countries with the subsequent replacement of land line telephones. An example of the latter case in Nigeria is discussed by Nnorom & Osibanjo (2008).

Social pressure (e.g. fashion), affordability and market forces such as advertising can exert influence on consumers to give up their perfectly functional EEE in favour of newer technology (see Ongondo & Williams, 2011a), a situation that can be referred to as "perceived obsolescence". For certain products such as mobile phones, perceived obsolescence may be of benefit, for instance, the reuse of unwanted handsets in secondary markets such as export to developing countries or local second-hand markets. However, in order to assess the true worth of this apparent benefit, the costs of acquiring raw materials to manufacture new technology would need to be weighed against the benefit of reusing "old" technology. In some cases, the evidence suggests that perceived obsolescence is not always beneficial as in some countries such as the USA and Australia it has led to stockpiling and massive landfilling of WEEE respectively. On the other hand, it can be argued that second-hand products allow less economically endowed members of the society access to technology.

Naturally, all EEE have a specific lifespan after which they are expected to reach their EoL and become WEEE. An important point question raised by Ongondo & Williams (2011a) was whether manufacturers of EEE deliberately design their products to have short lifecycles (although technically they could last longer) in order to gain financially from the purchase of replacement products. The authors gave the example of a mobile phone; from a technical point of view, handsets have a lifespan of 10 years. On the contrary, the evidence suggested that most phones are replaced since they get damaged well before their 10th anniversary. However, the authors posited that the proposition that EEE manufacturers intentionally design products with short lifecycles was not conclusive since it was possible that consumer lifestyles could contribute to the shorter lifespans of the devices.

4.3 Interaction of factors influencing WEEE generation, collection and disposal

At the heart of this proposed alternative approach to handling WEEE is the interaction of factors that influence the generation, collection and disposal of WEEE. In this section, a few examples illustrating the nature of such interactions will be discussed.

A decision that policy makers and designers of takeback schemes may need to make is whether reuse/recycling options dictate takeback services or if the opposite is true, i.e. takeback services dictate what reuse/recycling options should be provided. In the former case, only WEEE that has capacity to be reused or recycled would be catered for via a takeback system. An example of this is mobile phone takeback programmes. Before mobile phones were invented, the infrastructure for their takeback already existed, for instance, courier and postal services. However, no takeback services existed until reuse/recycling options for mobile phones were "discovered". When reuse/recycling options were created, matching services for the returns of the devices were started. This is true for both formal reuse/recycling operations such as those found in USA and Europe and informal ones such as those observed in Africa. For the latter case, where the existence of a takeback service shapes the reuse/recycling options, an example is the previously mentioned "fridge mountain" experience in the UK in 2001. A service for collecting bulky waste via local council civic amenity (CA) sites already existed that led to the system collecting a vast amount of EoL refrigerators which contained such banned substances. What were lacking in this case were complementary reuse/recycling options to deal with the "contaminated" WEEE. The end result was a huge pile-up of EoL refrigerators at CA sites (see Ongondo et al., 2011b). Similarly, it is reported that in Japan, takeback services for specific types of WEEE have influenced manufacturers to tailor their recycling operations to match the types of WEEE coming through the takeback system (Aizawa et al., 2008). Hence, these examples demonstrate that either of these factors can influence the other and no one approach is superior to the other.

The interplay of product variables and other consumer variables such as attitudes, perceptions, storage space and other geographical regional differences strongly dictate how WEEE is produced and how it is managed. To illustrate, consider the case of a refrigerator which from a product point of view is relatively bulky in size. In the UK, a refrigerator that has come to its EoL would most likely be disposed of via a HWRC or paid-for retailer takeback since the consumer would not have enough room to store it in their residence. In the USA, there is a high chance that the product could be stockpiled in a garage since, generally, space constraints would not be a hindrance. In Australia, it would probably be disposed of via landfill whereas in South Africa, the product could be stockpiled due to the perception that it retains a residual value despite the household space limitations. In the case of smaller WEEE, such as mobile phones and portable music players, their size (easy to store) and the residual values (monetary, sentimental, etc.) attached to them mean that at their EoL (which for mobile phones may incorporate upgrading or replacement by a more fashionable model) they would probably be stockpiled. Similarly, Case Study 1 highlighted that in the UK smaller items would most likely not warrant a trip to the HWRCs. The same is probably true of items such as toasters, hairdryers and irons. In this case, although the products are relatively small in size, their lack of appreciable residual value would probably see them end up in the general refuse. Due to the previously discussed (cultural) reasons, the same may not be true in certain poorer regions of Asia and Africa; such WEEE would most likely be hoarded. In the case of medium sized products with a high residual value such as computers, stockpiling of the products would most likely occur in the USA (due to availability of space), Africa, South America and many Asian countries (due to culture, perceptions and economic status) whereas in affluent Europe the products would most likely be donated to charities (e.g. in the UK) for reuse. Passing on the items to relatives and friends would also be a likely scenario in all these regions/countries.

5. Discussion

The fact that WEEE is the fastest growing waste stream in many countries and regions is incontrovertible. Similarly, the issues that make WEEE a global priority are beyond refute. Although various countries/regions have taken positive steps to deal with the challenges posed by WEEE, the desired outcomes – prevention and minimisation of WEEE; increased reuse, recycling, recovery of EoL EEE; deterrence of illegal exports; and minimisation of negative environmental and health risks are still not occurring at palpable levels.

Drawing together the findings from the literature review and the case studies as well as the discussion in Section 4 reveals a crucial point; the generation, collection and disposal of WEEE is a product of the interface of various factors whose nature of interactions varies over space and time. This supports the argument that factors affecting the generation, collection and disposal of WEEE should not be considered in isolation. The extent to which the factors interact and the nature of those interactions and how they affect decisions (from consumers, policy makers and other stakeholders' point of view) about EoL appliances vary by country, over time and by product type. On the issue of geographical variations in how WEEE is generated and handled, EoL products are rarely thrown out in developing nations of Asia and Africa in comparison to the more affluent societies. In the case of the former, cultural attitudes and limited incomes influence the consumers to stockpile their WEEE (consumer variables) whereas in the latter, a throw-away culture, better incomes (consumer variables) and availability of takeback services (takeback system variables) drive the consumers to dispose of their WEEE. The DSO (Case Study 4) in the UK highlighted the temporal nature of some factors that interact to affect EoL decisions. The DSO has the capacity to generate large amounts of TV WEEE thereby possibly necessitating, albeit temporarily, increased returns services of such WEEE. In this case, takeback system and product variables (TV sizes, types, etc.) assume an important role for a limited time. Similarly, the "fridge mountain" experience in the UK serves as an example of an event with temporal influence over the generation, collection and disposal of WEEE.

Due to the nature and interactions of the influencing factors, it is not possible to generically conclude which factors are more important than others since spatial, temporal, consumer, takeback services and product variabilities as well as other external dynamics dictate which factors are significant on a case by case basis. However, on a broad geographical basis, the findings by Ongondo et al. (2011a) serve as a reasonable (though not unequivocal) basis for conclusions about which factors are most important in influencing the generation, collection and disposal of WEEE in various regions. These factors are summarised in Table 2 under their respective factor/variable groups. Although the factors are generic to the respective regions, differences (in some cases significant) at the country level should be expected, for instance, the case of Japan versus poorer countries within Asia.

All these factors taken together clearly demonstrate the complexities involved in the management of WEEE.

6. Conclusion and recommendations

This chapter has discussed the global challenge of managing WEEE and illustrated why it should be regarded as a priority waste stream. Generally speaking, on a global and country level, where available, the desired outcomes of systems and strategies designed to manage WEEE have not been fully achieved; the quantities of WEEE generated remain high and the policies to successfully tackle the waste stream are either largely inexistent in many parts of the world or ineffective. The current situation calls for a rethink on how WEEE is managed. Hence, an alternative strategy to rethink how WEEE is managed has been proposed and discussed.

Central to the proposed approach is a critical understanding of the factors that influence the generation, collection and disposal of WEEE and how these factors interact both spatially and temporally. We have identified so-called W-KFs that have been classified into four distinct groups; consumer, takeback system and product variables as well as external factors.

On a factor group level, it has been shown that consumer variables such as age, gender, culture, attitudes, etc. influence the use and disposal of EEE leading to various disposal outcomes. Consumers' perceived value of products such as mobile phones has led to stockpiling of large amounts of the devices. Likewise, in poorer regions of the world, it has been observed that WEEE is rarely thrown out due to the perception that the equipment has some residual value.

Infrastructure, both physical and virtual (e.g. the Internet), and service provision (both formal and informal) are key factors that influence the collection and disposal of WEEE. Lack of infrastructure is a primary limiting factor to the takeback of WEEE as typified in developing countries. On the other hand, the existence of infrastructure and related service provision for takeback is no guarantee that WEEE will be collected nor disposed of properly. This was illustrated in Case Study 4 where it was established that some householders in the UK would bin small items of WEEE despite the existence of a collection system. In addition, the nature of the takeback services in terms of awareness about the service, the level of convenience and ease of use it offers consumers as well as incentives offered to encourage returns of WEEE influences the type and amounts of WEEE collected. Due to the many variations in the interplay of these variables, the logistics of designing and implementing a takeback system for WEEE are complex.

Product variables such as size, quality, quantity and reusability dictate what happens to EEE at its EoL. In some cases, consumers considered some WEEE too small to warrant a trip to the established WEEE collection centres. On average, devices with higher reusability value such as mobile phones are either stockpiled at their EoL or sold for their monetary residual value (typically for reuse).

With regard to external factors, the review of WEEE generation and management practices in selected countries and regions showed that emergence of new technologies is an important factor that influences the generation of WEEE across geographical borders. The example given of the abandonment of fixed telephone equipment in Nigeria in favour of mobile phone devices is a testament to this.

Policy, or the lack of it, is a key driver affecting the generation, collection and treatment of WEEE. Policy includes regulations/legislation and related management principles such as EPR. The DSO policy in the UK and other countries is an example of a policy that leads to the generation of WEEE. In the EU, the EPR-based WEEE Directive and its enactment in MS exemplifies the effect of legislation on the collection of WEEE. Conversely, lack of legislation in many other countries has meant that WEEE is not collected and/or disposed of properly. However, it was also established that collection and treatment networks for WEEE can exist despite lack of legislation, for instance, the informal WEEE management practices in China, Kenya, etc. and the voluntary mobile phone collection networks in Australia and the USA.

Factors that influence the generation, collection and disposal of WEEE do not operate in isolation but interact to influence end-of-use/EoL decisions and outcomes for EEE. More important than the individual factors themselves, the significance of how the factors interact in space and time to influence the generation, collection and disposal of WEEE have been critically discussed. Recognising the nature of these interactions is crucial to the management of WEEE.

	Consumer variables	Product variables	Takeback system variables	External factors
Africa	Perceived residual value, limited incomes	Product reusability/secondary uses	Lack of takeback services, infrastructure and proper treatment facilities	Lack of legislation
Asia	Perceived residual value, limited incomes	Product reusability/secondary uses	Lack of takeback services, infrastructure and proper treatment facilities (with notable exception of Japan)	Lack of/weak legislation
Australia	Cultural norms (throw-away society), higher incomes	Product reusability (primarily in the case of mobile phones)	Lack of takeback services	Lack of/weak legislation, technological change
Europe*	Storage limits, cultural norms (throw-away society), higher incomes	Product reusability (primarily in the case of mobile phones), material composition	Established takeback services and infrastructure	Stringent legislation, technological change
Latin - South America	Perceived residual value, limited incomes	Product reusability/secondary uses	Lack of takeback services, infrastructure and proper treatment facilities	Lack of legislation
North America	Large storage spaces (limits collected amounts), cultural norms (throw-away society), higher incomes	Product reusability (primarily in the case of mobile phones)	Lack of/limited takeback services	Lack of/weak legislation, technological change

*Europe- mostly the EU and other affluent European countries.

Table 2. Key factors influencing the generation, collection and disposal of WEEE in various regions (adapted from Ongondo et al., 2011a)

Despite the potential inherent challenges and limitations of this proposed approach to managing WEEE (such as a clear understanding of relevant factors, hence need for access to data), this alternative way of thinking offers a novel approach to contextualise the genesis of WEEE generation and how it is collected and disposed whilst offering insights on how to rethink strategies to best manage it. The approach fits into the idea of a closed-loop system for the management of WEEE since it promotes the design of systems and strategies to recover different types and volumes of WEEE (see Guide & Van Wassenhove, 2009). We propose that recognition of the factors that influence the generation, collection and disposal of WEEE and their interactions is crucial in decision making when designing systems and strategies for the management of WEEE.

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19

Preliminary Study of Treatment of Spent Test Tubes Used for Blood Tests by Acidic Electrolyzed Water

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1. Introduction

Test tubes are widely used in medical facilities, for example, for collecting blood specimens of patients undergoing health checkups. Plastic-made and disposable tubes are increasingly replacing glass-made tubes, owing to the fact that they are convenient and hygienic. Because of the increase in the population of senior citizens in Japan and the increase in people's interest in their health, the amount of used test tubes will be much higher in the future. In Japan, recycling of medical waste is not a common practice, but there has been some research on medical waste management (Kagawa et al., 2006; Tamiya, 2004, Yamaguchi et al., 2002). Recycling of medical waste is gaining increasing popularity abroad, and it continues to attract the attention of researchers (Kushida, 2000; Bohlmann et al., 2005; Lee et al., 2002; Bartholomew et al., 2002). Test tubes used for blood tests are mostly made from polyethylene terephthalate (PET). In 2005, the total domestic demand for PET resin was 544,500 tons (Editorial Office of Monthly the Waste, 2006). Materials made of PET can be sold at a high price in the market; consequently, recycling industries in Japan are finding it increasingly difficult to source used PET materials. China in particular has a high demand and pays a good price for PET materials: Japan exported 338,000 tons of PET to China in 2009 (The Council for PET Bottle Recycling, 2010).

Incineration has been the main treatment method for PET tubes; however, social consensus against dioxins discourages incineration. Heating treatment followed by direct disposal is another option for treating the tubes, but this option is not reliable since complete inactivation of pathogens in the tubes by heating treatment is not guaranteed. Besides, the heating treatment has another problem. Unlike the incineration treatment, heating leaves blood in the tubes after the treatment. The blood that remains in the tubes drips from the tubes during direct disposal process, which has ethical non-acceptance and implications even though pathogens in the blood would be completely killed.

Acidic electrolyzed water has been used in various fields, such as agriculture, dentistry, food industry, livestock industry, and medicine, for the purpose of disinfection. Used blood testing tubes could be safe if they are treated with acidic electrolyzed water properly, which could introduce new ways of recycling. Tubes treated with acidic electrolyzed can be recycled. For example, the treated tubes can be used as feed stock for alternative energy source and waste heat recovery technologies; they can also used for recycling cloth. However, the main purpose of the complete disinfection of blood testing tubes is the reduction of hospital management cost. In Japan, since the disposal cost of infectious waste by a third party waste management company is approximately five times higher than that of non-infectious or general waste (Tanaka, 2007), hospitals could save significant management cost if they could achieve complete disinfection of blood testing tubes before disposal.
The purpose of this study is to investigate the total annual generation of the used test tubes used for blood tests and the possibility of treating the tubes by acidic electrolyzed water to reduce hospital management cost and to promote material recycling. The effective and proper treatment of the spent tubes by acidic electrolyzed water was also studied. This is the first report on the application of acidic electrolyzed water to the treatment of test tubes used for blood tests and on the recycling of the disinfected tubes.

2. Proposal of a treatment process for used test tubes used for blood tests

Fig. 1 shows the treatment process for used test tubes used for blood tests. The process consists two steps: the pretreatment and the disinfection processes.

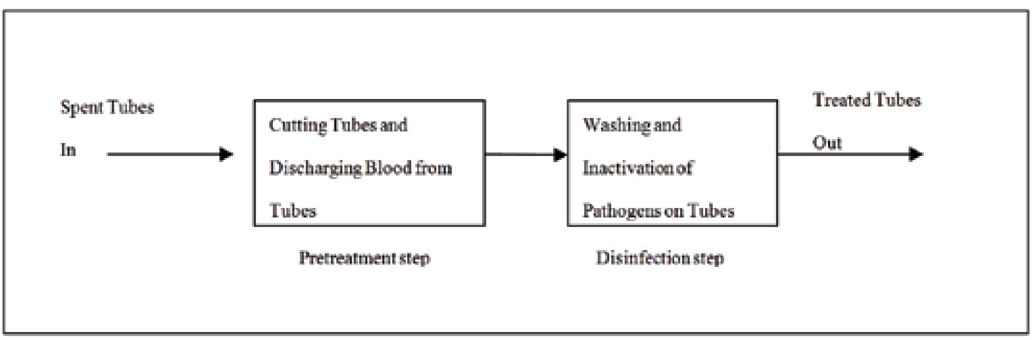


Fig. 1. Proposed treatment system for spent test tubes used for blood tests

The tubes are cut into the most appropriate shape, and the blood in the tubes is discharged during the pretreatment step. The cut tubes are sent to the disinfection step and are washed by acidic electrolyzed water. The ultimate goal is to complete the process in one box and to let the tubes fed to the process come out automatically after complete disinfection.

3. Materials and methods

3.1 Questionnaire survey for the annual generation of test tubes used for blood tests in Japan

The annual production of disposed test tubes used for blood tests was 800 million tubes in 2003, and all of these were consumed domestically (Muranaka, 2005). Then, when the relationship of "production = generation" was valid, the annual generation can be easily estimated. To confirm the relationship, flows of test tubes used for blood tests in hospitals were investigated by sending questionnaires to 80 hospitals nationwide through the postal service; these hospitals had large bed numbers and were randomly selected. Questions and information needed in the questionnaire were as follows. 1. Is the following relation on test tubes for blood tests "purchase numbers = disposal numbers" valid in your hospital? (Does your hospital store or keep test tubes for blood tests for a long period of time for the purpose of such as sample storage?) 2. What are the reasons if the answer in question 1 is "no"? 3. What is the annual number of purchased test tubes used for blood tests in your hospital? 4. Name of your hospital. 5. Number of beds. 6. Address of your hospital., 7. Name.

3.2 Test tubes for blood tests

Ten ml Venoject II vacuum test tube for blood tests for blood coagulation promotion (15.6 × 100 mm, TERUMO Corporation) was used for the experiments. The tube was made from PET. A coagulation promotion sheet in a tube was removed before the experiments.

3.3 Acidic electrolyzed water (AEWater)

AEWater was produced by the Hoshizaki electrolyzed water generator (ROX-10WA, Hoshizaki Electric Company, Ltd., Japan). The electronic current and voltage for the generator were set at 1.5 A and 100 V (single-current phase), respectively.

3.3 Washing apparatus

Toshiba AW-422V5 (TOSHIBA Corporation, Japan), a commercially and widely available home washing machine, was used to wash the tubes. The electric current and voltage were 3.3 A and 100 V, respectively; the maximum volume of the washing machine was 45 liters. Since the washing machine started with laminar flow mixing when the operation started with the ON/OFF switch button, the washing machine started at stand-by mode in order to obtain turbulent flow mixing at the beginning of the wash. The water level chosen for the experiments was 24 liters, or half of the volume of the washing drum.

3.4 Indicator microorganism

Strain *Escherichia coli* ATCC10798 K-12 was used as an indicator microbe for disinfection. *E. coli* K-12 was cultured in 100 ml LB broth at 30°C with an agitation of 120 rpm. After two rounds of 24-hour precultivation, a culture of *E. coli* K-12 was used for the experiments. Plating count of *E. coli* K-12 was done using deoxycholate agar (Oxoid, United Kingdom).

3.5 Marker

Tomato ketchup (KAGOME, Japan, hereafter called "artificial marker") was used as a marker to evaluate the efficacy of washing. The ketchup (1,000–10,000 cP) was selected on the basis of the following criteria: color, economical value, high accessibility, constant quality, and high viscosity than blood (approximately 4.6 cP). The evaluation of washing efficacy was done through visual observation for HACEP Mate (wiping type simple culture medium kit) assay.

3.6 *E. Coli* assay

HACEP Mate for detecting *E. coli* and total coliform bacteria (F&S Research Center, Japan) was used for the disinfection assay. This kit is widely used for checking hygienic safety of food and in the kitchen. Knives or cutting boards were wiped carefully and thoroughly with cotton swab, and the swab was submerged in prepared agar for incubation. After 24 hours of incubation at 35°C, the survival of *E. coli* K-12 was evaluated, and the color of the agar turned to yellow from red when it reacted with *E. coli* or the coliform. The color stayed red if *E. coli* or the coliform was inactivated. The sensitivity of HACEP Mate was as low as 1 CFU/ml.

For a submerged assay, deoxycholate agar (Oxoid, United Kingdom) was used. After the test tubes were treated with AEWater, they were placed in a Petri dish, and then deoxycholate agar was poured on the tubes until the tubes were submerged. The Petri dish was incubated at 37°C and observed after 24 and 48 hours.

3.7 Experiment on investigation disinfection capacity of AEWater

The disinfection capacity of AEWater against *E. coli* K-12 was studied. Five, 10, 15, and 20 ml of *E. coli* K-12 (5.6×10^7 CFU/ml) were separately transferred into 200 ml of AEWater, and they were mixed on a magnetic stirrer with mild stirring level for 15 and 30 seconds. After mixing for the a particular period of time, HACEP Mate was used for detecting the survival of *E. coli* K-12. The effective chlorine concentration was measured before and after the experiments with chlorine test paper, 10–50 ppm (Advantec, Japan).

3.8 Experiments for finding the best cutting type and most effective washing condition

A 1.2 g of the artificial marker was placed into each test tube and was uniformly spread on the inside wall of the tubes by a touch mixer (MT-31, Yamato Japan). Then, the tubes were left for 1 hour under room temperature. Afterward, the tubes were cut by a fret saw BANDSAW K-100 (HOZAN, Japan) into the following three types: half pipe cut, half length cut, and bottom edge cut. The cut types were shown in Fig. 2. The tubes were washed with tap water (24 liters and 15°C), and the best cutting type was decided based on the least amount of the marker left on the tubes, which was done by visual observation.

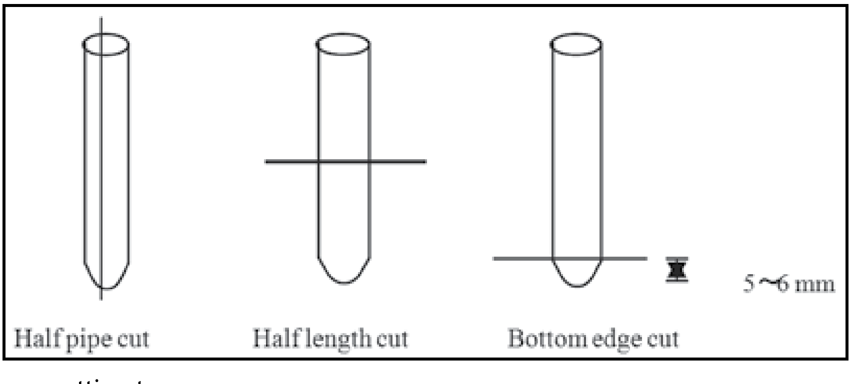


Fig. 2. Three cutting types

After the best cutting type was known, the optimal washing condition was studied. The same experimental procedure as the previous one for deciding the best cutting type was applied for finding the optimal condition. Under the optimal conditions found in the previous experiment, the disinfection test of *E. coli* K-12 was carried out. A 100 ml of *E. coli* K-12 was put in 10 liters of LB broth, and the test tubes used for blood tests, which were already cut according to the best cutting type, were placed in the broth. The broth was heated at 35°C by a ribbon heater Flexible Heater FHU-8 (ADVANTEC, Japan) controlled by a portable temperature controller TC-1N (ADVANTEC, Japan) and stirred at 120 rpm on Hyper Starter HPS-200 (AS ONE, Japan) for 24 hours. After 24 hours, the parts of the tubes were transferred into 24 liters of AEWater for washing. After washing under the optimal condition, the *E. coli* assay was carried out at parts of the tubes using HACEP Mate, as described in the previous *E. coli* Assay section.

3.9 Experiment for investigating dead spots on tubes against disinfection by AEWater

A 100 ml of *E. coli* K-12 was put into 10 liters of LB broth, and then the test tubes used for blood tests, which were already cut in several parts (upper part and bottom part) were put into the broth. The broth was heated at 35°C by a ribbon heater Flexible Heater FHU-8 (ADVANTEC, Japan) controlled by a portable temperature controller TC-1N (ADVANTEC, Japan) and stirred at 120 rpm on Hyper Starter HPS-200 (AS ONE, Japan) for 24 hours.

<table>
<tr><th>Test number</th><th>Test number of tubes</th><th>Cutting type</th><th>Cut condition</th><th>Treatment time (min)</th></tr>
<tr><td>1</td><td>5</td><td rowspan="4">Top edge cut</td><td rowspan="2">Cut litter remained With aluminum cap</td><td rowspan="5">5</td></tr>
<tr><td>2</td><td>100</td></tr>
<tr><td>3</td><td>24</td><td>Cut litter removed With aluminum cap</td></tr>
<tr><td>4</td><td>5</td><td rowspan="2">Cut litter removed Without aluminum cap</td></tr>
<tr><td>5</td><td>4</td><td>Bottom edge cut</td></tr>
</table>

Table 1. Test conditions

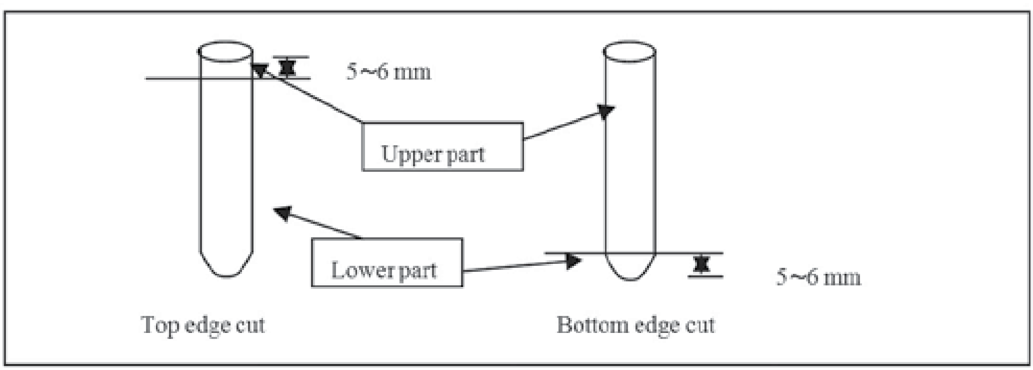


Fig. 3. Tube cutting and cutting parts

After 24 hours, the parts of the tubes were transferred into 24 liters of AEWater for washing. After washing, those parts were placed into Petri dishes for the assay to be submerged, which is described in the previous *E. coli* Assay section. The test conditions are shown in Table 1. The cutting types of a tube and the cut parts for this experiment are described in Fig. 3 and Photos 1 and 2. The conditions of cut litter that remained and was removed are shown in Photo 3.

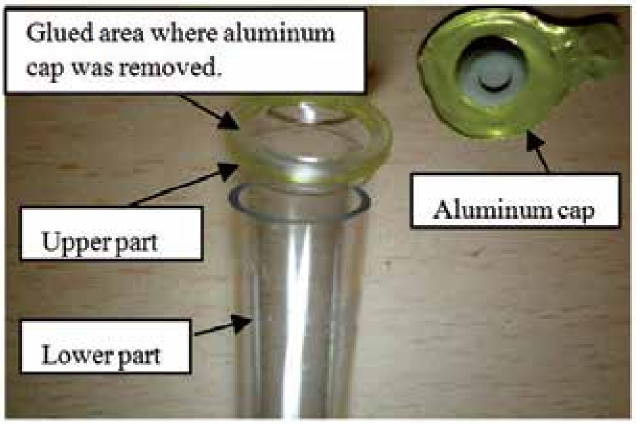


Photo 1. Top edge cut

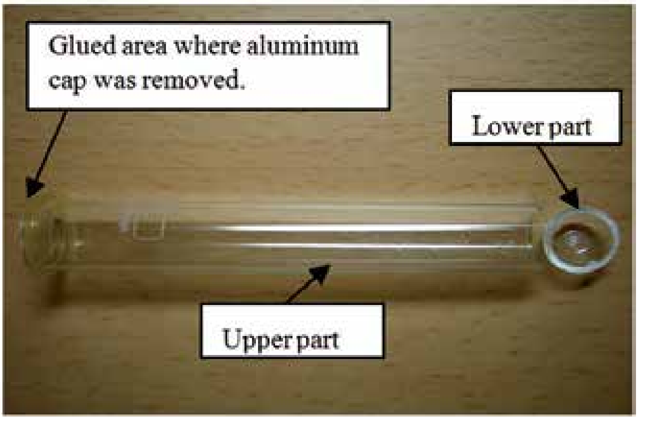


Photo 2. Bottom edge cut

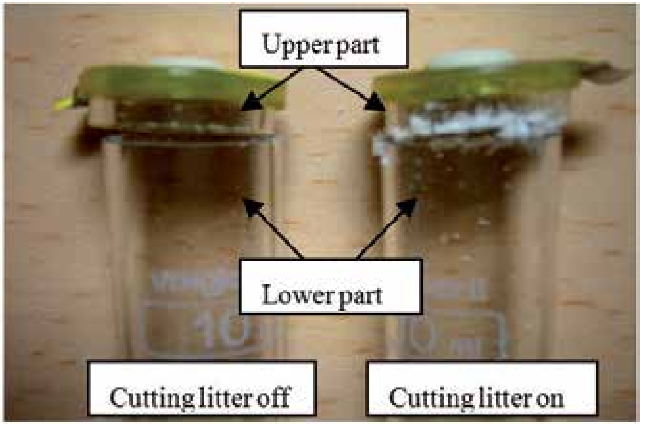


Photo 3. Tubes with cutting litter off and on

4. Results and discussions

4.1 Results of questionnaire survey for the annual generation of test tubes used for blood tests in Japan

Twenty-eight hospitals out of 80 responded to the survey questionnaires (collection percentage of about 35%). The results were summarized in Table 2. To avoid the specification of hospital names, the locations of the hospitals were stated through the prefecture level and bed numbers were expressed as more than or less than 700 beds. Most of the hospitals gave exact numbers for their test tube purchase; however, the numbers were expressed by only the third digit. According to the results, 24 of 28 hospitals answered that the relationship "purchase = disposal" on test tubes used for blood tests was valid (86%). Three hospitals answered in the negative with regard to the relationship "purchase = disposal," and the answer of "unknown" was obtained from one hospital. As Table 2 shows, the flow of disposal test tubes used for blood tests was very smooth from purchase to disposal in hospitals, and the tubes were disposed within a period of one month including sample storage. Hospital ID Nos. 18, 19, and 20 answered "not valid" to the relationship "purchase = disposal." At Hospital ID No.18, blood tests were not conducted in the hospital but in other organizations; that is why the relation was not valid. At Hospital ID No.19, the relationship was not valid because it found a large number of storage in wards, and a large number of test tubes used for blood tests purchased for tests became unnecessary due to cancellation of the tests for some reason. This hospital showed the relationship "purchase ≉ sample number," and the sample number tallied 95% of the purchase number, which was approximately 850,000 sample tubes. Hospital ID No. 20 had always some stock of the tubes in case of emergency, and that is the reason why "purchase = disposal" was not balanced. At Hospital ID No.16, which answered "unknown" to the relationship "purchase = disposal," spent test tubes were disposed mixed and along with other infectious medical wastes; therefore, the disposed number of spent test tubes was unknown. Observing the purchase number of test tubes used for blood tests, a wide range of 17,000–880,000 on purchase number can be noticed.

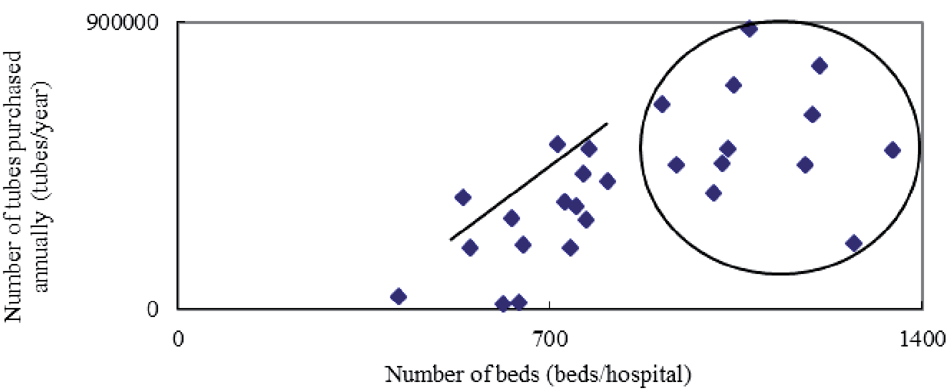


Fig. 4. Number of disposable test tubes used for blood tests purchased annually as a function of number of beds in the hospital

Fig. 4 shows the relationship between bed number and annual purchase number of test tubes used for blood tests. Avoiding the exact number of beds, the scale in Fig. 4 was made very roughly on purpose.

ID No.	Location	Quantity of beds	Quantity of Tubes Purchased per Year	"Quantity Purchased = Quantity Disposed" What can be said?	Remarks
1	Hokkaido	< 700	190,000	Yes	Stays for 1 month in Biochemistry and Immune serum Division. Stays for 2 days in Blood Test Division.
2	Iwate	> 700	335,000	Yes	
3		> 700	700,000	Yes	There are some stocks, but consumption and disposal are smoothly taken place in a short period.
4	Miyagi	< 700	40,000	Yes	No long stay in the hospital. There is a time lag from purchase to consumption.
5		> 700	200,000	Yes	
6	Saitama	< 700	22,000	Yes	
7	Kanagawa	> 700	765,000	Yes	
8	Shizuoka	< 700	350,000	Yes	In case of a long storage, transfer the samples to special storage tubes.
9		-	244,000	Yes	
10	Niigata	> 700	280,000	Yes	Consumption and disposal are smoothly taken place within 20 days.
11		> 700	323,000	Yes	
12	Toyama	> 700	400,000	Yes	
13	Ishikawa	> 700	453,000	Yes	Consumption and disposal are smoothly taken place within 1 week. Dispose the tubes as infectious waste.
14		< 700	200,000	Yes	
15	Fukui	< 700	290,000	Yes	No long stay in the hospital. Dispose the tubes as industrial waste after autoclave treatment.
16	Aichi	> 700	500,000	No answer	Since the tubes are disposed with other infectious waste; the quantity of the tubes disposed is unknown.
17		> 700	190,000	Yes	
18	Shiga	< 700	17,000	No	Since a part of blood analysis is ordered from outside affiliations, a number of the tubes disposed are different from those purchased. Some samples are stored for 1 year.
19	Osaka	> 700	880,000	No	"Quantity purchased = Quantity disposed" is not correct but "Quantity sampled = Quantity

					disposed" is, because some tubes purchased are forgotten and left in a ward and blood sampling was sometimes suddenly canceled due to unexpected events. Quantity of sample was 840,000.
20		> 700	364,000	No	The tubes were stocked for emergency use. Stays in freezers for 2 weeks.
21		> 700	456,000	Yes	
22	Hyogo	> 700	640,000	Yes	Some are stored but do not stay for long in the hospital.
23		> 700	610,000	Yes	
24	Okayama	> 700	450,000	Yes	
25	Hiroshima	> 700	520,000	Yes	Stays for 1 week in the hospital
26		> 700	500,000	Yes	Serum and plasma are separated and stored in special tubes. "Quantity purchased = Quantity disposed" is not correct for small hospitals that do not have basic analyzing equipments because they ask blood testing from outside testing affiliations.
27		> 700	425,000	Yes	
28	Fukuoka	> 700	500,000	Yes	

Table 2. Summary of questionnaires on management of blood sampling tubes

The purchase number of the tubes increased as the number of beds increased until some level. With regard to the data in the circle, there was no relationship between the purchase number and bed number. According to the results, it cannot say that the hospital with large number of beds always purchased a large number of disposal test tubes used for blood tests, and the purchase number of the tubes totally depended on the hospital condition.

Hospital ID No.17 used extra number of test tubes used for blood tests so that the extra number of the tubes should be also included in the calculation of the balance of "purchase = disposal." Moreover, Hospital ID No.19 proposed that the sample number, not purchase number, should be counted in order to know the disposal number of the tubes. Taking those comments into account, the trend seen from 28 hospital results implied that it would be acceptable even if the relationship "purchase = disposal" on test tubes used for blood tests was concluded as valid for the estimation of annual disposal tubes. Hence, the annual generation number of spent disposal test tubes used for blood tests was 800 million tubes in 2003.

A 10 ml Venoject II vacuum test tube for blood tests for blood coagulation promotion (15.6 × 100 mm, TERUMO Corporation) is 6.8 g. Suppose 800 million tubes estimated above were all 10 ml Venoject II vacuum test tube for blood tests for blood coagulation promotion (15.6 × 100 mm, TERUMO Corporation), 5,440 tons of PET resin was disposed annually. Since the annual generation of infectious medical wastes was estimated as 290,000 tons (Tanaka, 2007), the annual generation number of spent disposal test tubes used for blood tests amounted to 2% (probably more than 3%, including specimens). Regarding treatment cost, suppose the weight of a test tube used for blood test with blood is approximately 12 g (blood density of 1.0), then the total weight of the tubes becomes 9,600 tons, resulting from the multiplication of 5,440 by 12/6.8. The 9,600 tons was multiplied with 160,000 yen/ton (Tanaka, 2007) of treatment cost by third party waste management companies for infectious medical wastes, and the total treatment cost of the tubes that hospitals have to pay to is 1,540 million yen. In case that the disposal was made after the complete disinfection treatment, changing the condition from infectious medical waste to general medical waste, the total cost treatment cost of the tubes becomes 290–670 million yen, a half to one-fifth reduction of the cost, since it is 30,000–70,000 yen/ton (Tanaka, 2007) of treatment cost by third party waste management companies for general medical wastes. The treatment cost estimation of used test tubes used for blood tests at each hospital is shown in Table 3. The estimation was done by assuming that the weight of a used test tube used for blood tests with blood was 12 g, and the treatment cost by third party waste management companies for infectious medical wastes was 161 yen/kg (Tanaka, 2007). According to the table, the minimum treatment cost was 32,844 yen and the maximum was 1.7 million yen at Hospital ID Nos.18 and 19, respectively.

At Hospital ID No.19, it can be said that the treatment capacity of a treatment system should be 2,500 tubes/day at least if the system for treating spent test tubes used for blood tests was developed according to Table 3. In Fig. 5, the relationship between daily treatment capacity of used test tubes used for blood tests and the production cost for making a used tube disinfection treatment system. The production cost was calculated by a fixed rate method (annual depreciation = (actual cost – remaining price) / duration period), the while annual treatment cost of spent tubes is equal to depreciation and the duration period of the machine's lifetime is 10 years.

For instance, in the case of Hospital ID No.19, an estimated price of a spent tube disinfection treatment system would be around 19 million yen since the current annual treatment cost for spent tubes was 1.7 million yen. In case that the treated used tubes went for material recycling under an assumption of complete disinfection of used tubes, the selling revenue would be that as shown in Table 3, assuming 140 yen/PET resin kg. From Fig. 5 and Table 3, simply excluding running and maintenance cost, the treatment of used tubes at each hospital by purchasing the machine reduces the annual treatment cost of spent tubes and produces new revenue by selling treated tubes. Kagawa et al. (2006) reported that increase in the use of disposal goods in hospitals greatly contributed to increase in infectious medical wastes at hospitals. It is also already commonly known that plastics compose most of the medical waste in hospitals (Lee et al., 2002; Yamaguchi et al., 2002). It is obvious that the disposal of plastic medical goods will increase further in the future and that the treatment cost of these goods would become a tremendous burden to hospital management. Test tubes used for blood tests, unlike other medical goods, have an advantage over those treatments because those tubes have a very low possibility to be mixed with other medical waste during disposal; these are handled through a special room called a central analysis room. It can be said that changing infectious waste to being non-infectious and selling non-infectious wastes as resources reduce the economical burden of hospital management. Hospitals with low generation of used test tubes used for blood tests should cooperate with other hospitals for the treatment in order to reduce the treatment cost of its medical wastes.

ID No.	Estimated number of tubes disposed annually (tubes/year)	Estimated number of tubes disposed monthly (tubes/m)	Estimated number of tubes disposed daily (tubes/d)	Annual disposal weight (kg)	Annual treatment cost (yen)	Annual revenue by selling (yen)
1	190,000	15,833	528	2,280	367,080	319,200
2	335,000	27,917	931	4,020	647,220	562,800
3	700,000	58,333	1,944	8,400	1,352,400	1,176,000
4	40,000	3,333	111	480	77,280	67,200
5	200,000	16,667	556	2,400	386,400	336,000
6	22,000	1,833	61	264	42,504	36,960
7	765,000	63,750	2,125	9,180	1,477,980	1,285,200
8	350,000	29,167	972	4,200	676,200	588,000
9	244,000	20,333	678	2,928	471,408	409,920
10	280,000	23,333	778	3,360	540,960	470,400
11	323,000	26,917	897	3,876	624,036	542,640
12	400,000	33,333	1,111	4,800	772,800	672,000
13	453,000	37,750	1,258	5,436	875,196	761,040
14	200,000	16,667	556	2,400	386,400	336,000
15	290,000	24,167	806	3,480	560,280	487,200
16	500,000	41,667	1,389	6,000	966,000	840,000
17	190,000	15,833	528	2,280	367,080	319,200
18	17,000	1,417	47	204	32,844	28,560
19	880,000	73,333	2,444	10,560	1,700,160	1,478,400
20	364,000	30,333	1,011	4,368	703,248	611,520
21	456,000	38,000	1,267	5,472	880,992	766,080
22	640,000	53,333	1,778	7,680	1,236,480	1,075,200
23	610,000	50,833	1,694	7,320	1,178,520	1,024,800
24	450,000	37,500	1,250	5,400	869,400	756,000
25	520,000	43,333	1,444	6,240	1,004,640	873,600
26	500,000	41,667	1,389	6,000	966,000	840,000
27	425,000	35,417	1,181	5,100	821,100	714,000
28	500,000	41,667	1,389	6,000	966,000	840,000

Table 3. Estimation of annual treatment cost and revenue on spent test tubes used for blood tests

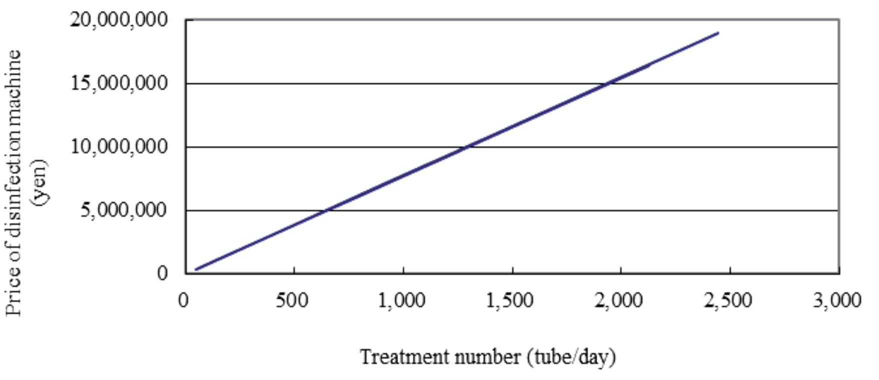


Fig. 5. Price of a used tube disinfection system as a function of treatment capacity of tubes

4.2 Results of experiment on investigating the disinfection capacity of AEWater

Results of disinfection capacity of AEWater are shown in Table 4. Despite the difference in mixing time, the results showed the same trend. A 5 ml or 2.8 × 10^8 CFU of *E. coli* K-12 was inactivated in 200 ml of AEWater in 15 and 30 second mixing times. Any cases that were more than 10 ml or 5.6 × 10^8 CFU of *E. coli* K-12 did not show disinfection capacity of AEWater. According to the results, it can be said that it requires more than 35 ppm of effective chlorine concentration to reach complete disinfection of *E. coli* K-12.

Mixing Time (sec.)	*E. coli* (ml)	Effective Chlorine Conc. before (ppm)	Effective Chlorine Conc. after (ppm)	HACEP Mate Color
15	5	more than 50	35	RED
	10	more than 50	20	YELLOW
	15	more than 50	10	-
	20	more than 50	less than 10	YELLOW
30	5	more than 50	35	RED
	10	more than 50	30	YELLOW
	15	more than 50	20	YELLOW
	20	more than 50	10	YELLOW

Table 4. Change in population of *E. coli* K-12 and disinfection capacity. Note: RED means no detection of *E. coli* K-12, YELLOW means detection of *E. coli* K-12, “ -“ means experimental error.

Suppose that the thickness of *E. coli* K-12 attached to the inner surface of the blood testing tubes was 0.1 mm. Since the inner surface area of a tube was approximately 41 cm^2, then the volume of *E .coli* K-12 on a tube was 0.41 ml or 2.3 × 10^7 CFU. Considering a 24 liter of AEWater in the washing apparatus, 3.4 × 10^{10} CFU or 600 ml of *E. coli* K-12 could be treated. Hence, it could be estimated that 1460 tubes could be theoretically treated with a 24 liter of AEWater.

4.3 Results of experiments of finding the best cutting type and most effective washing condition

Results of finding the best cutting are shown in Table 5 and Fig. 6. Control (no cut) tubes were completely washed for 300 seconds, but the efficacy became just 2% when washing time was shortened to 120 seconds. All tubes in half pipe cut type was almost washed as the tubes in the bottom edge cut type showed a very good efficacy. Among the cut types, the tubes in half length cut type showed poor efficacy. Photo 4 showed the washing performance on each cut type. For the washing of control tubes, the marker remained mainly at the bottom of the tubes and drew a line from the bottom to the upper sites of the tubes (Photo 4 (a)). The washing performance in Photo 4 (a) indicated that water current did not reach sufficiently the bottom sites of the tubes in 30 seconds and resulted in the marker being left at the bottom sites of the tubes. In Photo 4 (b), the washing performance on half pipe cut type was shown. As seen in the figure, the tubes were completely washed, which indicated that water current reached the entire parts of tubes and removed the marker thoroughly in 30 seconds. The washing performance of the half length cut type showed differences in upper and lower parts (Photo 4 (c)). Almost a complete washing was shown in upper parts of the tubes. It could be said that water flowed sufficiently through the pipes and washed out the marker. In the case of lower parts, like control tubes, the marker was not cleaned and a lot of it was left in the lower parts. The washing performance of the bottom edge cut type was very good and showed almost complete removal of the marker at the upper and bottom parts, like the performance on half pipe cut (Photo 4 (d)). Unlike the performance of the half length cut type, the upper and lower parts in bottom edge cut type were thoroughly cleaned. The lower parts could receive sufficient water flow to remove the marker. According to the results, the washing performance of both of half pipe cut and bottom edge cut types was very good, and none is apparently inferior than the other. Considering the ease of cutting and least time consumption, it can be said that the bottom edge cut was the best cutting type for washing the tubes.

Cut type	Number of tubes	Washing time (sec.)	Washed	Not Washed
Control (no cut)	50	300	50 tubes	0 tube
	50	120	1 tube	49 tubes
Half pipe cut	50	30	98 parts	2 parts
Half length cut	50	30	(upper) 22 parts	28 parts
			(lower) 0 parts	50 parts
			(sum) 22 parts	78 parts
Bottom edge cut	50	30	(upper) 50 parts	0 part
			(lower) 47 parts	3 parts
			(sum) 97 parts	3 parts

Table 5. Efficacy of washing on different cut types

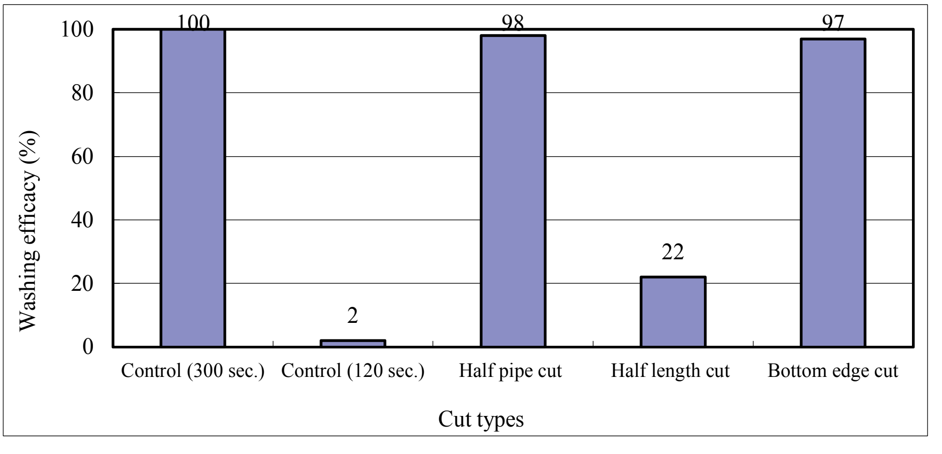


Fig. 6. Washing efficacy of the different cut types

(a) Control (no cut) tubes after 120 second washing

(b) Half pipe cut tubes after 30 second washing

(c) Half length cut tubes after 30 second washing

(d) Bottom edge cut tubes after 30 second washing

Photo 4. Washing performance of each tube cut type. Note:Tube(s) with a full of marker showed tubes before washing.

Number of tubes (tube)	Water Temp. (°C)	Washing time (sec.)	Upper part (part)	Lower part (part)	Total (part)
60	30	15	55/60	57/60	112/120
70	45		70/70	70/70	140/140
80			77/80	80/80	157/160
50	15	30	50/50	47/50	97/100
60			49/60	56/60	105/120
70			14/70	47/70	61/140
70	30		70/70	70/70	140/140
80			40/80	73/80	113/160
80	45		80/80	80/80	160/160
90			90/90	90/90	180/180
100			100/100	100/100	200/200
120			120/120	120/120	240/240
150			150/150	150/150	300/300
170			169/170	170/170	339/340
200			197/200	200/200	397/400

Table 6. Washing conditions and washing results. Note: washed numbers/total numbers.

With the best cutting type, the best condition for washing the tubes was investigated. Although it could be estimated in the previous experiment that 1,460 tubes could be theoretically treated with a 24 liter of water, disinfection efficacy may be significantly different from the estimation when *E. coli* K-12 on the tubes was tried to be disinfected because the tubes became obstacles against the flow of AEWater. Table 6 and Fig. 7 showed the washing conditions and their results. According to the results, water temperature influenced washing efficacy more than washing time did. At 15°C of water temperature and 30 seconds of washing time, washing efficacy on 60 and 70 tubes was 87.5% and 43.6%, respectively, whereas the efficacy was 93.3% on 60 tubes and 100% on 70 tubes for 30 and 45°C of water temperature, respectively, for 15 second washing time. At 70 tubes and 30 second washing time, washing efficacy changed drastically from 43.6% to 100% when water temperature increased from 15 to 30°C. The same trend could be seen when 80 tubes were washed at a 30 second washing time. The efficacy increased from 70.6% to 100% when water temperature increased from 30 to 45°C. The advantage of a longer washing time could be seen on 80 tube washing at 45°C water temperature. Washing efficacy was 98.1% for 15 seconds and that was 100% for 30 seconds. A difference of 15 seconds contributed an increase in efficacy from 98.1% to 100%. At 45°C of water temperature and 30 seconds of washing time, 100% efficacy was shown on up to 150 tubes. In the case of 170 and 200 tubes, the efficacy did not reach 100% and was 99.7% and 99.3%, respectively.

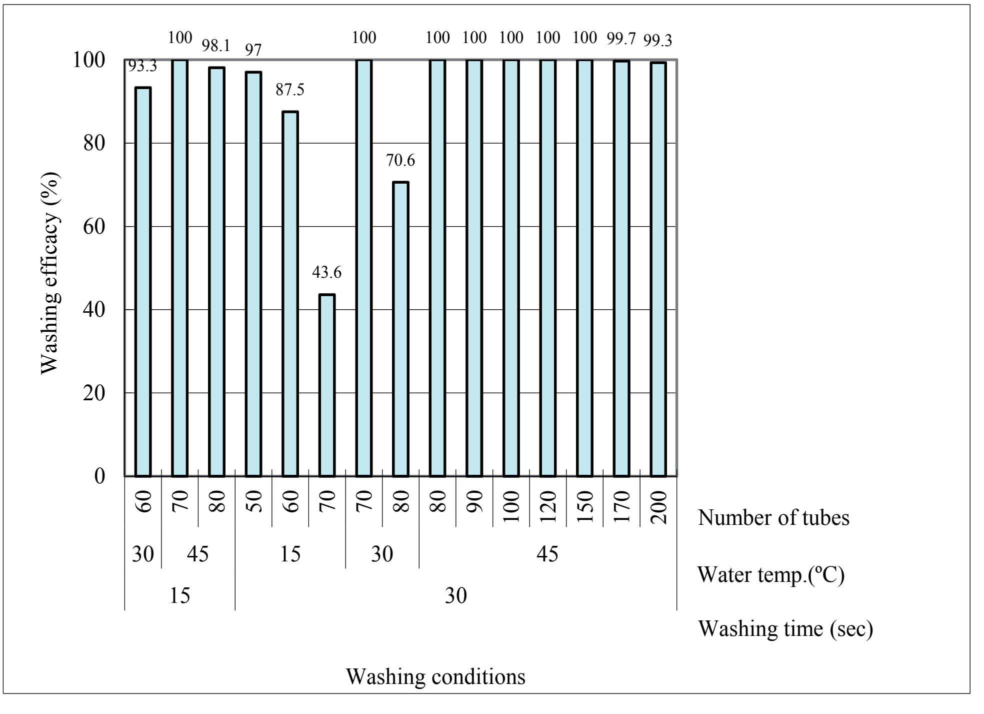


Fig. 7. Washing efficacy of different washing conditions

Number of tubes (tubes)	AEWater temp. (°C)	Washing time (sec.)	Positive part numbers (parts)	Negative part numbers (parts)	Disinfection percentage (%)	*E. coli* positive/negative in AEWater
50	10	30	0	100	100	negative
200	45	30	1	399	99.8	negative

Table 7. Disinfection test on the optimal condition

The disinfection test was performed under optimal conditions, i.e., a water temperature of 45°**C** and a washing time of 30 seconds. Two hundred tubes were used for 24 liters of water in this experiment as it was the maximum number of tubes used in the previous experiment. This experiment was also performed for a water temperature of 10°**C**. This temperature was considered, as 10°**C** is the temperature of tap water; hence, minimum operating cost can be expected using AEWater obtained from tap water without heating, thus conserving the energy supply that would otherwise be unnecessarily used for increasing water temperature. The results are shown in Table 7. Fifty tubes (9.36 $\log_{10}$ CFU) were completely disinfected in 24 liters of AEWater at 10°C. In the case of 200 tubes (9.96 $\log_{10}$ CFU), 1 part of a tube remained positive; therefore, it can be said that 150 tubes could be the safe amount for complete disinfection under the optimal condition. In both cases, the *E. coli* reaction in AEWater was negative.

Venkitanarayanan et al. (1999) reported that *E. coli* 157:H7 on the 100 m^2 area of a plastic cutting board was disinfected from 8.14 $\log_{10}$ CFU to 2.43 $\log_{10}$ CFU and from 8.01 $\log_{10}$ CFU to 0 $\log_{10}$ CFU for 5 and 10 minutes of washing time, respectively, at 45°C of AEWater temperature. The disinfection time in their study was much longer than that of this study: 30 seconds in this study and 5 or 10 minutes in their study. The reason for that could be attributed to the disinfection conditions with agitation or without agitation. Venkitanarayanan et al. (1999) used no agitation during disinfection, whereas this study used agitation during disinfection since a home cloth washing machine was used as a washing apparatus. According to this comparison, the efficacy of disinfection with AEWater dramatically improved when agitation was performed, which agreed with the result of the study conducted by Park et al. (2002).

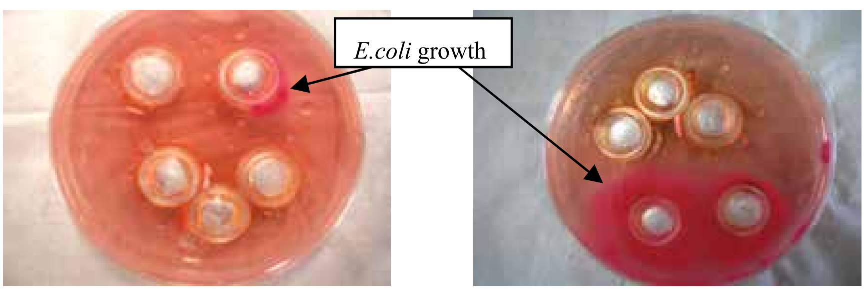
4.4 Results of experiment of investigating dead spots on tubes against disinfection by AEWater

In order to specify dead spots that the disinfectant cannot reach or is difficult to approach on the surface of the test tubes used for blood tests, a submerged tube assay in deoxycholate agar was carried out. Results were shown in Table 8. Comparing among test conditions 1 to 3, it is obvious that the existence of cut litter on tubes influenced efficacy during disinfection. The complex structure of the litter would play a role of a shelter for *E. coli* K-12 and protect *E. coli* K-12 from being exposed by AEWater. As a result, more numbers of positive tubes were seen in test numbers 1 and 2. The growth of *E. coli* K-12 was seen in Photo 5.

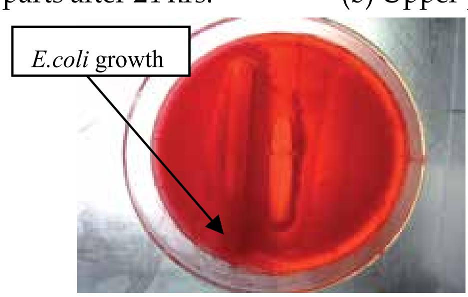
The effect of areas glued where an aluminum cap was attached on the efficacy of disinfection could be seen by comparing test numbers 3 and 4. Three parts over five showed positive for *E. coli* K-12 growth in test number 4. In test 5, a positive sample was also recognized on the area where an aluminum cap was removed after 48 h (Photo 6). As a

Test number	1		2		3		4			5		
Cut part of a tube (part)	Upper	Lower	Upper	Lower	Upper	Lower	Upper	Aluminum cap removed	Lower	Upper	Aluminum cap removed	Lower
Positive tube numbers after 24 hrs	4	0	2	0	1	0	0	3	0	0	0	0
Positive tube numbers after 48 hrs	4	4	4	2	1	0	0	3	0	4	1	0

Table 8. Results of the assay submerged

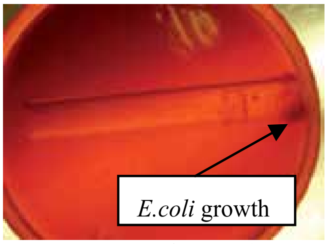


(a)Upper parts after 24 hrs. (b) Upper parts after 48 hrs.



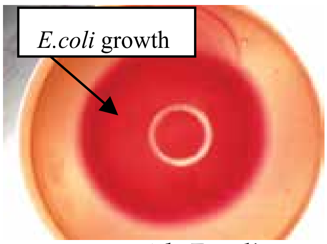
(c) Bottom part after 48 hrs.

Photo 5. Submerged assay for top edge cutting



(a) An upper part with *E. coli* growth.

(b) An upper part completely disinfected.



(c) A bottom part with *E. coli* growth.

(d) A bottom part completely disinfected.

Photo 6. Submerged assay for bottom cutting

conclusion, the existence of cut litter and the area glued where an aluminum cap was removed influenced in a negative way the efficacy of disinfection against *E. coli*. Those findings were very important aspects of developing a used tube disinfection treatment machine. According to Table 8, all lower parts in test conditions 3–5 were negative on *E. coli* K-12 growth. Disinfection washing in this experiment lasted for 5 minutes, which was far longer than the washing time in previous experiments (30 seconds). It can be said that 5 minutes is good enough as disinfection time. This corresponds to the result in Table 5 and the result in the study of Venkitanarayanan et al. (1999).

5. Conclusion

The total number of disposable test tubes used for blood tests was 800 million in 2003. Results of questionnaires reveal that the cost of disposing test tubes used for blood tests became a heavy burden to hospitals. It can also be deduced that hospitals with a large number of beds were always a large generator of used test tubes. The price of a used tube disinfection system would be 19 million yen for hospitals with a daily generation of about 2,500 tubes according to the calculations. A system that turns waste into resources will contribute to hospital health management; therefore, the development of this system is extremely important. The following conclusions are obtained from this experiment.

1. Acidic electrolyzed water can be successfully applied to the disinfection of test tubes used to collect blood samples.
2. The best cut type was the bottom edge cut type.
3. One hundred and fifty tubes were effectively disinfected by acidic electrolyzed water under these conditions: 24 liters of acidic electrolyzed water, 45°C of the water temperature, and 30 seconds of washing time.

4. The existence of cut litter and some special spots such as sticky areas reduced the efficacy of disinfection.

Further research, for example, on the disinfection efficacy for Hepatitis B and C, is absolutely needed for completing disinfection data collection; however, this preliminary study will contribute to the production of a complete system for a spent test tube used for blood tests, which will reduce drastically hospital medical waste management costs.

6. Acknowledgment

The authors deeply appreciate the support extended by 28 hospitals in Japan, which provided responses to our questionnaire survey.

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(Re-)constructing Nuclear Waste Management in Sweden: The Involvement of Concerned Groups, 1970–2010

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1. Introduction

Since the mid 1970s, the Swedish government has exerted considerable pressure on the nuclear power industry to develop a method for the final storage of spent nuclear fuel. In 2009, after almost thirty-five years of intensive research, large investments, and heated conflict with the anti-nuclear movement, SKB—a company launched by the Swedish nuclear energy industry to manage nuclear wastes—submitted a proposal for disposing of high-level nuclear waste in Sweden. This development has attracted considerable international attention. For example, the European Commission integrated the Swedish solution into its plans to develop a new sustainable nuclear energy technology platform for Europe (Elam and Sundqvist, 2009), making Sweden a frontrunner in nuclear waste management research. The advanced technological method proposed by SKB, referred to as the KBS-3 concept, involved high-level nuclear waste being contained in copper canisters, surrounded by benthonic clay, and put in a final storage repository at a depth of 500 meters in solid bedrock. According to SKB's proposal, the repository should be located in the municipality of Östhammar, near the Forsmark power plant, on the east coast of Sweden. In years to come, the Swedish government is expected to decide whether or not the KBS-3 concept meets its standards for safe disposal.

If KBS-3 is proclaimed safe and reliable, this will automatically be of great interest for countries using nuclear energy around the world. The Swedish solution is generally acknowledged as one of the most ambitious, technologically advanced, and carefully prepared projects for final storage, putting Sweden in the vanguard of nuclear waste management (Anshelm, 2006, p. 78, pp. 115, 163; Kantara, 2007). This estimation is supported by the British government's recent adoption of KBS-3 as a reference repository concept for disposing of spent nuclear fuel (Elam and Sundqvist, 2009). Accordingly, successful implementation of KBS-3 in Sweden may facilitate the expansion of nuclear power, at least in Europe. How is it possible that a small country such as Sweden has become a frontrunner in this endeavor? A preliminary explanation could refer to Swedish political culture, to its characteristic consensus and cooperation, and to the strategic actions of the nuclear energy industry, politicians, and authorities (Sundqvist, 2002; Lidskog and Sundqvist, 2004; Elam and Sundqvist, 2007). However, for almost five decades, the issue of nuclear waste management in Sweden has been permeated with profound public controversy between the nuclear power industry and various concerned groups (cf. Callon, 2003). In several major nuclear powers, such as the USA, the UK, and France, multiple social groups have also become involved in making nuclear waste storage a social and political problem (see, e.g., Rydel, 1984; Kronick, 1984; Chilvers, 2005, p. 242; Callon et al., 2009, p. 25). What has been the case in Sweden? Answering this question calls for multifarious inquiry to detect how concerned groups have coalesced around KBS-3, how they have influenced its development, as well as how they are influenced by being involved in a technoscientific controversy.

Accordingly, this article's aim is twofold: a. to empirically examine the technopolitics (we will come back to this notion) of the contemporary demos, including the influence exerted by concerned groups and their research efforts; and b. to analyze the transitions and social/epistemic identity-making that concerned groups undergo in order to maintain and increase their influence while seeking to (re)construct solutions to technoscientific problems. Technoscientific controversies imply a conflict of identities that is played out through conflicts over various kinds of knowledge, since it is impossible to distinguish the production of knowledge from the production of social identities (Callon et al., 2009, p. 105). Thus, we are interested in analyzing how concerned groups shape technopolitics and in how technopolitics shapes concerned groups. Whereas science and technology studies (STS) has previously been preoccupied with the former issue, one of our intentions is to demonstrate the usefulness of dealing with the latter. Our empirical investigation aims to enrich and broaden conceptual inquiry into how the "political" is implicated in the "technoscientific" (and vice versa) and into the role, epistemic identity, and scientific/research capacity of concerned groups.

To further our investigation, we will employ concepts and theoretical arguments emerging from STS. In the past three decades, STS has accentuated the congruence between technoscience and politics (what we call technopolitics), partly by challenging the expert authority of powerful scientific, state, and economic institutions and partly by articulating and representing lay, non-hegemonic epistemic views, engaging with science and technology via environmentalism, anti-globalization, consumer boycotts, patient movements, and the like (Jasanoff, 2004, p. 91). Through the conceptual lens of STS and given our focus on the historic endeavors of groups concerned with nuclear power and spent nuclear fuel storage, we will attempt to tell the story of the technopolitics of KBS-3.

2. Conceptual discussion: From politically legitimate deconstruction to scientific legitimate reconstruction

The nuclear waste management controversy in Sweden, as introduced above, resembles what Jasanoff (1992) calls a "quadrille dance": a historic movement performed by actors shifting between epistemological positions. STS has a long tradition of empirically dealing with controversies regarding the politics of technoscience, that is, who can legitimately make and control crucial policy decisions concerning science and technology. The latter immediately raises new questions: should science and technology be identified with politics or should they be preserved from the "prattle" of the public? Are technoscience and politics incompatible or inseparable notions? For Callon et al. (2009), it would be tragic to separate politics from technoscience, since they nourish each other. Social analysis ought to avoid fabricating identities such as the powerful and objective scientist while consigning laypeople to the level of emotion and passion: science and passion, knowledge and social roles are inseparable and co-constructed (p. 106). This is not to say that science and technology can be reduced to mere politics. In line with Barry (2001), we do not mean to denounce or condemn technology by asserting that it can be political; on the contrary, we also believe that controversies over technoscience open up new objects and sites of politics (p. 9).

Despite their efforts to radically redefine the concept, STS scholars seem to have made the same mistake regarding the concept of "politics" as they accused earlier social scientists of making regarding the concepts of "science" and "technology." They have partly to entirely entrenched politics within a general and abstract citadel of "decision-making," either in the "public domain" or exercised by the official machinery of the government, and partly used the same kind of modernist rhetoric that separates scientists from politicians, science/technology from politics, and experts from laypeople (de Vries, 2007). Indeed, Latour (2007) recognizes that STS has been preoccupied with focusing the analysis on deconstructing technoscientific practice, making the adjective "political" a meaningless concept: everything is political, but it is not explained how (p. 812–813). Identifying science and technology with politics without discussing how these domains intersect gave rise to a double criticism of STS articulated, on one hand, by those skeptical of the politicization of science and technology and, on the other hand, by activist social researchers who were worried about shining analytical light on areas unrelated to traditional political action. This makes for a tricky balance, and STS researchers have sometimes debated for and against both kinds of "deviations."[1] One way to go about this is to mix the public with technoscience, but reshaping the concept of "politics" this time. Yes, science and technology are "politics by other means," but what exactly does "by other means" mean? In the same paper, Latour offers five new meanings/interpretations of the word "political," situating them in five domains of technopolitical practice:

Meanings of "political"	What is at stake in each meaning
Political-1	New associations and cosmograms
Political-2	The public and its problems
Political-3	Sovereignty
Political-4	Deliberative assemblies
Political-5	Governmentality

Source: Latour (2007), p. 818

Table 1. Five meanings of "political"

In the first meaning, political-1, Latour argues that every non-human entity interacting with humans helps (re)configure the sociotechnical order. In our case, spent nuclear fuel, soil, water, etc., demanded redefinition of the discussion of nuclear power and nuclear waste management by scientists. When the redefined issue of nuclear waste management generated a concerned public, then it became political-2. To resolve the concerns of social groups, governments often attempt to turn a problem into a clearly articulated question of

[1] See, for example, the debate, organized by Evelleen Richards and Malcolm Ashmore, "More sauce please! The politics of SSK: neutrality, commitment, and beyond," in Social Studies of Science 26(2), pp. 219–228, May 1996; see also, for example, Collins and Evans (2002) and Woodhouse et al. (2002).

common good. For example, in the 1950s and 1960s, the state-owned Swedish nuclear power industry insisted that, unlike other industries, they paid serious attention to questions of industrial waste in the planning phase, which guaranteed that there was no reason to worry (Anshelm and Galis, 2009, p. 273). However, when the machinery of government fails to cope with public concerns, as was the case with nuclear waste in Sweden at an early stage, then the issue becomes political-3. The questioning of both expert knowledge and governmental authority to resolve a (previously) strictly technoscientific issue creates the opportunity for deliberative procedures (political-4). This is when concerned groups begin to claim the right to participate in the configuration of policy-making, for example, as various lay groups did in the anti-nuclear movement, and when concerned scientists entered the Swedish debate on nuclear waste in the early 1970s. This is when radioactive waste becomes "socio–active" (cf. Callon et al., 2009, p. 109). Finally, Latour defines political-5 as the "repoliticization" of all technoscientific issues that have become daily apolitical routines and rigid parts of the sociotechnical stratum. As mentioned, the renaissance of nuclear power called for repoliticizing the issue of nuclear waste management.

We should mention here that a technoscientific issue can be political in all five senses at the same time, the sense emphasized depending on where the analytical weight and epistemological focus lies. The aforementioned definitions do not constitute chronological phases that historically succeed each other, but rather overlapping aspects or layers of a single issue, in this case, nuclear waste management. Now that we have established a conceptual basis for the congruence of technoscience and politics to help us examine the technopolitical nature of nuclear waste management, let us define the ontological and epistemological margins of this study; in other words, at what loci and how will this paper address technopolitical action?

To answer the first part of the question, the empirical examination begins by briefly reviewing the configuration of nuclear waste management in Sweden from the 1950s to 1970s (political-1 and -2), continues by thoroughly analyzing the deliberative involvement of the environmental and anti-nuclear movements in areas previously dominated by scientific experts (political-3 and -4), and concludes by adding another meaning to Latour's interpretive schema of the word "political." As we will demonstrate, the development of the nuclear waste management issue never became "part of the daily routine of administration and management," since it was actively and closely monitored by a group of concerned activists, journalists, academics, and citizens. This group progressively advanced its engagement from merely participating in deliberative processes to constructing sophisticated scientific proposals and participating equally in the technoscientific debate. This circular epistemic identity-shifting, what we previously called the sociotechnical quadrille, is characterized as political-6 according to our schema.

The ontological exercise of this paper is to identify and discuss the quadrille steps involved in the nuclear waste management issue, which range from political-1 to political-6 (possibly bypassing political-5). In doing this, we will employ the main methodological approach that Latour's STS version has propagated since the mid 1980s, namely, following the actors "both as they attempt to transform society and as they seek to build scientific knowledge or technological systems" (Callon, 1986). In other words, we seek to analyze technopolitical action as taken by concerned groups within the complexity of a sociotechnical issue/controversy. As Latour (2007, p. 814) acknowledges, without an issue there is no politics. How do we identify and trace an issue? In our case, and answering the question on epistemology, to follow the actors and trace the issue, one must delve into the historical backgrounds of the environmental and anti-nuclear movements in Sweden. The 1970s were marked by general attacks on science, the political system, modernity, and the mainstream approach to nuclear power, contributing to what Foucault (2003) termed "the insurrection of subjugated knowledges." This sort of autonomous and decentralized knowledge, previously disqualified as non-conceptual, insufficient, and hierarchically inferior by the gatekeepers of "scientificity," appeared from below, based on what people knew and experienced at a local level (pp. 5–10). The insurrection of subjugated knowledges establishes the basis for the construction of knowledge outside the secluded laboratory and policy room, making nuclear waste management a political-4 issue.

Nevertheless, the model of activist and academic critical deconstruction of scientific "facts" and artifacts that accentuated the "lack of scientific certainty" has also begun to face critical scrutiny. As Latour (2004) argues, the issue was never to abandon the construction of facts or to reject dogmatically the production of technoscientific knowledge, but to come closer to them and renew empiricism by adding reality to established matters of fact (pp. 231–232). In doing so, the close study of technopolitical complexity in knowledge production also includes answering questions regarding who counts as a legitimate participant in a technoscientific dispute and who makes decisions concerning issues of participation (e.g., a substate organ, the nation state, a suprastate organization, the scientific community, or the claimants of citizenship) (cf. Jasanoff, 2004, p. 92). This line of argumentation is valuable in two ways, epistemological and ontological. On one hand, it marks a conscious effort by the social sciences and STS to go beyond the label of "deconstruction" by embarking on epistemological inquiry that allows interventionist modes of more socially grounded accounts of science and collaborative research and by promoting collective deliberation and co-construction (or co-production) of science and society, or even reconstruction of technoscience, in a manner that may be productive for both scientific scholars and activists (Jasanoff, 2004, pp. 68–69; see also Woodhouse et al., 2002; Mesman, 2007). On the other hand, it allows concerned groups that have been earlier neglected by confined scientific communities (Callon et al. 2009, p. 46) to reclaim their position not only in social analysis but also in the technopolitical agora as legitimate carriers of technoscientific knowledge (political-4).

In the words of Callon et al. (2009), the research collective cannot be identified with a simple group of researchers, but constitutes a community of distributed intelligence (p. 57), a community of distributed epistemic capacity. The "concerned groups" concept describes a dynamic process in which groups concerned with technoscience not only assert their existence, enact their identity, and formulate their demands, but also transform their epistemic status by actively participating in technopolitics. Accordingly, we are interested in how the epistemic status/identity of laypeople (e.g., activists, journalists, academics, and citizens—all concerned groups) is distributed, depending on their negotiability and participation (or lack thereof) in configuring various technopolitical practices. In this study, the anti-nuclear movement and its advocates will be treated as a concerned group in transition. In line with Callon (2003), a theoretical focus on the participation of laypeople and on the transformations of their epistemic identity implies a shift from traditional decision-making models: lay involvement delivers a preliminary blow to the traditional division between scientists and laypeople (p. 57).

This is the process that Callon describes as "research in the wild," whereby laypeople take action and participate in producing scientific facts ("the insurrection of subjugated knowledges"). To achieve this participation, laypeople establish new practices, exploit existing knowledge, negotiate with other groups, and form new organizational configurations (Galis, 2006, p. 40). Callon and Rabeharisoa (2003, p. 62) define "research in the wild" as the process by which laypeople accumulate and compare their experience and build collective expertise as authentic as that of "experts or scientists," even though it is different. In contrast to confined research, research in the wild does not possess or claim "scientific" purity; instead, it directly confronts compound, impure, polluted realities. The concept of research in the wild highlights the perspective that the management of technoscientific facts and artifacts does not have to follow the traditional laboratory route, which often implies a relatively passive role for the public sphere and the domination of scientists and engineers (Galis, 2006, p. 40). After all, laypeople possess expertise concerning their own needs, which is crucial knowledge when it comes to designing and implementing various technologies and which emerges from research in the wild. This conceptual apparatus allows us to "trace the issue" and follow the (de/co/re)construction of nuclear waste management by concerned groups and their research in the wild in Sweden since the 1950s.

3. A note on method

This paper analyzes the technopolitics of nuclear waste management in the public debate/controversy involving the nuclear industry and the anti-nuclear energy movement in Sweden. We are interested in the part of the debate articulated in printed documents. We agree with Ockwell and Rydin (2006) that controversies involving various knowledge holders can be understood fully by investigating the discursive bases of the various linkages between these parties (p. 381). Such a methodological choice lets us monitor the argumentation and rhetoric of the anti-nuclear movement, which has mainly employed the strategies of writing discussion pieces for publication in the mass media and scientific journals and of publishing books and leaflets to articulate their standpoints, instead of launching costly major campaigns. In this way, SKB was forced to confront the anti-nuclear movement in the formation of public opinion and to participate openly in media debate concerning nuclear waste management. The conflict/controversy has thus mainly been articulated in the public debate.

Accordingly, our analysis will be based on a close reading of articles in newspapers, websites, and scientific journals and of official reports, leaflets, and books. This material, which comprises over 1200 documents, was collected through extensive searches of databases (i.e., Artikelsök, Presstext, Mediaarkivet, and Biblioteksstjänsts tidnings- and tidskriftsindex) covering all Swedish newspapers and the most important Swedish journals. Books and official reports published in the study period were also examined. We also collected and analyzed articles and reports published on the websites of anti-nuclear organizations such as MKG and MILKAS. Our method entailed the close reading of every text situated in the debate in order to identify central meanings and themes. Individual texts were analyzed using a thematic coding technique for detecting patterns in qualitative data. This approach to coding was inspired by the analytical procedures developed by Strauss and Corbin (1990), but does not claim to be a grounded theory approach; instead, this study uses grounded theory methods as ways of categorizing and identifying patterns. Specific questions stood out as central, while others were found to be peripheral or even absent. Since the texts were produced over five decades, it was possible to detect shifts in arguments over time. More extensive empirical evidence than can be provided here to support certain interpretations can be found in the book Bergsäkert eller våghalsigt? (Anshelm, 2006).

4. The issue of nuclear waste management in Sweden: from confined research to technopolitics

4.1 The nuclear power industry and the declared absence of risk

In the 1950s and 1960s, the issue of nuclear waste in Sweden was the sole purview of scientific or state authorities. It was generally understood that the Swedish nuclear industry, comprising Atomenergi AB (Atomic Energy Ltd.), and state authorities shared the responsibility for nuclear waste management. When the issue was raised, nuclear physicists, reactor engineers, and politicians maintained that future technology would eliminate any waste-handling problems. Information leaflets from the nuclear energy industry did not even mention risks related to nuclear waste. In newspaper articles, nuclear physicists and reactor engineers predicted that people in the future would learn to live with and handle radiation, just as they had learned to live with and handle electricity (Westermark, 1952; Brynielsson and Eklund, 1954; Svedberg, 1955a; Funke, 1956; AB Atomenergi, 1957). In the same period, in Europe as in Sweden, communication specialists explained, popularized, and reassured: there really was no risk (Callon et al., 2009, p. 14; Anshelm, 2006, p. 34). At the time, it was very unusual to question the statements of confined research concerning nuclear waste and its management, even though these statements were sometimes only speculations concerning future technology. In that period, both scientific and political lobbies perceived nuclear waste as essentially a risk-free scientific issue (Anshelm, 2006, p. 28). Laypeople still constituted outsiders who were unable to influence the scientific/political configurations of nuclear waste.

4.2 The rise of the anti-nuclear movement and the entrance of laypeople into the debate

In the early 1970s, the first conflict emerged concerning the social and environmental characteristics of risks associated with nuclear waste. Plans to open a reprocessing facility in Bohuslän in western Sweden drew protests. A local environmental movement was mobilized to prevent the building of what they called a "plutonium factory." It was argued that the facility would expose the local inhabitants to great risks and that plutonium produced during reprocessing was one of the most dangerous and long-lived poisons known (Jacobsson, 1969; Johansson, 1969). At the same time, scientists such as Hannes Alfvén[2] and Sten Lindeberg began to criticize the scientific reasoning of the authorities responsible for future nuclear waste management as well as the science underlying the first official report on nuclear waste. The aforementioned physicists made clear in public debate that the scientific statements underpinning the political decision-making were nothing but qualified interpretations molded by political, economic, and

[2] Alfvén, a physicist awarded the Nobel Prize, was engaged early on as an expert in planning the Swedish nuclear energy program and later became familiar with the nuclear energy debate in the USA. He wrote an open letter to the Swedish government pointing out that nuclear energy and especially nuclear waste involved enormous risks; he also warned that the mass production of nuclear waste could eventually poison the earth and jeopardize the future of humanity (Alfvén, 1972).

technological interests. Science was not politically neutral, and it was possible to formulate alternate scientific interpretations that were as trustworthy as the established ones (Blomfeldt, 1976; Lindeberg, 1976).

Following the Bohuslän events, in spring 1970, a lengthy debate concerning the risks related to nuclear waste was initiated in the pages of Dagens Nyheter, the leading Swedish daily newspaper. On one side were spokespeople for what some years later became the anti-nuclear movement, while on the other stood representatives of the authorities responsible for radiation and of the nuclear energy industry. Suddenly, nuclear waste, which reactor engineers and radiation experts still insisted was not a problem, had become the subject of public controversy (Gillberg, 1970; Lindell, 1970; Jugnell, 1970). The intense discussion led to nuclear waste management being seen as the most urgent environmental issue facing Sweden in 1972. Criticism that the risks had been neglected for years grew as the anti-nuclear movement increased its influence (Anshelm, 2006, pp. 39–41). In 1973, the Swedish parliament imposed a moratorium on commissioning new nuclear power plants until a reliable plan for nuclear waste management had been presented (Lindquist, 1997). In fewer than four years there had been considerable reconfiguration of public discussion of the risks related to nuclear waste. The same period marked the entrance of laypeople, in the form of an anti-nuclear movement, into the debate on nuclear waste (Anshelm, 2006, pp. 37–66); confined research and its scientific authority were under attack. These were the very first critical voices that set the stage for the beginning of the quadrille.

Inspired by Alfvén, politicians in the Centre Party also began to question the scientific statements of researchers in the nuclear energy industry, winning a historic victory in the 1976 parliamentary election by emphasizing the overwhelming risks related to the storage of spent nuclear fuel (Rainer, 1973; Fälldin, 1976). After 44 years in office, the social democrats were thrown out when the Centre Party joined forces with the anti-nuclear movement and researchers in the wild, aligned against the nuclear power industry and state authorities. The first steps in the sociotechnical quadrille, taken when Prime Minister Thorbjörn Fälldin was elected based on the epistemic status of concerned groups and oppositional activist-scientists, had severe consequences for Swedish energy politics for decades to come (Anshelm, 2000, pp. 111–193).

4.3 Researchers in the wild go deconstructivist

The anti-nuclear movement and environmental organizations engaged their own scientific experts to conduct research in the wild, to deconstruct the scientific statements and truth claims articulated in state investigations and industry reports. This marked a turning point in the debate, since it was the first time that nuclear waste management was referred to as a hazard to the public (Anshelm and Galis, 2009, p. 273). At the same time, it was obvious that the nuclear waste issue no longer constituted an object of purely scientific concern, defined solely by participants in confined research. On the contrary, a social movement expressed its distress and questioned scientific and state authority regarding nuclear waste management, making it a political-2 issue. The hegemony of confined research was being seriously deconstructed. Researchers in the wild—i.e., oppositional scientists, journalists, environmentalists, politicians, and concerned laypeople—questioned "facts" pronounced in the name of science by confined scientists, because they interpreted these "facts" as expressions of political interests/concerns and thought that they exceeded what could be confirmed from a strictly scientific viewpoint (Lindeberg, 1976; Moberg, 1976; Westman, 1976). It was no longer possible to talk about nuclear waste management without at the same time saying something about what constituted a desirable society. The first steps in the quadrille had been taken.

In 1976, the first state investigation of nuclear waste management was presented (SOU, 1976).[3] What is important about this investigation was its insistence on conducting a completely new risk evaluation and on finding convincing methods for securing final storage. Risks related to nuclear waste were no longer peripheral unquestionable scientific facts to be left to confined research—that is, scientists and engineers working on the assumption that people would learn to live with the risks. Instead, presenting a reliable storage method in advance became a prerequisite for the further development of nuclear energy in Sweden. Accordingly, the nuclear industry and reactor engineers had to change their risk evaluations as well and thereby their position in the dance. For the first time, the anti-nuclear movement did not simply oppose confined research and nuclear industry, but made concrete suggestions concerning safe storage methods. In that sense, laypeople no longer constituted just a skeptical activist group, but now sought to "stick their noses" into technoscience. It was truly an insurrection of subjugated knowledges against unreliable storage methods.

Ultimately, the nuclear energy industry yielded to the pressure of the anti-nuclear movement and its advocates, though this was certainly no unconditional capitulation. The first traces of research in the wild convinced the government to force the industry to embark on an ambitious research project to develop secure methods for managing and storing nuclear waste. A special company, SKB, owned by the nuclear energy industry, was created for this purpose. Through SKB, the nuclear energy industry took full responsibility for its byproducts and made large investments in research and technological development. Three suggestions for the final storage of nuclear waste were presented in proposals that became known as KBS-1–3, which focused on geological and technical aspects of storage. The proposals for final storage presented in the official report of the first state investigation had been developed and refined by a cadre of contracted experts (Rosenberg, 1977; Thunell et al., 1977; Mosesson, 1977). New proposals were circulated for comment nationally and internationally. SKB eventually presented a method for final storage of nuclear waste that was described as one of the most advanced in the world. The company insisted that this method would ensure secure storage of nuclear waste (Bjurström, 1986; Bjurström, 1988; Falk, 1995; Papp, 1995).

However, detailed examination of the various alternatives, as well as the considerable funds and research energy invested in the project, indicate that, in the 1970s and 1980s, decision-makers in the nuclear energy industry defined risks differently from how their predecessors had. The risks posed by nuclear waste were taken much more seriously than before. Accordingly, the conflict between the nuclear energy industry and the anti-nuclear movement no longer concerned whether or not nuclear waste involved large risks. The conflict now concentrated on whether these risks could be handled using methods

[3] The so-called AKA investigation began on 25 April 1973, just one month before the moratorium was imposed. In this investigation—in which representatives of confined research (e.g., nuclear physicists and radiation experts), representatives of the nuclear energy industry, and politicians with an optimistic view of nuclear energy played important roles—risks related to nuclear waste were discussed more seriously than ever before in Sweden.

developed solely by the nuclear energy industry, or whether researchers in the wild could deliver serious criticism and discuss alternatives. In answering that question, the nuclear energy industry, government, and the anti-nuclear movement, together with their consulting geologists, engineers, and radiation experts, came to completely different conclusions (Lindeberg, 1976; Mörner, 1978; Bjurström, 1986, 1988). This constitutes the first concrete dispute between research in the wild and confined research during the controversy regarding the storage of nuclear waste management in Sweden, a dispute that involved deconstructing "objective" scientific facts regarding nuclear waste and the occurrence of impure technopolitics, that is, different perceptions and proposals regarding storage methods. In the words of Callon et al. (2009), disorder arises partly when confined research fails to deliver pacifying knowledge on which a socially reassuring political debate can be developed and partly when laypeople are allowed to participate in discussing experiments and their results (p. 120). In this case, researchers in the wild even managed to receive the support of the Swedish government, which put considerable pressure on SKB by passing a law requiring the presentation of a completely secure method for storing spent nuclear fuel before any new reactors, including those already built, would be permitted to start operating (Anshelm, 2000, pp. 190–193).

4.4 Test drilling and the insurrection of local resistance

In the 1980s, SKB began test drilling in Swedish bedrock in order to meet the requirements established by the Centre Party-led government. Environmental organizations and critics of KBS perceived the Swedish bedrock as "living" and changeable. They enlisted the support of several geologists who testified that the bedrock had undergone any number of transformations, especially during ice ages. Geologists revealed traces of earthquakes, faults, fissure formation, and altered subsoil water flows. It would clearly be impossible to predict the condition of bedrock for a period of 100,000 years, especially if the bedrock contained a nuclear waste repository that might affect the flows of both subsoil water and water in fissures. Accordingly, the anti-nuclear movement, Greenpeace, and several local resistance groups opposed all forms of underground storage of nuclear waste. They asserted that the concept of "getting rid of" nuclear waste by simply burying it rested on the same view of nature that had caused earlier environmental disasters. The critic's view of nature could not accept the construction of a facility intended to keep nuclear waste isolated from the biosphere for an almost incomprehensibly long period. They argued that dry deposit and storage at ground level were preferable, since the nuclear waste could then be kept under constant surveillance and its containment would not be dependent on the changeable and unpredictable natural environment (Anshelm and Galis, 2009). Several geologists publicly deconstructed the company's geological assessments and local resistance groups refused to accept test drilling in their municipalities (Noresson, 1985; Mörner, 1988; Alfvén, 1988; Edberg, 1988; Gahrton, 1988).

As a result, in several places, drilling was obstructed by laypeople (concerned local residents), who claimed responsibility for future generations and the environment. However, they interpreted and acted on this responsibility in a way that differed completely from SKB's. They did not act on their responsibility by seeking methods for storing nuclear waste, but by protecting their home districts from the radioactive contamination they feared could result from inappropriate and untested methods of underground storage. Accordingly, they completely opposed KBS-3 and all underground storage of nuclear waste.

This interpretation of responsibility was based on research in the wild, which mobilized counter-experts who deconstructed the claims of geologists and engineers contracted to SKB (Noresson, 1985). By the end of the 1980s, this resistance forced SKB to realize that the company could not take sole responsibility for nuclear waste storage: it would have to change tack again and allow local residents to share responsibility by involving them and winning their consent to any project. Any attempts to force waste storage measures on people were doomed to failure since they led to serious conflicts (Arpi, 1990). Concerned groups had not only forced the nuclear waste industry to dance with them, but had even made it change positions.

4.5 Deconstructing the confined KBS

This was definitely not the last step in the quadrille. Enlisting researchers in the wild was becoming increasingly unavoidable as concerned laypeople grew in number, formed alliances (e.g., local residents with counter-experts), and increasingly made themselves heard (cf. Callon, 2003, p. 54). Their voices helped co-construct both the risks related to nuclear waste management and how responsibility for this issue was articulated and manifested in the debate. Nuclear waste management was no longer an issue that concerned only confined experts and politicians, but was the subject of contention between research in the wild and confined research, something that prompted new research efforts and criticism. This also marked a shift in the epistemic identity and technopolitical involvement of concerned groups as well as the democratization of an initially purely technoscientific issue.

In the 1970s and 1980s, concerned groups in Sweden did not limit their activism to merely deconstructing SKB's project; they also attempted to co-construct the issue of responsibility for the management of spent radioactive fuel. This allowed them to take a position in the debate on nuclear waste storage by defining the key issue (i.e., determining and evaluating sustainable and environmental policy alternatives for managing radioactive waste) and by assessing, with the help of their advocates, environmental risks and impacts as well as acceptable methods (i.e., dry deposit and ground-level storage instead of bedrock drilling). Nevertheless, there was still much to be done to integrate concerned groups completely into the policy-making.

The democratization of the spent fuel issue allowed critics of nuclear energy to consider every scientific statement about the future management of nuclear waste as technopolitical. Awareness of scientific and technical weaknesses led to the co-construction of the terms of the debate and the emergence of new inquiries and critiques (cf. Callon et al., 2009, p. 15). KBS-3 and SKB's technical reports were thoroughly reviewed and heavily deconstructed in every detail. Every single report was scrutinized by scientific counter-experts (including researchers in the wild) who maintained that the nuclear energy industry was presenting energy policy in a scientific guise (Lindeberg, 1978; Anér, 1978). In scientific disciplines such as geology, intense controversy arose between researchers working for SKB and independent researchers: the latter feared that the former were allowing geology to be used for purposes that could undermine its trustworthiness. Some critical researchers were part of the anti-nuclear movement and publicly denounced their colleagues for participating in KBS research (Mörner, 1978; Nilsson, 1978; Mörner, 1979; Sundqvist, 2002, pp. 150–171). Geologists spoke out against geologists, hydrologists against hydrologists, physicists against physicists, and engineers against engineers. Though most researchers defended KBS and considered its proposals scientifically valid, the critical researchers and researchers in the

wild received not only much public attention but also considerable support in the Swedish parliament. Their accusations that SKB's reports had been revised to suit company interests undermined efforts to demonstrate that there was widespread scientific consensus concerning KBS-3. As a result, SKB was repeatedly forced to "refine" the KBS-3 concept, to defend itself from the attacks of research in the wild (Anshelm, 2006; Sundqvist, 2002).
This distrust was hardly reduced when KBS-3 was circulated for scientific comment in 1984. The anti-nuclear movement and several environmental organizations maintained that feedback from critical scientists and institutions had been removed and that only positive opinions had been retained (Ringsberg, 1984; Eriksson et al., 1986; Noresson, 1986). Critical remarks from researchers in the wild were also ignored, reinforcing the conviction in the anti-nuclear and environmental movements that the nuclear energy industry was conducting unscientific research and that all further production of nuclear waste should be stopped immediately. The critical geological and technological experts taking sides with the environmental organizations questioned the assumptions on which SKB's construction of repositories and encasements were based, addressing such issues as hydrological conditions, fissures, stability, and corrosion (Eriksson et al., 1986; Åhäll, 1986; Holmstrand, 1987; Mörner, 1988; Holmstrand, 1990). SKB rejected these criticisms and pointed out that a massive body of international scientific expertise had declared that KBS-3 was secure (Bjurström, 1988). The opposing positions were thus rigidly fixed and accusations of unscientific behavior were reciprocal. During that period, the technopolitical setting of nuclear waste management implied hard deconstruction of the opposing party's scientific profile and competence.

4.6 The nuclear power industry's efforts to redistribute responsibility

As demonstrated earlier, SKB's position was that disposal in massive bedrock would guarantee the secure storage of nuclear waste. Searching for the most suitable bedrock had accordingly been a priority. However, in the late 1980s, SKB announced that it was possible to construct a final repository almost anywhere in Sweden and that the most important barrier preventing radiation from reaching the biosphere was not the bedrock but the copper encasement of the waste. According to SKB, the role of the bedrock was no longer to prevent radioactive waste reaching humans but to prevent humans reaching nuclear waste (Bjurström, 1989; Ahlström, 1989; SKB, 1992; Sundqvist, 2002, pp. 113–116). This assertion rendered moot the whole question of whether the bedrock was stable. The problem was no longer the sustainability of nature but the sustainability of technological construction. This reorientation enabled SKB to avoid awkward geological questions while allowing nuclear waste storage facilities to be located wherever the political will to cooperate existed. The same reorientation occurred in the USA in 2000, when encasement rather than geological conditions was declared the most important factor in safe nuclear waste storage (Macfarlane, 2003:793). All of a sudden, SKB's rhetoric was based on political argumentation, i.e., striving for cooperation with a municipality, rather than purely scientific facts, i.e., geology.
Following this line, in 1992, SKB invited local residents to a new turn in the dance. The company wrote to all 286 municipalities in Sweden asking whether they were interested in cooperating with investigations of local bedrock to find suitable sites for a final repository for high-level nuclear waste. The technocratic model having failed, this letter invited municipalities and citizens to share responsibility for managing nuclear waste, making the

issue a matter of common good. During this information campaign, SKB also pledged that the municipality selected as a repository would be guaranteed a considerable number of jobs for many years to come. Once cooperation with two municipalities in northern Sweden, Malå and Storuman, had begun, SKB representatives argued that the responsibility for handling nuclear waste rested on the entire Swedish population. SKB accordingly began to enlist laypeople in their project. All municipalities that might have suitable sites for a final repository had a moral obligation to help solve the problem, since it was in the common interest to ensure that storage was as secure as possible (Thegerström, 1993). These arguments indicate that a considerable shift had taken place. First, SKB did not base its campaign to recruit Swedish communities simply on scientific facts, but employed technopolitical, moral, and economic arguments. Second, SKB no longer saw itself as having sole responsibility for the storage of nuclear waste and for choosing sites on scientific grounds: SKB attempted to redistribute responsibility. Confined research no longer claimed a monopoly over scientific research and policy making; instead, it appealed, with varying success, to people's sense of responsibility for the common good. Early in the twenty-first century, the company managed to persuade several municipalities to express willingness to share this responsibility (Elam and Sundqvist, 2007).

4.7 Concerned groups turn to science

Nevertheless, SKB's consensus-making efforts and attempt to include laypeople in the discharge of responsibility was met with suspicion and resistance. Laypeople and their advocates had learned to mistrust information provided by nuclear agencies, even when they seemed above suspicion technically and morally (cf. Callon et al., 2009, p. 14). In the anti-nuclear and the environmental movements, there was strong opposition to SKB's proposal for underground storage, KBS-3, and to how the company supported its views by presenting more-or-less tendentious and made-up scientific results as unquestionable facts. The company's information campaigns were characterized as distorted, biased, and dishonest (Åhäll, 1986; Holmstrand, 1987). The distribution of responsibility was also deemed defective, as the opponents did not think that responsibility should rest with a private company owned by the nuclear energy industry, which would inevitably base its actions on principles that differed completely from those required to address serious environmental problems. The anti-nuclear movement argued that the state should assume responsibility for nuclear waste as soon as possible, to enable public control of the process (Bildström, 1997). The government's response was to assert that it considered the distribution of responsibility to be appropriate and based on the sound principle that the producers of nuclear waste should take responsibility for managing it. Moreover, state authorities had input into the process and the government retained the right to approve or reject SKB's proposals for final storage. The government was satisfied that the process being followed was democratic (Larsson, 2000).

The issue of responsibility emphasized the divide between laypeople, politicians, and experts. The developing dynamics of the anti-nuclear movement, the Swedish government's approach, and the nuclear industry's methods together led to the conflict. While the government and the nuclear industry considered SKB's suggestions appropriate, laypeople representing the anti-nuclear movement opposed all forms of underground storage, while struggling to find an alternate disposal method. The final objectives of both parties were clear: on one hand, SKB and the authorities urged maintaining nuclear waste management as a political-3 issue (i.e., articulating the problem as an issue of common good by attracting the interest of several Swedish municipalities) and, on the other hand, laypeople were struggling to breach the citadel of biased confined research and to integrate their own technopolitical actions into the debate on radioactive waste (political-4). While the intentions remained the same, we cannot say the same about the epistemic status of the opposing parties. On the contrary, it became apparent that several steps of the sociotechnical quadrille had been taken, and SKB was now employing social methods while the concerned groups began basing their arguments on technoscientific evidence. Such a redistribution of epistemic identity produced an extraordinary result. Those (e.g., researchers in the wild and the anti-nuclear movement) who had earlier accused SKB of being undemocratic when it carried out geologically motivated test drillings to find the most suitable bedrock, now accused it of neglecting geological and hydrological factors and of being prepared to base its choice of final storage site on the amount of political opposition the site would attract. SKB's critics maintained that the new focus on political acceptance rather than geological conditions made the company untrustworthy and dangerous (Holmstrand, 1990; Avfallskedjan, 2001).

In the 1980s, when SKB had made great efforts to find the most suitable bedrock, critics in the anti-nuclear movement and local resistance groups had strongly opposed this on the basis that all underground storage of nuclear waste was unjustifiable for geological reasons. Now, however, organizations that had previously opposed underground storage raised questions of the quality of bedrock in relevant municipalities, discussed alternative sites on geological grounds, and accused SKB of neglecting geological and hydrological factors. In contrast to the 1980s, the anti-nuclear movement was discussing geological and hydrological criteria in a positive way. Instead of simply condemning all underground storage, it pointed out that the most suitable geological formations were in inland areas with slow subsoil water flow (Holmstrand et al., 2002). SKB and the anti-nuclear movement had now switched positions, but there was nothing in the sociotechnical quadrille that prevented them from switching back again if they considered it beneficial. However, and according to the logic of the quadrille, it was absolutely out of the question that they could appear to occupy the same position at the same time.

It currently appears that consensus has been reached, at least in some places, as the municipalities of Oskarshamn and Östhammar have decided to accept a final repository for nuclear waste within their boundaries (Elam and Sundqvist, 2007). Once SKB had come to an agreement with these two municipalities, it was hard for the national environmental movement to object to these municipal decisions on democratic grounds. Nevertheless, this does not imply technopolitical closure regarding nuclear waste management. On the contrary, as we will demonstrate in the following section, the divide between the anti-nuclear movement and the nuclear energy industry remains. The anti-nuclear movement's intention was clearly to cast doubt on the coastal sites, Oskarshamn and Östhammar. Thus, even SKB's opponents reconfigured their positions, by adopting scientific arguments regarding geology and hydrology.

4.8 From political-1 to political-4

The previous discussion of the shift in the technopolitical balance between confined research and concerned groups as articulated in the public debate from 1950 to the early twenty-first century indicates a radical deconstruction of nuclear waste management and the co-construction of the implicated actors, enforced by the anti-nuclear movement and research in the wild. From total absence from the debate, concerned environmental and anti-nuclear groups in Sweden succeeded initially in deconstructing the actions and efforts of the confined nuclear power industry. They also succeeded in turning the nuclear waste management issue from a purely scientific and technical issue into an open technopolitical issue involving not only the expertise and authority of confined scientific actors, but also the knowledge produced by research in the wild (from political-1 to political-4). Concerned groups managed, in a way, to become co-constructors of radioactive waste management by challenging scientific investigations, measurements, and assessments and by counter-proposing alternative methods. This shift was also to transform their epistemic status. Their involvement and entrenchment in the controversy mirrored their efforts to have their knowledge production recognized, which also legitimized their existence (cf. Callon et al., 2009, p. 105). In the following, we will analyze how concerned groups have become key actors in resolving the nuclear waste management issue over the last decade. We will also demonstrate how confined research, research in the wild, and their mutual antagonism helped reconstruct technoscientific knowledge and the KBS project, making radioactive waste into a political-6 issue.

5. The environmental movement's efforts to reconstruct the final storage of spent nuclear fuel

The fact that SKB had managed in 2002 to persuade two Swedish municipalities to accommodate a final repository for spent nuclear fuel implied a concrete consequence for the anti-nuclear movement: they could no longer question the democratic and dialogic grounds of KBS-3 implementation. When the company and the municipalities' elected representatives came to an agreement, it was impossible for the Swedish anti-nuclear movement to denounce the cooperation as democratically illegitimate. This reconfiguration produced extraordinary results. The quadrille was on and the concerned parties were to reassess their steps. An immediate result of the agreement between SKB and the municipalities of Oskarshamn and Östhammar was that the public debate concerning spent nuclear fuel was considerably toned down. As demonstrated, several environmental and anti-nuclear groups articulated the results of their research in the wild in the public debate on nuclear waste management, strongly challenging technoscientific practices and proposals of the nuclear power industry (e.g., the KBS-3 project). However, the public debate that, from the late 1960s to the early twenty-first century, had democratized the issue of radioactive waste storage by opening the citadel of technoscience to various concerned groups was to be replaced by another form of technopolitical strategy, mainly expressed in the work and research of the environmental movement. Such research was no longer "in the wild," but instead was based on purely technoscientific practices and principles.

5.1 Concerned groups in new roles

Indeed, the environmental movement had developed a new strategy by the end of 2004, and at the beginning of 2005, resistance to KBS-3 entered a new phase. Two new non-profit, non-governmental organizations began operations: Miljöorganisationernas kärnavfallsgranskning (MKG; the Swedish NGO Office for Nuclear Waste Review) and Miljörörelsens kärnavfallssekretariat (MILKAS; the Swedish Environmental Movement's Nuclear Waste Secretariat). The first organization, MKG, consisted of the Swedish Society for Nature Conservation, Fältbiologerna (Field Biologists), and several local resistance groups in the regions that were candidates for hosting final repositories. The second organization, MILKAS, was founded by the national anti-nuclear group Folkkampanjen mot kärnkraft-kärnvapen (FMKK; the Swedish Anti-nuclear Movement) and Miljöförbundet Jordens Vänner (MJV; Friends of the Earth, Sweden). Both MKG and MILKAS are financed by the state Nuclear Waste Fund, a governmental authority that receives and manages the fees paid by the nuclear power companies and owners of other nuclear facilities in Sweden for developing safe methods for storing nuclear waste (MKG, 2004). According to a law enacted in 2003, environmental organizations were entitled to economic support from this fund, despite SKB opposition (MKG, 2004). Thereby, the role of the critics was officially sanctioned and institutionalized.

Despite what had been the case for environmental organizations before this century, MKG and MILKAS did not mainly seek to create negative publicity for KBS-3 through mass media and information campaigns. On the contrary, MKG and MILKAS aimed to persuade state authorities, through strictly scientific reports, letters, and investigations, that KBS-3 constituted an unjustifiable project based on shaky scientific grounds. Concerned scientists, among them nuclear engineers and geologists who had leading roles in both organizations, claimed to be able to reveal and scientifically analyze the shortcomings of KBS-3. MKG and MILKAS conducted their own scientific investigations of geological conditions, invited foreign scientific experts to workshops, and continuously cited the latest scientific results concerning groundwater flows, corrosion, radiation, and earthquakes during the ice age (Mörner 2009a; MKG 2008a; MILKAS 2007; Åhäll 2006; MKG 2005a; MKG 2005b). They maintained that their criticism of KBS-3 found strong support in refereed scientific papers published in peer-reviewed journals. In contrast, SKB's scientific reports were internally published and not subject to independent scientific scrutiny. In other words, MKG and MILKAS claimed to be maintaining a more scientific profile than were their SKB-employed colleagues. The research conducted by MKG and MILKAS was highly sophisticated scientific work; in addition, they sought support in scientific reports and scientifically reviewed specialist articles that they made available on their websites (MKG 2009a; MKG 2009b; MKG 2008c; MKG 2007a). These articles were essentially incomprehensible to anyone other than researchers familiar with fields such as geology, physics, and chemistry.

This research material was not posted for the sake of the public, but to acquire as much scientific credibility as possible in promoting their viewpoints. They now exploited the natural sciences' monopoly on truth, in the same way as SKB had done since the mid 1970s, but maintained that their use was undistorted by conflicting interests, in contrast to what the history of SKB bore witness to. MKG and MILKAS had abandoned their reluctance to engage indisputable scientific facts and instead enlisted them as allies by directing them against SKB's scientific reports ("polluted by financial profit interests"). Suddenly, the environmental organizations had become eager advocates of pure, undistorted science that could serve the common good. This time, it was not journalists, activists, or local residents who would unmask KBS-3, but scientists producing articles in refereed journals of high quality and credibility.

5.2 Getting inside and reconstructing alternative solutions

Since 2005, both environmental organizations have participated in meetings with municipalities, initiated debates with state authorities, participated in SKB's information meetings for concerned parties, and joined in workshops arranged by responsible governmental ministries. MKG and MILKAS were clearly no longer pariahs in seeking a resolution to the radioactive waste storage problem. Their aim was to become an obligatory passage point[4] in nuclear waste management, and they sought to accomplish this by establishing themselves in the decision-making processes by "constructing" an alternative final repository concept and by continuously and directly addressing the government through letters, writings, investigations, suggestions, and the like (Mörner 2009a; Mörner 2009b; Hulthén et al 2010; MKG 2008a; MKG 2008b; MKG 2008d; MKG 2006a). Their criticism of KBS-3 focused on their allegation that it was impossible to keep spent nuclear fuel separate from the biosphere for over 100,000 years, something that the KBS-3 project considered necessary. MKG had several reasons for making this claim. MKG emphasized that the copper encasements would corrode even in oxygen-free water, as scientists at the Royal Institute of Technology in Stockholm had proved (MKG, 2007a). When corrosion occurred, contaminated groundwater would eventually reach the earth's surface. The fact that SKB had situated the final repository in two potential coastal sites, with municipal support, unmasked an emergency plan for a potential environmental disaster. In inland Sweden there were geological formations where groundwater flows were considerably slower. According to MKG and MILKAS, if a repository were, against all reason, constructed following the KBS-3 concept, it ought to be located in an inland region with slow groundwater flows. The sites that SKB had selected were therefore the worst available.

The anti-nuclear movement's new strategy was based solely on discussing technical and scientific matters in the same way as SKB had for decades. In other words, at a time when agreement had been reached between SKB and two municipalities (a political decision), concerned groups and researchers in the wild aimed to reconstruct better technological alternatives instead of merely deconstructing existing ones. What alternatives for handling spent nuclear fuel did MKG and MILKAS advocate? Answering this question also involves discussing why the environmental movement split into two main organizations regarding the issue in the first decade of the twenty-first century.

5.3 Deep drill holes and trust in natural barriers

MKG, which received strong support from nuclear engineers at Chalmers Institute of Technology and from the Swedish Society for Nature Conservation, chose not to explicitly question nuclear power as an energy source, but instead to focus entirely on what method was most suitable for the final disposal of spent nuclear fuel (MKG, 2008a; Swahn, 2008). In line with SKB, MKG sought a technological solution. Although MKG claimed to be open to several disposal methods, it concentrated all its efforts on promoting a method for the final storage of spent nuclear fuel in 3–5-km-deep drill holes in bedrock (MKG 2008a; KASAM 2007; MKG 2007b; Lihnell Järnhester et al 2006; MKG 2006b; MKG 2006c; MKG 2006d). Accordingly, MKG financed an investigation of the potential to apply this method in Sweden, conducted by a geology professor, Karl-Inge Åhäll, who had been involved in nuclear waste issues for several decades. Åhäll argued that the method was promising and that the necessary technology seemed to be within reach, although some problems remained to be solved (e.g., how the encasements should be placed in their final destinations). Åhäll and MKG were optimistic about the approach because the encasements would be stored at a depth at which the groundwater was stratified by density and a saltwater barrier prevented

[4] An obligatory passage point constitutes a control station within a technoscientific dispute that must be passed in order for specific actors to accomplish their interests (Callon 1986, p. 205; see also Galis 2006, p. 27).

radioactive substances from reaching the earth's surface, even if the encasements leaked (Åhäll, 2006). In other words, MKG's concept rested on the theory that one natural barrier could guarantee safety rather than several artificial ones.

MKG's support of deep underground storage of radioactive waste constituted somewhat of a paradox. MKG represented a significant part of the environmental movement that had previously opposed all underground storage of nuclear waste material and had accused SKB of putting its trust in a project (i.e., KBS-3) that relied on an unforeseeable future. This organization now advocated a concept that anticipated storage of spent nuclear fuel in ten-times-deeper bedrock and presupposed complete trust in the ability to predict the stability of nature for what comes close to eternity. The reconstructed underground storage of spent nuclear fuel that MKG supported also presupposed complete trust in the ability of science, above all geology and hydrology, to control nature comprehensively and a strong belief in engineers' ability to find infallible technical solutions to "grand-scale" management problems. Accordingly, MKG's argumentation rested on a strictly positivist understanding of science, a view of nature as fundamentally stable, and a technocratic view of society; people's future health and welfare were entirely dependent on the expert knowledge of scientists and on the competence of future engineers. MKG was committed to science and technological optimism—what SKB had earlier been accused of by environmental organizations.

On the other hand, both MKG and MILKAS maintained that their proposal for final storage was considerably more scientifically and technologically sophisticated than KBS-3 ever had been. Among other things, nuclear transmutation was proposed as a potential future alternative in combination with final storage in deep drill holes. If transmutation made it possible to reduce the volume of nuclear waste, the number of deep drill holes would also be considerably reduced. At the same time as demand mounted to investigate the potential of the deep drill hole concept, MKG asserted that test drillings to locate the most suitable bedrock ought to be conducted for research purposes (MKG 2008a; Lihnell Järnhester et al 2006; MKG 2006c; MKG 2006D; MKG 2007B; KASAM 2007). Important fractions of the environmental movement that previously had opposed all test drilling became now its most passionate advocate (MKG 2008b). To convince the authorities that it was a progressive actor, MKG repeatedly underscored that KBS-3 was a "30-year-old concept" based on "old-fashioned" technology, thereby insinuating that SKB was unprepared to evaluate the potential of contemporary and future technology. On the contrary and according to MKG's critique, SKB wanted to implement a method in which it had invested 30 years of effort. From this point, it was the environmental organizations, not SKB, that became the strategically most prominent interpreters of the notion of perpetual cumulative progress (MKG 2006e; MKG 2006d). It was rather obvious that MKG had changed its attitude towards underground storage; this time they had learned their lesson well, and refrained from discussing political matters, confining themselves strictly to technological alternatives. From this point on, the anti-nuclear movement in general made technopolitics using "purely" scientific means, while SKB made technopolitics using "purely" political instruments (cf. Anshelm and Galis, 2009, p. 278).

Spokespeople for SKB observed that, counter to the environmental organizations' past stance, they were now suddenly advocating an untested method that relied on the everlasting burial of nuclear waste material in the bedrock. SKB also claimed that the proposed method would require at least another 30 years of development, and that it entailed several uncertainties that would make it impossible to remove the radioactive material if something went wrong. SKB warned that the idea of leaving spent nuclear fuel in the bedrock and surrendering all control caused them unease. Although SKB did not seriously consider the concept of deep drill holes, responsible state authorities displayed significant interest in the idea, hosting special workshops in which the concept of deep drill holes was addressed. At the same time, the Swedish government maintained that SKB had to investigate alternative methods to KBS-3, especially the deep drill hole method. The government also criticized SKB for ignoring this demand in their R&D reports (Miljödepartementet 2008; Miljödepartementet 2009; KASAM 2007). Since MKG had received this kind of response from state authorities and the government, their efforts may be perceived as somewhat successful. Moreover, MKG's research inspired the submission of a motion in parliament; picking up on MKG's rhetoric, the Swedish Green Party proposed a thorough investigation of the deep drill hole disposal method (Swedish Parliament, Bill 2006/2007:2406).

As already mentioned in this section, despite the fact that the environmental organizations participating in MKG represented 168,000 members the public debate in the media and via information awareness campaigns declined considerably in the first decade of the twenty-first century. MKG had learned its lesson and was now fighting KBS-3 using the nuclear energy industry's own arsenal. One consequence of this was that MKG, like SKB, kept the question of whether or not nuclear power was a justifiable energy source separate from that of how spent nuclear fuel should be handled, a separation that had formerly been completely rejected by the environmental organizations. This implied, somewhat ironically, that MKG and its members—who had previously opposed nuclear power in general—were seeking to solve the most severe safety problem related to nuclear power production: reconstructing a promising method for underground storage, if suitable geological formations could be located (which MKG implied was possible). At least hypothetically, this meant that MKG could have removed a large obstacle to continued nuclear power production, permitting a new generation of nuclear reactors to be launched. An important fraction of the environmental movement was actively involved in solving the problem of the final storage of spent nuclear fuel, unintentionally to the benefit of the nuclear energy industry, even though SKB declared it was capable of developing its own solutions.

5.4 Dry rock deposit and resistance to underground storage

This constitutes one reason why the anti-nuclear movement is divided into MKG and MILKAS. In sharp contrast to MKG, MILKAS maintained that taking a stand for or against nuclear power as an energy source and working for the safe storage of spent nuclear fuel were two issues that should never be separated. To construct a solution for final storage, as MKG had, was unjustifiable for MILKAS. MILKAS also shared MKG's trust in science and employed scientific publications and experts as strong voices opposing KBS-3. It maintained, however, the early anti-nuclear movement's conviction that nuclear waste should not be stored underground in a final storage repository inaccessible to humans (Hulthén et al., 2010). The reasons for this position were the same geological and tectonic objections that MILKAS had raised against KBS-3. MILKAS underlined that geology was not a science that could foresee the future, and that nobody could reliably predict what could happen in an underground storage repository when earthquakes occurred, giving rise to fissures and faults, during any ice ages that might occur over the more than 100,000 years the spent nuclear fuel would remain deadly to almost all forms of life (Ahlin and Dörvaldt, 2007; KASAM 2007). Geologist Nils-Axel Mörner, who supported MILKAS, claimed that his research had proved that it was irresponsible to keep nuclear waste in the bedrock and that the notion of a final repository was a theoretical construct unsupported by geology. Accordingly, MILKAS opposed every idea for a final storage repository and questioned as unrealistic MKG's and SKB's objective of relieving future generations of the burden of radioactive fuel by permanently depositing it underground. The idea of a final storage in bedrock was perceived as insane. In contrast, the organization advocated storing nuclear waste to be accessible to future generations in what were called "dry rock deposits" (Hulthén et al 2010; Mörner 2009a; Mörner 2008; MILKAS 2007).

In this method, nuclear waste should be stored in carefully sealed deposits above the groundwater in rock formations that are continuously self-drained. The idea is that spent nuclear fuel should be conveyed transparently and responsibly via an open process, not simply dumped into a mutable and unpredictable nature. MILKAS justified its approach by invoking the potential of future technology, declaring that future generations would probably develop technological methods superior to KBS-3. Accordingly, it was important not to deprive future generations of their options. For example, MILKAS emphasized that nuclear transmutation and other future technologies might make it possible to neutralize or reduce the volume of spent nuclear fuel. According to MILKAS, simply dumping radioactive waste inaccessibly into the bedrock, as SKB and MKG advocated on the basis of strictly limited knowledge, would be an irreparable mistake. Correspondingly, MILKAS argued that all production of nuclear waste material must immediately stop, so as not to aggravate future problems of nuclear waste management (Mörner 2009a; Hulthén et al 2010; Ahlin and Dörvaldt 2007; KASAM 2007).

5.5 The redistribution of epistemic status and the strengthening of technoscience

The analysis of the efforts of MKG and MILKAS in the first decade of the twenty-first century has demonstrated that a former social protest group with limited technoscientific competence can transform itself into a significant group of scientific knowledge carriers and sophisticated technology producers. In many respects, the opposing sides have switched positions, bringing SKB closer to society (via communication with municipalities) while MKG and MILKAS became secluded in erudite research isolated from the public. In the case of MKG, this has led to a distancing from the social and ideological interests they were initially representing, i.e., struggling against nuclear power in general and condemning all underground storage of radioactive waste. When something becomes a technopolitical-6 issue, the involvement of concerned groups with technoscience entails the risk that these groups may abandon their ideological origins and slide into an introverted relationship with technoscientific practice. In the studied case, there has been a reversal and an absolute redistribution of epistemic status that strengthens the confinement of technoscience. The efforts of MKG and MILKAS to become legitimate and influential participants in the nuclear waste controversy by attempting to make their agendas, methods, and interests scientifically credible reminds us in STS that we have only been following one-half of the quadrille, that is, the deconstruction and, in the best case, the co-construction of technoscientific controversies. We have been following scientists and technologists, scientific theories and technological artifacts (i.e., political-1, -2, and -3 issues), and we have normatively advocated the recognition and participation of concerned groups in the technopolitical arena. What we have neglected, apart from analyzing what "politics by other means" stands for, is investigating the impact of technopolitics on the identity and practice of groups concerned by it.

In this paper and in our empirical case, we have attempted to analyze the issues technopolitics addresses and, in turn, how these issues shape technopolitics (cf. Latour, 2007, p. 819). Over a period of fifty years, the Swedish environmental and anti-nuclear movements were not only engaged with but also exposed to the technopolitics of nuclear waste management. Such engagement challenged the perception of nuclear waste management as a purely technoscientific matter based on indisputable scientific facts (cf. Callon et al., 2009, p. 16), making it a technopolitical issue that was recomposed and reconstructed by the conflicts, negotiations, and research of diverse groups of legitimized experts, laypeople, ordinary citizens, and researchers in the wild. This had a very important effect on the identity and epistemic status of the environmental movement. On one hand, we can detect the detachment of MKG from its initial standpoint, namely, opposing nuclear power. On the other hand, both MKG and MILKAS, by interacting with and including numerous scientists in their organizations, shifted both their profiles and lines of argumentation, making them more science based and abandoning earlier activist tactics.

It is important to stress, however, that MKG and MILKAS never became technocratic. They kept their democratic organization structures, stayed close to participatory ideals, and appeared as activist groups that adopted the scientific means they needed to profoundly criticize and challenge the KBS-3 concept. At the same time, they reconstructed and advocated alternative solutions for storing spent nuclear fuel. In other words, they made politics using scientific means. In 2002, when SKB persuaded two municipalities to declare their willingness to host a final repository for spent radioactive waste, the anti-nuclear movement lost the fight over democratic legitimacy (political-4) to SKB. If nuclear energy opponents still hoped to block KBS-3 (which they did), they had to resume the fight on technoscientific grounds (political-6), since this was where the issue could be reopened for thorough scrutiny and new negotiations.

6. Conclusion: Dancing the quadrille

We have treated the management of spent nuclear fuel as a sociotechnical quadrille: four involved couples representing various social, ideological, and technoscientific interests (i.e., concerned groups, concerned scientists, professional politicians, and the nuclear power industry) waver between science, policy-making, and activism, pressed by the urgency of the accumulating spent nuclear fuel. This sociotechnical quadrille implies that, while scientists and technicians sought to display awareness of social norms (both governmental and non-governmental), concerned groups sought to bolster their scientific credibility to advance their engagement and proposals. Dancing the quadrille has promoted the development of one of the most advanced methods for the final storage of spent nuclear fuel in the world, the KBS-3 concept. The involvement of concerned groups, such as the anti-nuclear and environmental movements and their advocates (e.g., journalists and academics), as well as the work of the nuclear energy industry, authorities, and parliament have considerably shaped the KBS-3 project.

We have actually identified three major forms of knowledge/action enacted by concerned groups (see Table 2): criticizing/deconstruction, bridging the lay–expert epistemic divide/co-construction, and taking leadership in technological development/reconstruction (cf. Jasanoff, 1997). Over approximately four decades, concerned groups in Sweden have exerted considerable influence on the nuclear energy industry and on governments and state authorities, vehemently opposing state authorities and the national nuclear industry.

This dynamic has been crucial for the development of a specific Swedish concept for the final underground storage of spent nuclear fuel. By extensively molding public opinion—including vast media debate, extended written communication with state authorities, protest marches, and civil disobedience—concerned groups essentially forced the nuclear energy industry to invest in technological development and scientific research to refine the KBS-3 concept and resolve the shortcomings and uncertainties that critics did their best to detect. Likewise, state authorities and governments have been forced to pass new laws and increase the security oversight of technoscientific developments, as the scrutiny of KBS-3 by concerned groups has evoked reconsideration of the sufficiency of the technological and scientific knowledge base.

Meanings of "political"	What is at stake in each meaning	Our case	Phases
Political-1	New associations and cosmograms	Nuclear waste management as a risk issue—new associations between society, safety, and energy production	
Political-2	The public and its problems	The creation of critical, concerned anti-nuclear groups	De-construction
Political-3	Sovereignty	The Swedish state and nuclear industry attempt to provide guarantees regarding nuclear waste management	
Political-4	Deliberative assemblies	Concerned groups claim the right to participate in configuring nuclear waste management	Co-construction
(Political-5)	(Governmentality)		
Political-6	The re-confinement of research in the wild	Concerned groups produce technoscientific solutions	Reconstruction

Table 2. Dancing the quadrille

In this way, concerned groups have played leading roles in configuring the technopolitics (political 2–4) of the Swedish concept for the final storage of spent nuclear fuel. By severely criticizing KBS-3 (de-construction), concerned groups initially dragged the nuclear energy industry and parliament into the dance by repeatedly revising, qualifying, and refining the proposals for constructing a repository for the final storage of spent nuclear fuel. By deconstructing the KBS-3 concept, these groups identified the points at which the proposed concept had to be improved. Concerned groups thus came to serve as a prominent critical voice in a process of technological development. The Swedish nuclear waste management case is interesting not only for its ambitious and technologically sophisticated project for disposing of spent nuclear fuel, but also for the critical and influential involvement of concerned groups. The anti-nuclear movement and its advocates did not limit themselves merely to democratizing decision-making regarding nuclear waste management through deliberative procedures (as was the case in several other major nuclear powers); they also claimed the right to participate in technological processes and developed, on their own, highly sophisticated scientific proposals.

Their engagement with technoscience unintentionally helped improve or co-construct KBS-3, which has become a reference concept for investments in sustainable nuclear power in the EU. As Jasanoff (1997) notes, there are hidden costs in relying on concerned groups, and some strategic choices can raise questions as to whose values and interests are represented by concerned groups (p. 587). In our case, concerned groups have increasingly come to perform a critical task that has been necessary for the progress of KBS-3, a task that neither the nuclear energy industry nor governmental authorities has succeeded in discharging, mainly due to profit interests or corporative alliances. Considered in this way, the development of the final storage of spent nuclear fuel stands out as a consequence of four decades of heated technopolitical struggle and not as an ex parte result of either the Swedish political culture, characterized by consensus-making and cooperation, or the achievements of purely technoscientific work. Concerned groups have, in sharp contrast to their intentions, been involved in co-constructing KBS-3.

The involvement of concerned groups in constructing a more secure storage method has exacted a significant cost: through this process, concerned groups have contributed greatly to increasing the political legitimacy of nuclear power. This occurred partly because the crucial efforts of concerned groups to make the KBS-3 a more secure concept enabled the right-wing government to repeal the moratorium on building new nuclear reactors, and partly because it enabled the EU to speak about sustainable nuclear energy. These ambiguous and unforeseeable consequences bear witness to the complex, important, and often paradoxical roles that concerned groups play in technopolitical controversies—something that deserves more attention from the STS field. In any case, one conclusion that can be drawn from the analysis of the empirical case is that technopolitical controversies cannot be satisfactorily understood without a detailed analysis of the role and epistemic status of concerned groups.

Another conclusion of this study concerns how technopolitics have profoundly shaped and reshaped all concerned groups involved in constructing Swedish nuclear waste management. The nuclear energy industry was taken to task by individual whistleblowers in the early 1970s for starting reactors without having developed reliable methods for the long-term storage of spent nuclear fuel. Most of these critics were oppositional scientists, most of them physicists, who aimed to deconstruct the truth claims and claimed scientific hegemony of the nuclear power industry. However, in 1972 a broad and widespread anti-nuclear movement achieved a sensational breakthrough in the Swedish media. As a result, between 1973 and 1975, Dagens Nyheter, the largest daily Swedish newspaper, published more than 150 full-page debate articles questioning nuclear energy (Anshelm, 2000, pp. 119–120). These articles were written by journalists, artists, engineers, physicists, researchers, politicians, and laypeople of all sorts. This “insurrection of subjugated knowledges” constituted loosely coupled activist networks and concerned groups that, supported by oppositional experts on nuclear physics, struggled against the nuclear power society and against the lack of reliable methods for managing the nuclear waste being produced. Their aim was to enlist public opinion on their side with the help of arguments, campaigns, public meetings, and protest marches.

During this process, the issue of spent nuclear fuel storage was transformed into a question of deliberative procedures (political-4). Purely technological and scientific questions were integrated into or even displaced by social and moral questions. What constituted a good society? What represented responsible behavior vis-à-vis coming generations? Who had the right to take huge risks on behalf of others? What were the moral and social consequences of making a whole society dependent on nuclear energy experts? Was the democratic deficit a morally acceptable price to pay for a potential rise in the material standard of living?

At the same time as the national anti-nuclear movement was gaining considerable influence among citizens and political parties (which resulted in a national referendum in 1980 on the future of nuclear power in Sweden), local resistance groups picked their own fights. In the 1980s, they opposed the search for a location for the final storage of spent nuclear fuel in their home districts. These groups engaged in civil disobedience and occupied workplaces to stop test drillings. Local resistance groups also consulted oppositional researchers, though they were primarily interested in protecting the local character of their struggle.

Early in the twenty-first century, the concerned groups' close and persistent scrutiny helped develop KBS-3 into one of the world's most elaborate and technologically reliable methods for storing spent nuclear fuel. This made it possible for two Swedish municipalities to declare that they were willing to host a repository. The great influence of concerned groups in co-constructing the repository forced them to change their own position in the dance yet again, to take a new step in the quadrille. For concerned groups to maintain their influence, they had to transform their (de-)(co-)constructing efforts into reconstructive ones. When the ideological struggle was lost, the fight over technological and scientific facts remained. After all, in times characterized by massive accumulations of information, concerned groups are expected to act as carriers/producers of technoscientific knowledge (Jasanoff, 1997, p. 589). Thus, an issue that since 1972 had been treated as social and moral was again transformed into a technoscientific one. The quadrille was turning again. There was no longer any room to lobby against or question the democratic legitimacy of KBS-3.

Instead, concerned groups, such as MKG and MILKAS, began to strongly oppose the scientific basis of KBS-3, adding another step to the quadrille. Moreover, they advocated alternative technological solutions for storing spent nuclear fuel. These concerned groups were dominated by engineers, experts, and scientists who, with the financial support of the Nuclear Waste Fund, organized seminars, invited international experts, ordered specific technoscientific investigations, produced scientific reports, discussed contemporary scientific articles on geology, hydrology, and corrosion, and critically assessed SKB's research and development reports and state investigations. They seldom courted public opinion through the media, but instead concentrated their pressure on state authorities, the government, municipalities, and the nuclear power industry. MKG and MILKAS acted as minor specialized sub-organizations, supported by major environmental groups. Their key task became finding alternative solutions to technologically complicated problems and demonstrating that more efficient technologies were available than that on which KBS-3 was based (political-6). Thereby, a necessary change in strategy changed the technopolitical configuration and completely reshaped how concerned groups exerted influence.

One question that emerges is how this change in strategy has influenced public support for the concerned groups. At the same time as the concerned groups gradually became more scientifically oriented and dependent on experts, public resistance to nuclear power and public fears regarding the long-term consequences of nuclear waste dramatically decreased. Did this shift signify the capitulation of subjugated knowledges and the domestication of research in the wild? Has the quadrille completely seduced the concerned groups? Which are the risks regarding the transformation of concerned groups from enactors of civil disobedience to "interactional or contributory experts" (cf. Collins and Evans, 2002, p. 254)? These are not easy questions and there are several obvious but also complicated answers. It is reasonable to assume that the transformation of concerned groups into core technoscientific actors, abetting the transformation of nuclear waste management into an absolute scientific and technological issue, has contributed to the re-confinement of technoscience and the disappearance of the nuclear waste issue from public debate. The public is not any longer invited to participate and make a difference. Once again, experts and interactional experts (including concerned groups such as MKG and MILKAS) are dealing with the public problems while the rest of the demos is left to passivity. Any necessary actions are to be undertaken by minor, specialized sub-organizations, reemphasizing the technopolitical democratic deficit. Whether this development is unavoidable or even, in some respects, desirable is hard to tell. However, it provides an excellent example of how concerned groups are shaped by technopolitics and a strong argument for conducting more in-depth studies of the phenomenon within the realm of social sciences.

The present case reminds us that the sociotechnical quadrille is all about balance. Technopolitical controversies contribute to the shifting of epistemic status, but this does not mean that all parties to a controversy will maintain their epistemic autonomy. In our interpretation, political-6 should refer to what Callon et al. (2009) call collaborative research that explores and redistributes epistemic identities (p. 265). Such research, however, should guarantee the equal and balanced participation of both the public and other interests in the configuration of technopolitics. This, in its turn, will guarantee the (re-)(co-)constructing of technoscience in progressive directions.

7. References

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21

Assessment of the Vulnerability Potential for an Unconfined Aquifer in Konya Province, Turkey

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1. Introduction

Only two to three percent of total water on earth is fresh water, and groundwater constitutes a significant portion of the fresh water resources. This scars and fragile resource is under the risk of degradation in both quality and quantity in many parts of the world.

Three major activities cause large quantities of human and industrial waste disposals, and hence, pose serious threat to the groundwater resources. The first of these activities is excessive use of fertilizers, pesticides and automation in agricultural areas. The second one is unregulated discharge of natural and artificial chemical substances to the environment. Finally, excessive pumping and improper management of aquifers result in reducing the pumping potential and degrading the water quality.

In order to mitigate the influence of such detrimental effects, precise delineations of the aquifer system and of the hydrogeological setting in concern areas are of primary importance. In those areas, vulnerable to such adverse environmental problems, two types of remedies have been proposed. The first type is called the reactive approach which involves aquifer remediation after the contamination occurred. The preferred proactive approach, on the other hand, calls for taking the necessary measures before the contaminants reach the groundwater surface. The reactive method of aquifer remediation suffers from unsatisfactory results since it is impossible to obtain 100% purification.

The proactive method mainly involves identifying the pollutant source at an early stage and assessing the potential danger it poses to the ground water system. Geographic Information System (GIS) has been proven as an effective tool for this purpose. As in this research, Chitsazan and Akhtari (2009) and Awawdeh and Jaradat (2009) determined the risk of pollutants to contaminate the groundwater, with the help of GIS. In recent years the increase in the popularity of applying environmental criteria in national planning and management has provided a wide range of scientific approaches to determine the best location of hazardous wastes to be land filled (Abessi and Saeedi 2009).

This method is applied in many countries, such as Palestine (Baalousha 2006) Ethiopia (Tilahun and Merkel 2009) Tunisia (Saidi et al 2009), both to determine the suitable solid waste areas and to dispose other contaminants. For example, it is a preliminary assessment of the possible environmental impact of a proposed landfill facility for the city of Jammu City in India (Nagar and Mirza 2002). The researches on aquifer vulnerability is increasing

day by day in Turkey (Nalbantcilar et al 2009, Sener et al 2009, Pusatli et al 2001) and that shows this research area is inevitable.
This work presents the results of a study carried out for the assessment of ground water contamination due to a solid waste deposition side near Konya, Turkey. The well-known DRASTIC model (Aller et al 1987) has been employed to prepare ground water vulnerability maps in an unconfined aquifer near Konya, Turkey. This aquifer supplies most of the water needs of Konya for municipal drinking and irrigation. DRASTIC utilizes a technique of developing seven layer hydrogeological maps consisting of depth to ground water, net recharge, and soil and aquifer units, topography, influence of vadose zone and hydraulic conductivity. The overlaying DRASTIC index map indicates the vulnerability degrees at various parts of the aquifer system.
Application of the DRASTIC index map to the unconfined aquifer of Konya city indicated high degree of vulnerability to contamination at the central part of the aquifer. The vulnerability map was compatible with the distance from the waste disposal site. Finally, this work also showed that while GIS is useful for over laying the hydrogeologic setting maps the DRASTIC model is useful for aquifer preservation.

2. Groundwater pollution in solid waste disposal area

The solid waste disposal site of Konya stores different types of domestic, industrial, medical wastes and so on. During winter and spring, the rainwater may wash out some of these waste materials, and carry them down to ground water system as it seeps through soil and rock layers. The main contaminants are Al, Cr, Fe, Li, and Zn. This study has attempted to quantify the variations/increases in concentrations of such elements in the regions' groundwater bodies. Mixing of the contaminant loaded rain water with the groundwater may result in significant reduction of the ground water quality.

2.1 Pollution control

In the following, we will present the results of pollution control studies in an area having been used for sewage disposal site over 35 years in Konya. This site has an area of approximately 300,000 meter square with an intake of 350,000 tonnes of garbage annually (Fig. 1). In order to monitor and characterize the ground water pollution caused by solid waste disposal site in Konya, a large number of ground water wells have been sampled from representative parts of the unconfined aquifer for a period of two years.
The main geological units or which the disposal site lays are called Sakyatan formation which is a unconfined aquifer with a significantly high permeability (Hakyemez et al 1992, Nalbantcilar 2002, Nalbantcilar and Guzel 2006, Nalbantcilar and Ozdemir 2009 and 2010).
The data having been collected over 50 years by Meteorology General Directorate indicate that this area receives 326.5 millimeter annual average precipitation. Such a rate of rainfall can wash out significant amounts of contaminating ions carrying them down to the ground water table. In order to quantify the degree of pollution, the monitoring wells were selected. This monitoring showed that they were in line with the ground water flow direction from the disposal site towards downtown Konya (Fig. 2). In situ conductivity values were measured in each of the wells, water samples were collected at specific periods and their chemical analysis was performed in the Selcuk University labs by ICP-AES.

Fig. 1. A view from the solid waste area of Konya before rehabilitation.

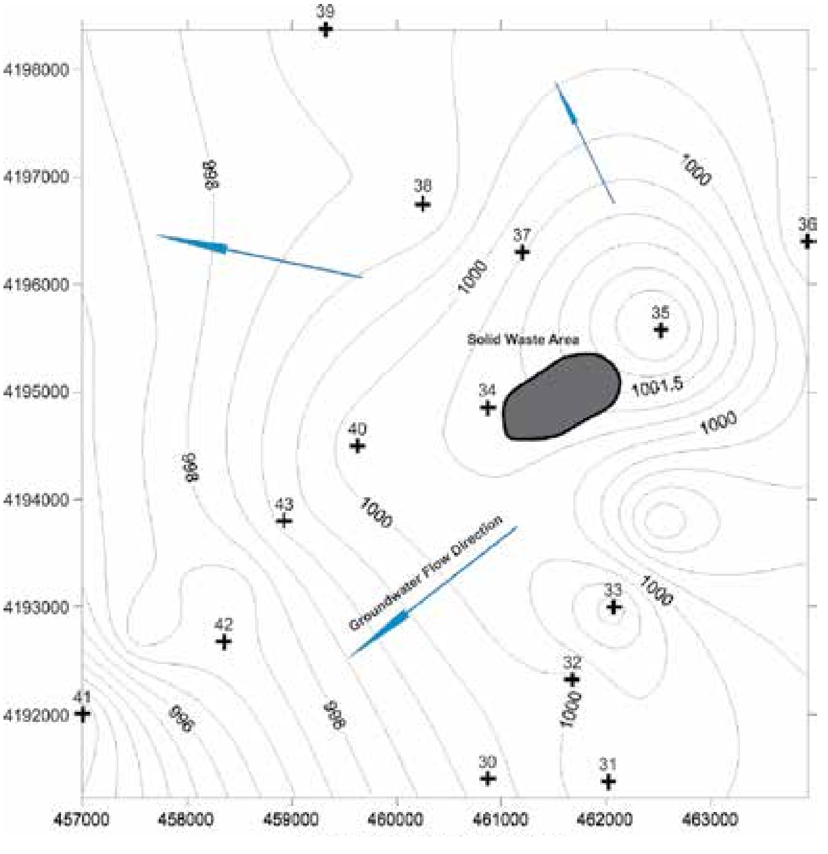

Fig. 2. Groundwater table map (in meters) and flow directions with solid waste area (Nalbantcilar and Ozdemir 2010).

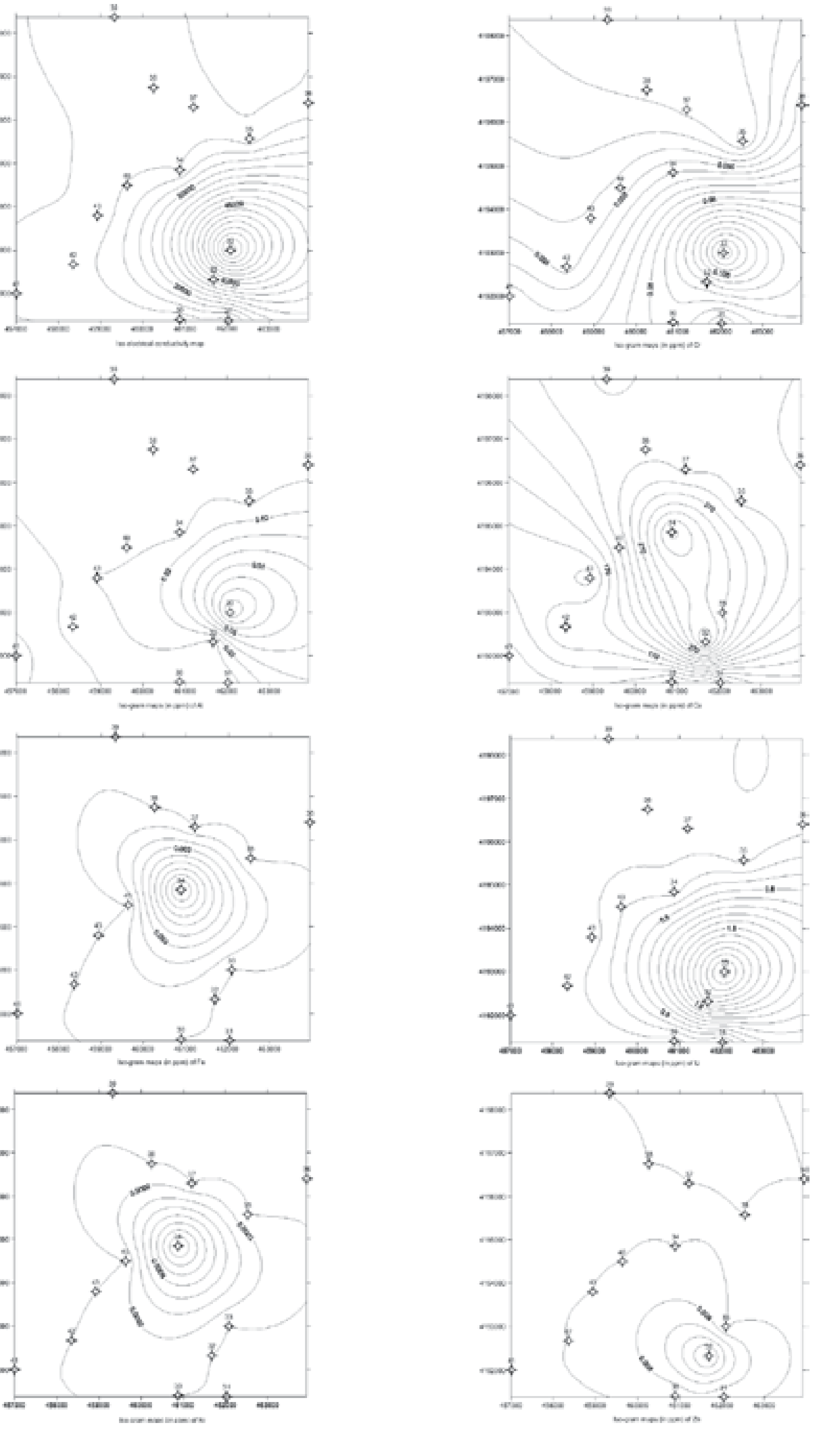

Fig. 3. Distribution maps of conductivity of the water samples and Cr, Al, Ca, Fe, Li, Ni and Zn concentrations (mg/lt) (Nalbantcilar and Ozdemir 2010).

The concentration maps concluded that Cr, Al, Li, and Zn were in high concentrations in the South Eastern part of the study area, Fe, Ni concentrations were more pronounced in the central parts and Ca concentration was dominant in the Southern part. As for the specific local values, well-33 had Ca, Li, and Al, well-32 had Zn, and finally well-34 had Ca, Fe, and Ni as their highest concentration contaminants.
Such findings lead to the conclusion that especially Al and Zn concentrations in the ground water were closest to the disposal site and declined proportionally with the distance from the site. Based on the main conductivity measurement and water sample analysis data, concentration contour maps were prepared for each of the elements (Fig. 3).

3. Drastic method/gis and assessment of contamination

The DRASTIC method developed by US Environmental Protection Agency (EPA) has proven to be a useful tool for assessing ground water contamination potential. The method depends on utilizing seven hydrogeologic parameters: Depth to water table, net Recharge, Aquifer media, soil media, Topography, Impact of the vadose media, and hydraulic Conductivity of aquifer. These perimeters are transferred to GIS media as separate layers. Finally at the bottom, the drastic layer is formed as an integrated summary layer (Fig. 4).

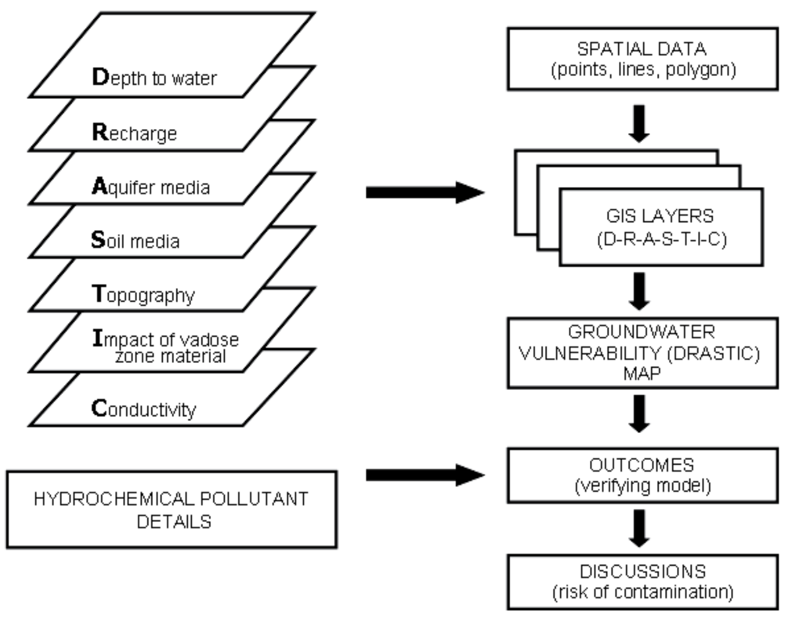

Fig. 4. Technique of DRASTIC with GIS layers (Nalbantcilar et al 2009).

DRASTIC is a numerical ranking system to assess ground water vulnerability in various hydrogeological settings. It is based on generating GIS layers for each of the seven hydrogeologic parameters and at the bottom, most layer is developed as the integrated DRASTIC layer. Each layer is assigned as a relative weight factor ranging from 1-5. The most significant factors have weights of 5 (Depth to water table and Impact of the vadose zone media), 4 (Net Recharge), 3 (Aquifer media and hydraulic conductivity), 2 (Soil media) and the least significant have weights of 1 (Topography).

The DRASTIC layers prepared for Konya in Figure 5 show that when the unconfined aquifer falls into DRASTIC categories of 1-2, it indicates low vulnerability and at 7-8 and 9-10, it indicates high vulnerability.

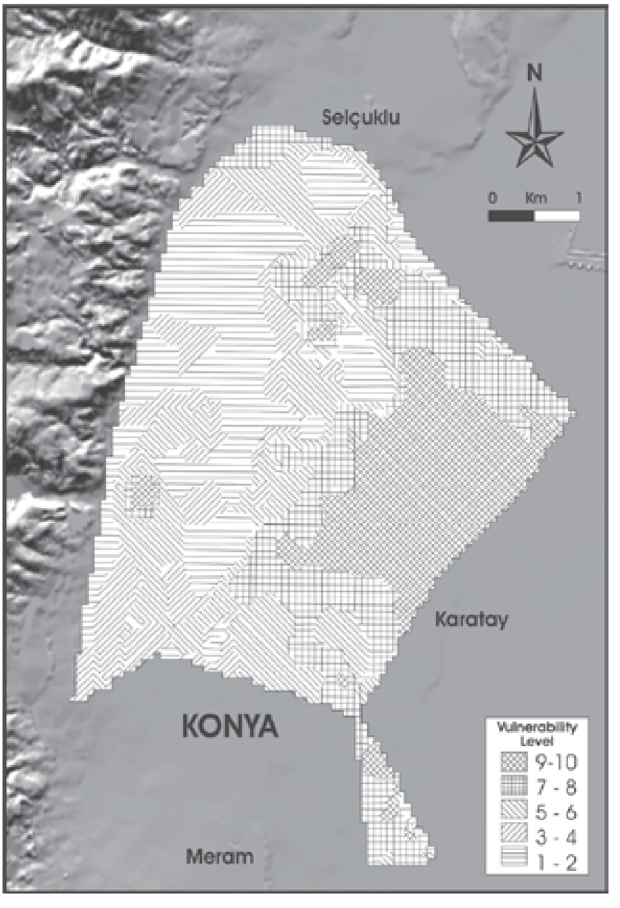

Fig. 5. The DRASTIC index map of the study area.

4. Results and discussions

An extensive sampling and in situ conductivity measurements of a large number of water wells have been carried out in the study area. Based on this data, the elemental concentration contour maps showed that the solid waste site has adversely influenced the ground water quality severely, contaminating the nearby zone. The contamination will get worse unless proactive precautions are undertaken. Therefore, the present situation poses serious threat to groundwater resources of Konya. The drastic map and the current city development plans with the present solid waste sites show that ground water resources are at great risk of contamination.

DRASTIC has proven to be a useful tool for evaluating vulnerability of Konya aquifers as well. Thus monitoring, an evaluation by GIS must continue in the future in this area. In addition, aquifer management techniques must be employed diligently.

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Part 4

Leachate and Gas Management

22

Sustained Carbon Emissions Reductions through Zero Waste Strategies for South African Municipalities

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1. Introduction

Greenhouse gas (GHG) emissions per person from urban waste management activities are greater in sub-Saharan African countries than in other developing countries, and increasing as the population becomes more urbanised (Couth and Trois, 2010). Waste from urban areas across Africa is essentially dumped on the ground with little or no control over the relative liquid and gaseous emissions. The Clean Development Mechanism (CDM), from the 1997 Kyoto Protocol, has been a vehicle to initiate projects to control GHG emissions in Africa. However, very few CDM biogas-to-energy projects have been implemented and properly registered in developing countries, and only one in Africa (Couth et al., 2010).

This chapter presents an integrated approach to quantifying greenhouse gas (GHG) emissions arising from the disposal of solid waste in Africa (and other developing countries) and reports on a large research project on Zero Waste and Waste Management Strategies towards the effective reduction of carbon emissions in the atmosphere from developing countries conducted by the University of KwaZulu-Natal since 2002. It has been estimated that over 60 million cubic metres of waste was produced in South Africa in the year 2010; 90% of these are managed by local authorities (LA) and are still disposed in landfills, at an estimated cost of over ZAR10 billion per annum (1ZAR=7US$). The focus of the study was to assist Local Authorities in the design of appropriate waste management strategies by providing a quantitative estimate of the potential for GHG reductions and landfill space savings that can be achieved through ad hoc zero waste strategies, assessing their economic feasibility and so addressing specific knowledge gaps regarding the quantity and quality of the local MSW stream.

Africa is the world's second-largest and most-populous continent after Asia. With around one billion people in 61 territories, it accounts for almost 15% of the world's population, of which 60% is rural and 40% urban or peri-urban. The rural growth rate is reported as static (0%) with an increasing urban population growth rate of 6.6% (Earthtrends, 2008). Rural waste is traditionally managed through reuse, recycling and composting. Urban waste is primarily disposed in landfills generating methane (CH_4) gas, which is 21-25 times more potent as a GHG than the natural carbon dioxide (CO_2) also produced by anaerobically degrading waste in landfills or through composting. The World Trade Organisation designated 96% of the countries in Africa with a low Human Development Index (HDI); 68% are designated by the United Nations (UN) as 'least developed countries' (LDCs), and all of the countries in sub-Saharan Africa are designated under the Kyoto Protocol as Non-Annex 1 parties (developing countries) (Couth and Trois, 2010). In most of the African countries, little of the gross domestic product (GDP) is allocated to waste management and therefore low cost, low technology solutions need to be provided.

A review of waste management practices across Africa (Couth and Trois, 2010, 2011) has concluded to date that the most practical and economic way to manage waste in the majority of urban communities is considered to be:

- Scavenging of waste at collection points to remove dry recyclables by door to door collection;
- Composting of the remaining biogenic-carbon waste in windrows, using the maturated compost as a substitute fertilizer. Non-compostable materials will need to be removed from the waste prior to composting; and
- Disposal of the remaining fossil-carbon waste to sanitary landfills.

This waste management practices will require limited capital in comparison to the complex and expensive waste treatment and landfill disposal systems in developed countries.

However, solid waste management in developing countries/emerging economies is generally characterised by highly inefficient waste collection practices, variable and inadequate levels of service due to limited resources, lack of environmental control systems and appropriate legislations, limited know-how, indiscriminate dumping, littering and scavenging and, most of all, poor environmental and waste awareness of the general public (Matete and Trois, 2008).

South Africa, as an emerging economy, is also facing the challenge of meeting high standards in service delivery with limited resources. The disparity in service coverage between different communities in the same area is a characteristic of waste management practices in South Africa.

The Polokwane Declaration in 2001 has set as very ambitious targets the reduction of waste generation and disposal by 50% and 25%, respectively, by 2012 and the development of a plan for Zero Waste by 2022, forcing South Africa to invest in the valorization of waste as a resource.

The rationale for this research stems from several factors influencing the waste management sector in South Africa, including legislative developments, national imperatives and international obligations (Kyoto Protocol, Basel Convention etc.): the growing emphasis on GHG mitigation; landfill space shortages; waste diversion and zero waste goals increased focus on waste to energy technology implemented under the CDM and similar schemes, and the requirement for waste quantification and development of a national Waste Information System as mandated by the 2008 Waste Act.

Therefore this study is intended to provide data and information to municipal waste managers with regard to potential alternatives to landfill disposal, using their carbon footprint and potential for GHG reduction as discriminants for their choice. This study focuses exclusively on commercial and residential (post-consumer) municipal solid waste (MSW) as collected and disposed in urban areas in South Africa. Two South African Municipalities (eThekwini (City of Durban) and uMgungundlovu (City of Pietermaritzburg)) were selected as representative of the waste management context in Africa, as suggested by extensive research conducted over the past 15 years (among others:

Trois et al., 2007; Matete and Trois, 2008; Trois and Simelane, 2010; Couth and Trois, 2010 & 2011;).

A zero waste model was developed to simulate various scenarios based on a dry-wet waste model that maximises diversion of recyclable fractions from disposal to landfill, as illustrated in Figure 1.

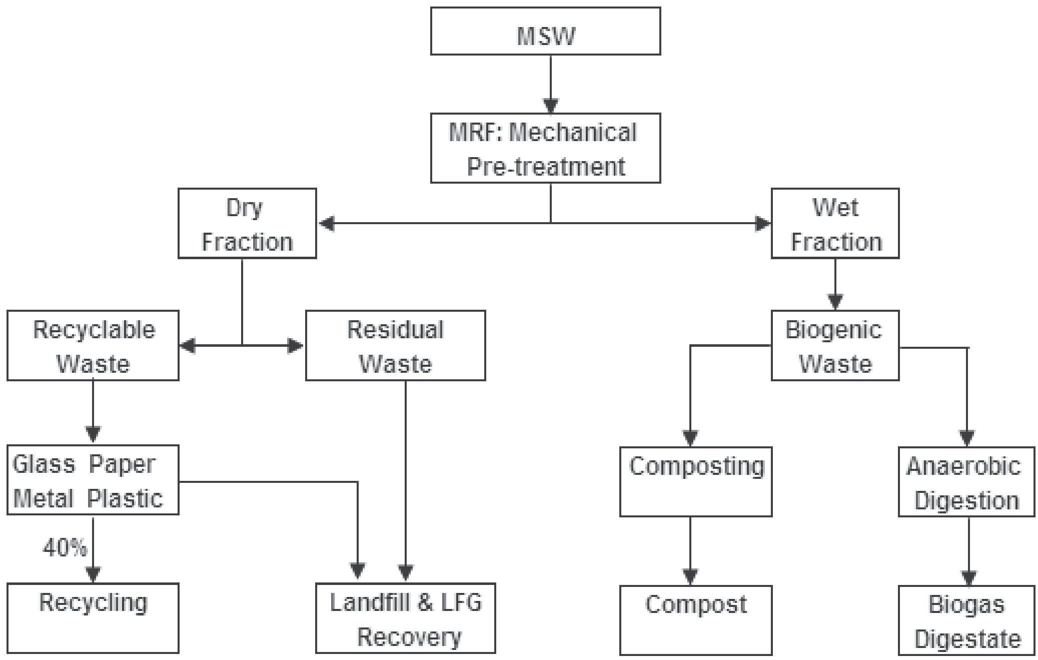

Fig. 1. A typical dry-wet waste diversion model (Matete and Trois, 2008; Trois and Simelane, 2010)

The zero waste model estimated the GHG impacts, landfill space savings and potential costs/income from the following simulated scenarios:

i. Mechanical pre-treatment, recovery of the recyclables, recycling,
ii. Biological treatment: composting or anaerobic digestion of the wet biogenic fraction, and
iii. Landfilling of all waste or residual wastes, with landfill gas recovery.

The selected strategies were considered the most appropriate for the South African context in terms of the strategy's implementation requirements, technical feasibility, and potential environmental impacts or benefits to municipal waste management systems. The selected strategies, including the status quo were then applied at municipal level for two local landfill sites in various scenarios, where the approximate waste quantities diverted, GHG reductions or emissions and resulting landfill space savings were calculated. The five scenarios evaluated were:

Scenario one: Landfill disposal of unsorted, untreated MSW.

Scenario two: Landfill disposal of unsorted, untreated MSW with landfill gas recovery.

Scenario three: Mechanical pre-treatment (MPT) of MSW, recovery of the recyclable fraction through a Material Recovery Facility (MRF) with landfill gas recovery.

Scenario four: MBT (MPT, recovery of recyclables through an MRF and anaerobic digestion of biogenic food waste with landfill gas recovery).
Scenario five: MBT (MPT, recovery of recyclables through an MRF, composting of all biogenic waste, landfill gas recovery).

2. Methodology

This study comprised of four different components in assessing potential zero waste strategies: a waste stream analysis to determine the waste stream composition and quantities of specific fractions in the waste stream; a carbon emission/reduction assessment of each strategy, a landfill airspace assessment and finally an evaluation of the costs and potential income and savings associated with each strategy.

2.1 Strategies and scenarios evaluated

Zero waste strategies that promote effective waste diversion through mechanical pre-treatment, recycling and composting or anaerobic digestion of biogenic waste were evaluated through a case study of the eThekwini (eTM) and uMgungundlovu municipalities (UMDM). These municipalities were selected as representations of a typical South African population's (in terms of social profiling and socio-economic factors) waste stream in a medium to large municipality. Scenario 1 represents the baseline scenario and the current status quo of the majority of landfill sites in South Africa. There are currently no other waste diversion/treatment methods or landfill gas recovery employed at the New England Landfill site, and thus scenario one reflects the current status quo of waste management in the UMDM. Scenario two assesses landfill disposal of all MSW, along with recovery of landfill gas produced through degradation of organic waste under anaerobic conditions within landfill cells. The third scenario evaluates the mechanical pre-treatment or separation of the recyclable dry fraction of the MSW stream, while both the biogenic wet fraction and residual waste are landfilled. In this scenario recyclables are sorted, baled and sold to local private recycling companies. Scenario three represents the status quo of the Mariannhill landfill site in Durban (eTM) that currently has both a Material Recovery Facility and a landfill gas recovery system. Currently, the Mariannhill MRF recovers between 9-13% of recyclables. Both the current rate (9.82%) and a potential recycling recovery rate (40%) were considered for the eThekwini Municipality to provide a comparison between the status quo and the potential emission reductions obtained from improving the recovery rate to 40%. A 40% recovery rate was also considered for the UMDM scenario three, as well as all further scenarios considering recycling for both municipalities. Scenarios four and five consider Mechanical Biological Treatment strategies including anaerobic digestion with energy generation and windrow composting respectively. Anaerobic digestion is currently limited to small-scale pilot projects, and industrial waste treatment in South Africa. The Dome Aeration Technology (DAT) windrow composting was chosen as the most appropriate MBT method due to its efficiency and cost implications (Trois et al., 2007; Trois and Simelane, 2010). A schematic summary of the strategies evaluated in each scenario is presented in Figure 2.

2.2 Waste stream analysis

A waste stream analysis (WSA) determines the composition of the waste stream in a particular area or region. WSA is necessary for the planning and design of waste management systems, the subsequent assessment of the efficiency of such systems and for

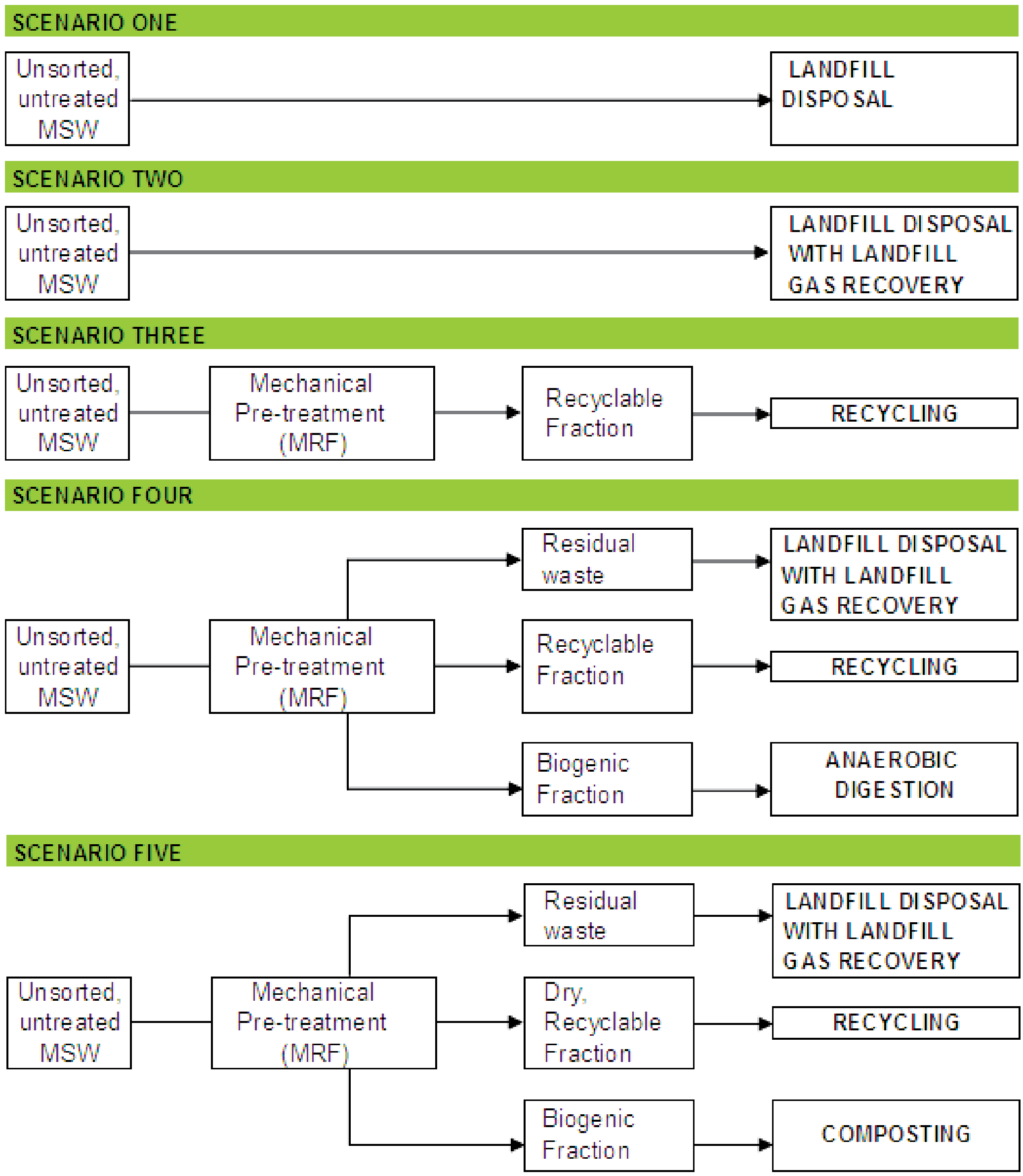

Fig. 2. Waste Management scenarios evaluated.

information and statistics reporting which is especially lacking in developing countries as far as current and up to date data is concerned. The data required was obtained through a WSA of the New England Landfill, which is the major landfill servicing the uMgungundlovu DM, and from Durban Solid Waste for the Mariannhill landfill in the eThekwini Municipality. Purnell (2009) identified two main issues concerning waste information. Firstly, the lack of reliable waste generation data which affects waste management planning; and secondly, the lack of guidelines regarding waste generation measurement, as currently only large municipal landfills operate weighbridges that determine waste quantities. The study, therefore, addresses these issues and knowledge gaps through the WSA and the subsequent data generated. A direct site specific approach to the WSA was adopted entailing sampling of waste loads, sorting and weighing of individual waste materials into over 30 waste categories (Tchobanoglous, 1993; Reinhart and McCauley-Bell, 1996). Statistical analysis of this data was employed to generate commercial and general household waste profiles for the New England Road Landfill. The results of the WSA together with data from the 1998 eThekwini WSA are presented in Table 2. The study's approach is localized to major municipal landfill sites, thus the composition results were used to simulate the average annual waste fraction quantities of the MSW stream entering the Mariannhill and New England Road Landfills, presented in Table 3.

Landfill	Mariannhill Landfill		New England Road	
Waste Stream	Residential	Commercial	Residential	Commercial
% Paper & cardboard	17.88	16.11	14.75	33.08
% Plastic	19.01	6.08	8.64	11.43
% Glass	6.82	14.86	7.00	3.87
% Metal	5.34	4.94	5.19	2.70
% Biogenic waste	45.67	35.56	34.58	29.65
% Other waste	5.27	22.45	30.04	19.47

Table 2. Composition of MSW streams to landfill in the two selected municipalities

Waste Fraction	Mariannhill (tons)	New England Road (tons)
Newspaper		5453
General mixed paper (CMW)	10627	7234
Scrap Boxes & Cardboard (K4)	9229	11402
Low density polyethylene (LDPE)	7147	2450
High density polyethylene (HDPE)	2839	1401
Polyethylene-terephthalate (PET)	1815	2037
Polypropylene (PP)	2542	1613
Polyvinyl Chloride (PVC)	1089	8
Polystyrene (PS)	2593	1101
Glass	10020	6861
Steel Cans/Tins	4817	4245
Aluminium Cans	1149	547
Biogenic Food Waste	49153	36608
Garden Refuse: Green	7887	637
Garden Refuse: Wood	807	46
Other	10800	32287
Total Waste Diverted/Disposed	122514	113930

Table 3. Average annual waste fraction quantities

2.3 Carbon emissions/reduction assessment

In terms of GHG modelling, a wide range of models using Life Cycle Analysis (LCA) have been developed. Of the models researched, few provide analysis for anaerobic digestion of biogenic wastes. The availability and applicability of the models were the limiting factors for their use and thus an ad hoc GHG quantification tool called the Waste Resource Optimisation Scenario Evaluation (WROSE) was developed as part of this study using emissions factors derived by the United States Environmental Agency (US EPA) for landfill disposal, landfill gas recovery, recycling and composting. The emissions factors used in WROSE are those derived by the United States Environmental Protection Agency using IPCC guidelines and were used as the most 'transparent' approach to modelling the GHG emissions or reductions. A streamlined LCA approach was used for the derivation of these factors – GHG impacts are considered from the point at which the waste is discarded by the waste generator, to the point at which it is disposed, treated, or recycled into new products (US EPA, 2006). The emissions factor for the anaerobic digestion of biogenic MSW was developed using the same streamlined LCA approach (on a wet weight basis) and considered the following emissions and reductions:

i. Direct emissions: Direct process emissions were determined using the IPCC greenhouse gas inventory guidelines (2006). The tier 1 approach was adopted, as this is the methodology for countries where national data and statistics are not available. The emissions factor for the biological treatment of biogenic MSW as listed by the guidelines is 1g CH_4/kg of wet waste. Nitrous oxide emissions are assumed to be negligible and an assumed 95% of methane is recovered for energy generation. Total direct emissions amounted to 0.00105 $MTCO_2eq$/ton.
ii. Transportation emissions from the collection and transportation of MSW: Transportation emissions were calculated using a similar methodology to that used in the 2009 study by Møller et al, 2009. The fuel efficiency of waste collection trucks over a 20mile distance was determined, assuming a typical value of 0.03L/ton/km. A 20mile distance to the AD facility was assumed to maintain consistency with the US EPA emissions factors. Total emissions from transportation of waste amount to approximately +0.0029794 $MTCO_2eq$/ton.
iii. Energy emissions/reductions: Energy emissions consist of emissions from the combustion of methane to produce energy; emission reductions from electricity generation and energy emissions from energy consumption. Energy reductions from substitution of fossil fuel energy due to energy recovery and electricity generation from waste. Total emissions from combustion amounted to 0.0024 $MTCO_2eq$/ton of wet waste. A typical emissions factor for combustion was chosen for the average yield of biogas from Møller et al. (2009). An average biogas yield of 110Nm^3/ton of waste digested and calorific value of 23 MJ/m^3 was used to calculate the total energy produced from combustion – with a 40% energy recovery rate (Møller et al, 2009). Approximately 18% of the total energy generated is assumed as the energy requirement for the anaerobic digestion process and operations on site. An average emissions factor of 1.015 kg CO_2/kWh was used for the electricity generated in South Africa by electricity provider ESKOM as derived by the University of Cape Town Energy Research Centre (2009). This factor is significantly higher than the average range of between 0.4 and 0.9 kg CO_2/kWh. This is likely due to the highly carbon intensive electricity grid in South Africa comprising of approximately 91.7% coal generated electricity (SA-Department of Energy, 2010). Emission reductions from the substitution of electricity amounted to -0.23397 $MTCO_2eq$/ton, thus producing an overall energy emissions factor of -0.23157 $MTCO_2eq$/ton of wet waste.

iv. Digestate Emissions: from digestate application and reductions from substitution of inorganic chemical fertiliser by compost produced from digestate. These emissions were approximated on the basis of European data (Boldrin et al, 2009; Møller et al, 2009) as no such data for the production of fertilisers is available for South Africa. A conservative value for fertiliser substitution was adopted as the nutrient composition of the digestate produced is variable and largely depends on the quality of input feedstock. The emissions from digestate amount to approximately -0.0443 $MTCO_2eq$/ton.

The resultant anaerobic digestion emission factor calculated was approximately -0.2718 $MTCO_2eq$/ton of wet waste, which is high due to the recovery of methane and production of electricity and substitution of fossil fuel energy in South Africa's carbon intensive energy supply. This factor has been calculated on a wet weight basis and therefore the WROSE model requires the amount of wet waste to be entered into the input screen under 'biogenic food waste'. For the modelling process, it was assumed 0.6 m^3 of water is added per ton of biogenic input feedstock.

2.4 Landfill space savings

The estimation of landfill space savings from waste diversion is largely an empirical calculation, as the unique conditions and operational activities on site, specifically, compaction of waste into landfill cells, influence the actual airspace saved. Actual landfill space savings (LSS) will therefore depend on the degree of compaction employed and the efficiency to which it is conducted. The calculation of LSS was based on three different methodologies to produce both a range of expected landfill space savings and an average LSS value for each scenario. The first methodology was used by Matete and Trois (2008) to calculate LSS for various zero waste scenarios. The total amount of waste in tons is divided by the average of compacted of MSW to yield the total landfill space savings. The value for the compacted density of MSW was assumed to be 1200kg/m^3 (1.2 tons/m^3) in accordance with the eThekwini Integrated Waste Management Plan (SKC Engineers, 2004). Landfill density factors of various waste fractions calculated by the United States Environmental Protection Agency (1995) and the Department of Environment and Conservation of Western Australia were used to produce further estimates, as these factors constitute a wide range of waste materials and specific fractions that can be diverted from landfill disposal.

2.5 Economic analysis

The parameters and assumptions used for estimating both capital and operational costs, and the potential income derived from the sale of recyclables, electricity, certified emissions reductions (CERs), and compost are based on research reports, journal publications, feasibility studies for local projects, and international projects where local data was unavailable. A full cost-benefit analysis should be undertaken to determine the costs and benefits over the duration of the design life for waste treatment and disposal facilities. Annual operating costs of landfill disposal amount to ZAR138 (approx. US$ 20) per ton of waste landfilled (Moodley, 2010). The capital cost of the eThekwini landfill gas to energy project for Mariannhill (0.5MW) was used as an estimate for the analysis.

A total throughput MRF capacity of 100,000 tons per year (385 tons per day) was assumed for the mechanical pre-treatment phase of the Mechanical Biological Treatment (MBT) scenarios for both landfill waste streams. The total fractions of biogenic and recyclable fractions from each waste stream amount to between 80,000-90,000 tons. It is assumed that waste loads from areas where the composition of recyclables and biogenic waste is insignificant are immediately diverted to landfill disposal. Operational and capital costs were approximated using a 2005 study by Chang et al., which approximated a linear relationship between capital and operating costs and design capacity. The total capital cost for mechanical pre-treatment and materials recovery therefore amounts to approximately US$ 33.8 million while the total annual operational cost is US$ 9.9 million/year. Recycling prices have been sourced from two local studies: The Waste Characterisation Study Report (Strachan, 2010) and the City of Cape Town IWMP (2004). It should be noted, however, that recycling prices vary in accordance with market conditions. Depending on the price of virgin materials, and other commodities such as oil, it may be cheaper to produce products from virgin materials, rather then through recycling. This reduces the demand for recyclables, and therefore directly affects prices (Stromberg, 2004; Lavee et al, 2009).

A study by Tsilemou et al. (2006) evaluated the capital and operating costs of 16 anaerobic digestion plants. A study reviewing anaerobic digestion as a treatment technology for biogenic MSW used this data to produce cost curves by Rapport et al (2008). The total biogenic fraction of the Mariannhill and New England Landfill waste streams amount to approximately 49,153 and 37,000 tons/annum respectively and therefore the chosen capacity for each anaerobic digestion plant was 50,000 and 40,000 tons/annum respectively. Using the cost curves, capital costs for anaerobic digestion plants for both the Mariannhill waste and New England waste streams amount to US$ 15.24 and US$ 13.46 million respectively, while operating costs amount to US$ 28.2 and US$ 32.4 per ton of waste respectively. The capital and operating expenses for the implementation of DAT composting plants have been determined at local level as ton and US$ 22/ton of input waste (Douglas, 2007). A degradation factor is used to estimate the yield of compost obtained from the process, and consequently the resulting income from the sale of compost. A DAT composting facility processing 180 tpd requires a capital investment of US$ 350k (Douglas, 2007). This approximation was used to estimate the capital costs for DAT composting facilities for the Mariannhill waste stream (230tpd) and New England waste stream (150tpd).

3. Results

3.1 Case study

3.1.1 eThekwini municipality – Mariannhill landfill

The eThekwini municipality is located on the eastern coastline of South Africa in the province of KwaZulu-Natal. Sub-tropical climate conditions are pre-dominant in the coastal areas of eThekwini. The municipality covers a total area of 2297 km^2 and has an approximate population of 3.16 million people. Areas of eThekwini vary in socio-economic climate from well developed urban areas of the metropolitan to newly integrated rural/peri-urban areas with little service coverage and infrastructure. Waste generation rates for the formal sector range from 0.4 - 0.8kg per capita per day, and 0.18kg per capita for the informal sector whilst the total waste landfilled per annum is approximately 1.15 million tons (SKC Engineers, 2004). There are currently three engineered landfills being operated by Durban Solid Waste in the eThekwini municipality: the Bisasar Road, Mariannhill and Buffelsdraai landfill sites. The Mariannhill landfill was selected for the study as a leachate treatment plant, landfill gas recovery and energy generation system and MRF are located on site. The landfill is therefore representative of an integrated waste management approach, which will be compared with other possible zero waste strategies. The landfill site has been operational since 1997, and has an approximate incoming waste stream of 550-700 tons per day. The landfill is expected to close in 2022 (Couth et al, 2010). The site incorporates environmentally sustainable engineering design and operational methods, and has been registered as a national conservancy site. The MRF was implemented in 2007 and recovers between 9-13% of recyclables from the waste stream (DSW, 2010). The MRF facility has since been upgraded, with the addition of mechanical sorting equipment and the extension of the pre-sorting line. The MRF has exceeded its potential in terms of initial greenhouse gas savings, has created jobs and resulted in landfill space savings, however problems have been experienced with regard to contamination of recyclable wastes by garden refuse.

3.1.2 uMgungundlovu municipality: New England road landfill

uMgungundlovu District Municipality (UMDM) is one of 11 district municipalities in KwaZulu-Natal (KZN) province and is situated within the KZN Midlands. uMgungundlovu District Municipality has a total of 234,781 households and a total population of 927,845 people (Statistics South Africa, 2005). The UMDM covers approximately 8,943 km^2 and encompasses areas of varying socio-economic conditions – from urban residential and commercial/industrial areas, to informal areas and rural, traditional areas. Waste generation rates range between 0.35-0.61 kg/capita/day for urban areas and between 0.1-0.61 kg/capita/day for rural areas (UMDM Review, 2009). An estimated 200,000 tons of waste is generated annually in the UMDM (Jogiat et al, 2010). The majority of municipal landfill sites in the UMDM does not have permits, or infrastructure such as weighbridges. This is characteristic of South African municipalities and highlights the need for improved infrastructure and waste reporting. Most of these landfill sites have been prioritised in integrated development plans. Consequently, weighbridge data is only available for the New England Road Landfill Site in uMsunduzi. The New England landfill was opened in 1950 as an open dumpsite, and was upgraded to an engineered landfill site in the 1980's, in accordance with the National Environment Act. The landfill receives an average of 183,531 tons of waste annually, which is equivalent to approximately 700 of tons of waste per day. Approximately 250,000 m^3 of compacted waste is landfilled every year (UMDM, 2009).

3.2 Carbon emissions/reductions assessment

A summary of the results obtained from the Carbon Emissions/Reductions assessment using the WROSE model is illustrated graphically in Figure 3 and 4.

The results of the carbon emissions/reduction assessment confirm that the scenario 1 (landfill disposal of all MSW) produces the greatest GHG emissions, and is therefore the least favourable waste management strategy in terms of environmental benefit. This is largely due to the degradation of biogenic wastes (food waste and garden refuse), contributing to approximately 70% and 65% of total emissions for the eThekwini Municipality and UMDM respectively as shown in Table 4. The methane produced from anaerobic conditions prevailing in landfill cells is considered in the analysis as this methane is produced through anthropogenic activity of landfilling of waste. The second greatest contributor to GHG emissions is the paper fraction of the waste stream, comprising common mixed waste and the K4 cardboard and scrap boxes (27-32% in total). This is due to the degradable carbon fraction of these materials, which ranges from 30-50% and degrades under aerobic conditions. Although the carbon in both biogenic and paper fractions degrades under aerobic conditions, some of the carbon that does not degrade is stored, causing a carbon sink. For example the degradation of lignin and cellulose varies depending on landfill conditions, and often, these compounds do not decompose to the full extent, and are stored within the landfill (landfill sequestration) (US EPA, 2006). This does not apply to other materials such as plastics, as the carbon present in plastic is obtained from fossil fuel sources and thus the carbon is considered to be transferred from one source to another (storage in the earth, to storage in a landfill). The emissions produced from landfill disposal of plastic, metal and glass fractions therefore comprise of emissions from transportation and the operation of vehicles and machinery on site.

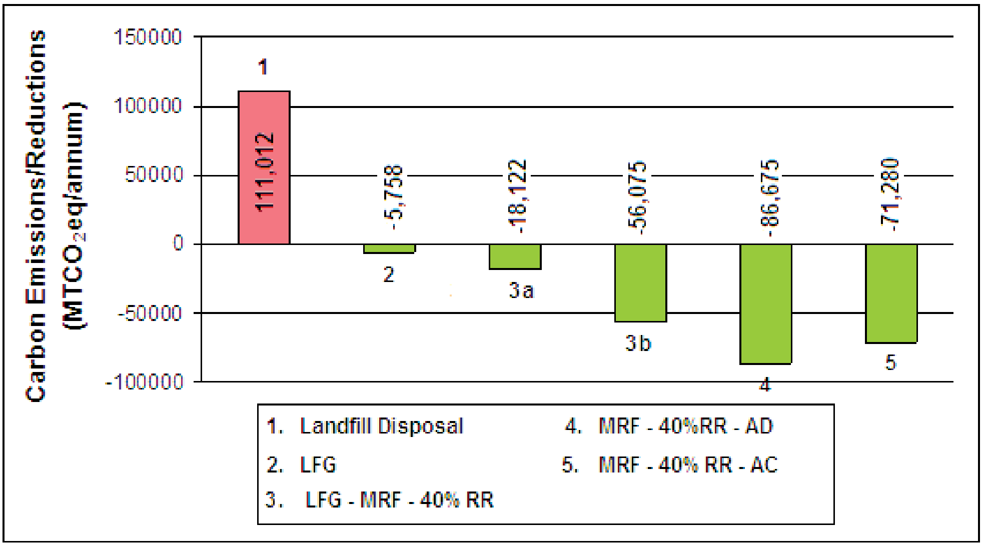

Fig. 3. CERs Assessment of the Mariannhill Landfill waste stream

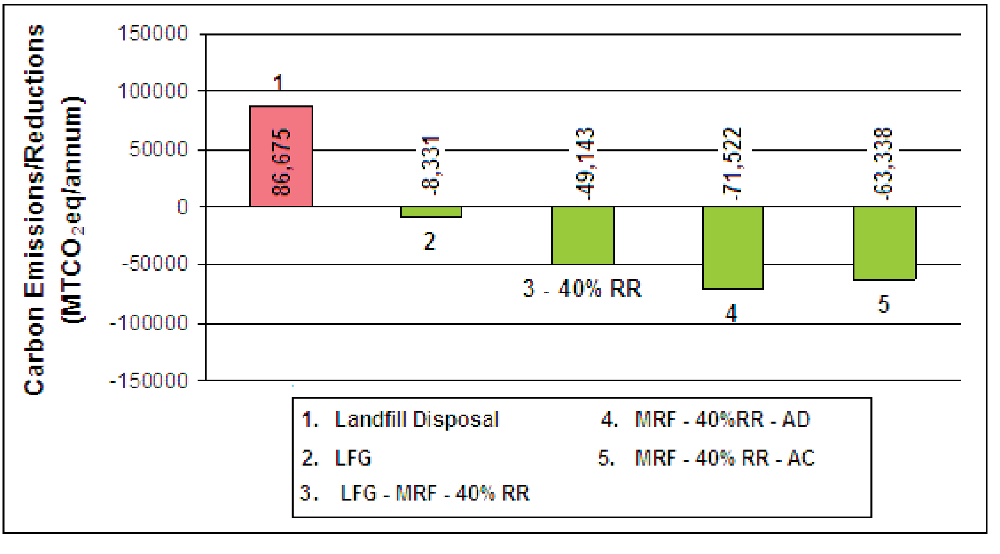

Fig. 4. CER Assessment of the New England Road Landfill waste stream

Waste Fraction	eThekwini Municipality		UMDM	
	Emissions ($MTCO_2eq$)	% Emissions Contribution	Emissions ($MTCO_2eq$)	% Emissions Contribution
General mixed paper (CMW)	15,814	14.25%	10,765	12.02%
Scrap Boxes & Cardboard (K4)	15,158	13.65%	18,727	20.91%
Low density polyethylene (LDPE)	315	0.28%	108	0.12%
High density polyethylene (HDPE)	125	0.11%	62	0.07%
Polyethylene-terephthalate (PET)	80	0.07%	90	0.10%
Polypropylene (PP)	112	0.10%	71	0.08%
Polyvinyl Chloride (PVC)	48	0.04%	0	0.00%
Polystyrene (PS)	114	0.10%	49	0.05%
Glass	442	0.40%	303	0.34%
Steel Cans/Tins	212	0.19%	187	0.21%
Aluminium Cans	51	0.05%	24	0.03%
Biogenic Food Waste	77,480	69.79%	57,705	64.43%
Garden Refuse: Green	522	0.47%	42	0.05%
Garden Refuse: Wood	62	0.06%	4	0.00%
Total Emissions from Landfilling	**110,536**	**100%**	**89,560**	**100.00%**

Table 4. Waste Fraction % contribution to GHG emissions from landfill disposal

The recovery of landfill gas at a 75% recovery rate through Scenario 2 produces a 110% and 105% decrease in emissions for the UMDM and the eThekwini Municipality respectively. These results highlight the value of landfill gas recovery for the reduction of GHG emission impacts from waste management and at the very least, landfill gas recovery systems should be employed at landfill sites. Landfill gas pumping trials would obviously be required to assess the actual yield of gas being produced as compared with the theoretical yield used in the model. The recovery of methane and generation of electricity results in GHG savings of 5,758 and 8,331 $MTCO_2eq$/annum from the eThekwini Municipality and uMgungundlovu DM respectively. Published carbon emission reductions for the Mariannhill landfill gas to energy project amounted to approximately 16,000 $MTCO_2eq$/annum (Couth et al, 2010). The difference between this data and the value calculated from the CER assessment differ by almost 10,000 $MTCO_2eq$/annum. This variation can be attributed to the nature of landfill gas production, which varies in composition and generation rate depending on the phase of degradation (Smith et al, 2001). Ritchie and Smith (2009) list factors such as waste composition, pH, moisture content, temperature and nutrient availability affect landfill gas generation. The amount of gas actually being generated and recovered could therefore differ from the calculated value depending on how these factors are taken into account. The parameters and assumptions used in the development of the US EPA emissions factors for landfill gas generation and recovery are based on experimental values; and have been identified as an area where more research is required (US EPA, 2006). The factors have also been based on the United States energy grid, which is less carbon intensive than the South African grid, and therefore a possible source of variation (underestimation of potential GHG savings) when considering the substitution of fossil fuel energy with electricity generated from landfill gas.

Recycling, which is implemented in Scenarios 3, 4 and 5, as expected produced significantly higher GHG emission reductions in comparison to all other strategies. This is largely due to substitution of recycled materials for virgin materials in production processes, and displaced energy emissions produced through the acquisition of raw materials. The status quo of waste management for the Mariannhill landfill site produces approximately 18,122 $MTCO_2eq$/annum. The current MRF recycling recovery rate produces approximately 13,000 $MTCO_2eq$/annum whilst an increase in the recovery rate to 40% produces 53,000 $MTCO_2eq$/annum. An MRF recycling facility recovering 40% of recyclables present in the New England waste stream together with landfill gas recovery would reduce emissions from the current status quo by approximately 160%. These savings (47,103 $MTCO_2eq$) could in reality be higher, as recyclables in the waste stream were found to be relatively clean and uncontaminated, as waste is not transferred, mixed and compacted at transfer stations as is the case in the eThekwini Municipality.

In terms of the treatment of the biogenic fraction of the waste, the energy generation capabilities of anaerobic digestion produce greater GHG reductions for the Mariannhill and New England waste streams: approximately 21,379 and 15,922 $MTCO_2eq$/annum respectively, and far outweigh the environmental benefits of both composting and landfill gas recovery therefore making it the most preferable strategy in terms of GHG impacts. Anaerobic digestion allows for the production of methane from the degradation of wastes to occur in a controlled environment and be captured efficiently (greater capture/collection efficiency in comparison to landfill gas recovery). The gas is produced, captured and converted into energy at a faster rate than the naturally occurring anaerobic processes in landfill cells (Ostrem, 2004). The environmental benefits of anaerobic digestion are clear; however they need to be weighed against the costs, in comparison with a less capital intensive and carbon neutral strategy such as composting. Scenarios four and five produce the greatest GHG emission reductions as they allow for integrated waste management where several strategies are implemented to target the biogenic, recyclable and residual waste fractions (Figure 5).

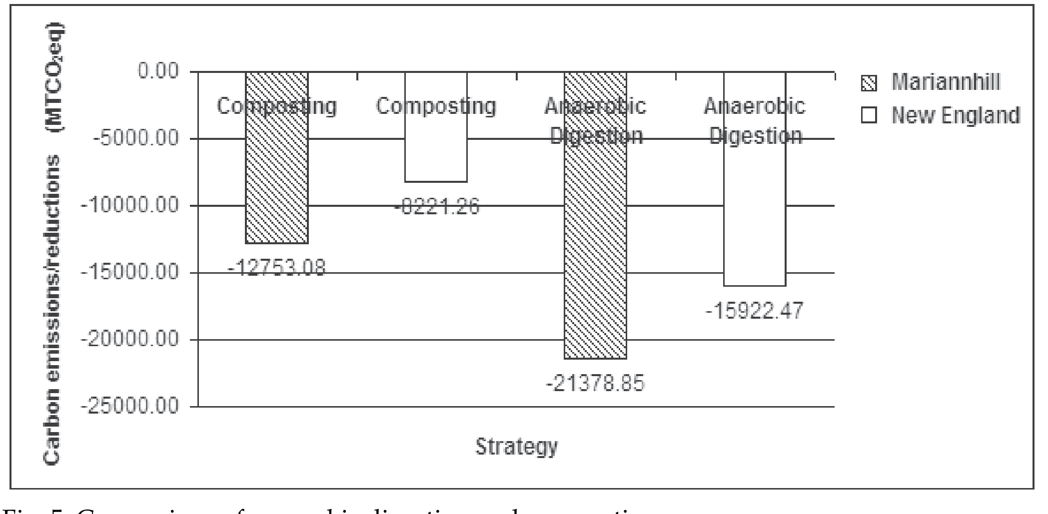

Fig. 5. Comparison of anaerobic digestion and composting

3.3 Landfill space savings analysis

The results from the landfill space savings estimate for the Mariannhill and New England Road Landfill waste streams are presented in Table 5.

Landfill Waste Stream	Landfill Space Savings (m^3)		
	Scenario 3	Scenario 4	Scenario 5
Mariannhill Waste Stream	48,503	94,375	103,302
New England Waste Stream	39,235	73,399	74,100

Table 5. Average landfill space savings

In both case studies Scenario five (MRF recycling and composting) results in the greatest average landfill space savings, with an annual saving of 103,302 m^3 for the Mariannhill landfill, and 74,100 m^3 for the New England Road landfill, as the scenario allows for the greatest amount of waste to be diverted from landfill disposal. It should be noted however that the greatest landfill space savings result from the diversion of recyclables (at a 40% recovery rate) which account for approximately 50% of the savings for both landfills if scenario five is implemented. The remaining airspace for the Mariannhill Landfill Site as at June 2002 was estimated to be 3.8 million m^3 (eThekwini Municipality, 2010). The expected date for closure of the site is in 2022 (Couth et al, 2010). Assuming 190 000 m^3 of waste is landfilled every year (3.8 million m^3 over a 20 year period), the current remaining landfill airspace amounts to 2.28 million m^3. This assumption is valid as currently 550-700 tons of waste is landfilled daily at the Mariannhill Landfill Site (Couth et al, 2010) which is equivalent to approximately 190 000 m^3 of MSW landfilled annually. The predicted landfill airspace capacity trends as illustrated by Figure 6 show that if Scenario 3 were to be achieved (40% recovery rate of recyclables) a further 4 years could be added to the landfill lifespan. The diversion of the recyclable and biogenic fraction to either composting or anaerobic digestion would extend the lifespan by 12-14 years.

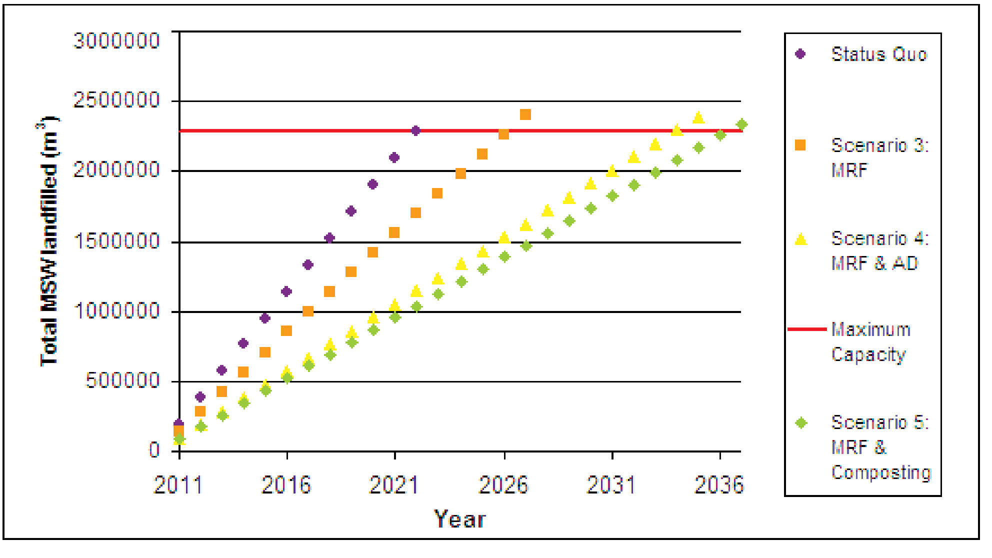

Fig. 6. Predicted airspace capacity trends: Mariannhill Landfill Site

An evaluation of landfill airspace of the New England Road Landfill estimated a remaining lifespan of six to nine years, provided that 250, 000 m³ of municipal solid waste is disposed of annually (Jogiat et al., 2010). Assuming a remaining average lifespan of eight years (expectant closure in 2016/2017 – a further six years landfill space currently remaining), the New England Road landfill currently has capacity for 1,500,000 m³ of municipal solid waste. The predicted landfill airspace trends are illustrated in Figure 7. If Scenario 3 was implemented, the landfill lifespan would be extended by a year, while if Scenario 4 or 5 were applied the lifespan would be extended by approximately two and half years.

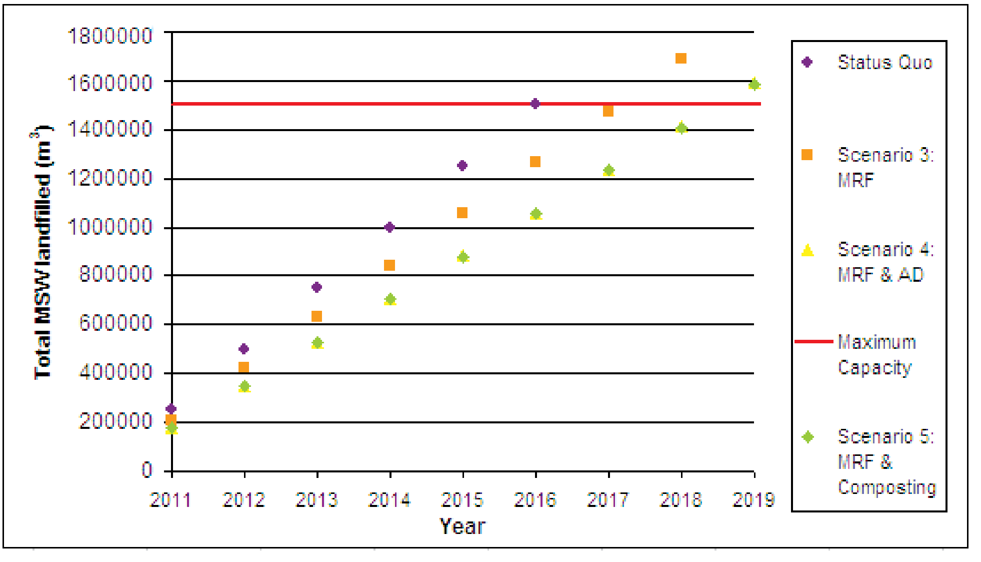

Fig. 7. Predicted airspace capacity trends: New England Road Landfill Site

3.4 Cost analysis

Table 6 presents the results of the economic analysis.

3.4.1 Landfill gas recovery

Landfill disposal with landfill gas recovery is the least capital intensive for the scale of application on both landfill sites. This highlights the previous recommendations that landfill gas recovery (at the very least) should be implemented at landfills planned in the UMDM. The actual operating costs for landfill gas recovery amount to 0.018$/kWh which equates to R 866,758/annum. The majority of operating costs stem from landfill disposal of waste (R13-14 million). Certified emissions reductions produce between R550, 458 and R796, 448 per annum. Income from the sale of electricity at the current tariff (0.047$/kWh) earns approximately R2.2 million per annum. This potential income could increase with the implementation if the Renewable Energy Feed in Tariff (REFIT), currently being developed by the government to provide incentives for investment in renewable energy sources. REFIT allows suppliers of renewable energy to sell electricity at a set price that covers generation costs and ensures a significant profit argue that both CDM and REFIT mechanisms should apply to landfill gas recovery projects, as long as it can be shown that such projects are only economically feasible with the implementation of both schemes (Couth et al, 2010).

Item	Unit	Mariannhill	New England
Landfill Gas Recovery System			
Landfill Gas Recovery System	R	1,100,000	1,100,000
Total Operating Costs	R/annum	16,906,932	15,722,340
Sale of Electricity	R/annum	2,263,201	2,263,201
Certified Emission Reductions	R/annum	550,458	796,448
Materials Recycling Facility			
MRF Capital Costs	R	33,848,875	33,848,875
MRF Operating Costs	R/annum	9,899,276	9,899,276
Sale of Recyclables	R/annum	19,598,660	15,714,260
Landfill airspace savings	R/annum	2,945,125	2,485,875
Anaerobic Digestion Plant			
AD Plant Capital Cost	R	104,066,340	90,545,910
AD Plant Operating Costs	R/annum	9,465,084	8,099,278
Sale of electricity	R/annum	5,818,124	4,333,202
Sale of Compost	R/annum	7,372,950	5,491,200
Certified Emissions Reductions	R/annum	2,043,797	1,522,172
Landfill airspace savings	R/annum	2,135,250	2,135,250
DAT Composting Plant			
Compost Plant Capital Cost	R	3,066,667	2,000,000
Compost Plant Operating Cost	R/annum	9,123,000	6,082,000
Sale of compost	R/annum	10,846,313	6,992,063
Certified Emissions Reductions	R/annum	1,219,182	785,944
Landfill airspace savings	R/annum	3,424,938	2,179,063

Table 6. Economic Analysis for all scenarios

3.4.2 Materials recycling facility

The implementation of an MRF processing 100,000 tons per annum requires significant capital investment of approximately R34 million (US$ 5m) however the greatest income and savings is achieved (approximately R19 million and R15 million (US$ 2.1m) for Mariannhill and New England Road waste streams per annum). Although price volatility in the recycling market is of concern, the MRF is still a requirement for mechanical pre-treatment phase of MBT strategies, as source separation is not implemented.

3.4.3 Anaerobic digestion and composting

A full scale anaerobic digestion plant with capacity of 40,000 and 60,000 tons for the New England Road and Mariannhill waste streams requires the greatest capital investment (R90-100 million – US$ 12.8-14.3m), with an estimated net profit of R 3 million (US$ 428k) for the NER waste stream and R 5 million (US$ 710k) for the MH waste stream. When compared to the 'carbon neutral' biological treatment of waste through composting plants, the capital expenditure required for an AD plant of this magnitude does not seem viable. A DAT composting plant produces a net profit per annum of R2 million and R3 million for a required capital expenditure of R2 million (US$ 285k) and R3 million (US$ 428k) for the NER and MH waste streams respectively, however this profit depends greatly on the establishment of a market for compost. Producers of compost often have to upgrade the nutrient content of composts, through blending with other nutrient rich organic sources, and these costs are not accounted for. In this respect anaerobic digestion plants have a definite advantage over composting, as the major potential income sourced are through the sale of electricity, and certified emission reductions, which account for approximately 50% of the total net profit for both waste streams.

4. Conclusion and recommendations

The results of the study clearly show that all waste management strategies would produce some level of environmental benefit, either in terms of greenhouse gas emission reduction and/or landfill space savings. An MBT scenario with mechanical pre-treatment and separation of the wet and dry fractions through an MRF; the consequent recycling of recyclable fractions; anaerobic digestion of biogenic waste with energy generation, and landfill disposal of all residual wastes would produce the greatest GHG reductions in both municipalities. This said there are many challenges associated with implementing new technology and waste treatment methods. The main areas to consider are costs, public perception and participation, and legislation, regulations and incentives needed to establish markets for the products yielded from landfill gas recovery, materials recovery, aerobic composting and in particular anaerobic digestion.

The capital costs for implementing waste diversion/zero waste strategies, in particular anaerobic digestion (R90-100 million) and MRF recycling (R34 million) remain the greatest challenge toward implementation on a large scale for the treatment of biogenic and recyclable fractions of MSW. The capital costs and investment required raises the issue of the relevance of these waste management strategies/technologies to a country like South Africa, where basic needs are not being met, waste management budgets are insufficient and municipalities are not able to deliver waste service coverage to all areas. Possible rationale for implementing an expensive technology such as AD is the investment in infrastructure that promotes growth and in the form of job creation and skills development. The most pressing point in evaluating the applicability of such a technology is that of environmental benefit. South Africa is the greatest producer of GHG emissions on the African continent and therefore has a responsibility to reduce carbon emissions.

Creating a market for the products of anaerobic digestion, composting and recycling – chiefly energy, compost and recycled products is vital in ensuring long-term economic viability. The UK government has created the Renewable Obligations Scotland (ROS) Policy, which requires conventional electricity suppliers to distribute a proportion of the total electricity demand from renewable energy sources, and therefore effectively guarantees a market for biogas electricity. Energy providers then purchase electricity from these renewable energy producers to satisfy these legislative requirements (Baker, 2010). Similar schemes are in the process of development in South Africa such as the Renewable Energy Feed in Tariff. Commitment from the government and initiatives such as these are required to make biogas energy an attractive and financially viable waste management option. Legislation governing the anaerobic digestion plants will also have to be developed. There are also significant challenges with regard to the implementation of recycling chiefly, improving stability within the recycling market. Subsidizing recycling initiatives would assist in keeping recycling prices constant (Nahman, 2009). The formulation of specific legislation that governs and regulates recycling, provides incentives, identifies targets for the recycling industry and provides a framework that consolidates all recycling efforts on both municipal and provincial levels into one concerted effort is necessary as currently recycling is governed by municipality specific by-laws.

This study evaluated the environmental impacts of various waste management strategies through the simulation of a zero waste management scenarios for local municipalities. The study focused on two landfill sites: the eThekwini Mariannhill landfill and UMDM New England landfill. The principal environmental impacts evaluated were GHG impacts. GHG emissions were quantified by developing the WROSE model, which primarily uses emissions factors developed by the United States Environmental Protection Agency. Herein lies the limitation of this research in that these factors are based on North American data and parameters, that may not be representative of actual emissions/reductions resulting from the implementation of these scenarios in South Africa. Despite this limitation, the research is intended to provide information and data for municipal waste managers and municipalities that will assist in assessing the alternatives to landfill disposal and derive the economic and environmental benefits of the MSW stream. The scenarios assessed are compared on the basis of theses benefits, and it is on this comparative premise that the results of the study are applicable for the purpose of assisting South African municipalities in evaluating sustainable and efficient waste management methods that promote both principles of waste diversion and GHG mitigation. The primary conclusion that can be drawn from this research is that Mechanical Biological Treatment (MBT) results in the greatest environmental benefit in terms of GHG reductions. The MBT strategy included mechanical pre-treatment of unsorted, untreated MSW which comprises sorting and separation of recyclables and biogenic wastes; recycling of the recyclable fractions and biological treatment of the biogenic fraction either through anaerobic digestion or composting. The study concluded that capital and operational costs of some technologies are the main barrier for implementation in developing countries, and the environmental and social benefits should also be evaluated further to truly gauge the costs/benefits involved.

5. Acknowledgments

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Greenhouse Gas Emission from Solid Waste Disposal Sites in Asia

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1. Introduction

1.1 Difficulties in estimating GHG emission from solid waste disposal sites (SWDSs) in Asian countries

From the viewpoint of sustainable development, appropriate waste management is crucial for conserving the local and global environments. Improvement of waste management in developing countries is directly related to preventing environmental pollution and expanding public health services. Appropriate waste management contributes to reducing not only the emission of water/atmospheric pollutants and odors, but also the emission of greenhouse gases (GHGs). Those involved in international cooperation via technology transfer should take into consideration the potential for shared benefits in terms of "co-benefit" of waste management and climate change. The recent framework of Nationally Appropriate Mitigation Actions (NAMAs) indicated in the Bali Action Plan requires measurability, reportability, and verifiability of emission reduction in mitigation action. Therefore, researchers in the waste management field have focused on finding precise and practical methods for estimating GHG emissions. Solid waste disposal sites (SWDSs) that include both managed landfills and unmanaged dump sites were recognized as major GHG emission sources in developing countries. Although the Intergovernmental Panel on Climate Change (IPCC) released guidelines for estimating GHG emissions, there is still considerable uncertainty regarding emissions from SWDSs in Asian countries, because of the lack of data about the precise emission behavior and waste degradation kinetics, especially at waste disposal sites. In this chapter, authors are going to describe the current situation of the GHG emission estimation and mitigation action in the waste management field in Asia.

1.2 Current situation of emission estimation methodology

The continuous compilation of each country's national GHG inventory is very important for understanding the status of the emissions appropriately and considering mitigation actions. However, most Non-Annex I parties cannot compile a national GHG inventory continuously. Therefore, the Greenhouse Gas Inventory Office of Japan (GIO) at the

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National Institute for Environmental Studies (NIES) has held a workshop annually since 2003 (WGIA; the participating countries are Cambodia, China, India, Indonesia, Japan, Korea, Laos, Malaysia, Mongolia, Myanmar, Philippines, Singapore, Thailand, and Vietnam), in collaboration with the activity of the workshop on improvement of solid waste management and reduction of GHG emission in Asia (SWGA), to build the capacity for the compilation of inventories in NA I countries in Asia.

At the 8th workshop of WGIA, held in July 2010, the secretariat conducted a survey by questionnaire to assess the current status of waste sector inventory in each country, and the results were shared in the waste sector working group session [Proceedings of the 8th Workshop on Greenhouse Gas Inventories in Asia (WGIA8), 2010]. Based on the survey results, we report the current status of inventory compilation for SWDS.

1.2.1 Documentation

The establishment of a common set of categories by emission source is very helpful for the comparison of countries' emissions by activity. Most of the countries estimate the GHG emissions for the categories in line with IPCC Guidelines. Most of the countries estimate the emissions using a consistent methodology and prepare the documentation describing their estimation methodology in the form of technical reports in the mother tongue and /or English, which is important to maintain its transparency.

In the estimation of GHG emissions, the Common Reporting Format (CRF) tables are a very helpful means for comparing GHG emissions and methodology by source among countries all over the world, and they are also a useful tool for verifying the completeness of emissions estimation. For that reason, several countries have generated CRF tables for their inventories, although there are no obligations to prepare these tables for the NA I countries. Instead of the CRF tables, or in parallel with them, several countries estimate emissions with the UNFCCC software for NA I countries.

1.2.2 Methodology

The level of estimation methodologies differs among the parties; for some countries, a simple method is used, and for some countries, a high-tier methodology with country-specific parameters is employed.

Cambodia, Indonesia, Malaysia, Mongolia, and Vietnam estimated potential emissions with the simple mass balance method (Tier 1) of IPCC methodology. China, Japan, Philippines, and Thailand employed the first-order decay (FOD) model to estimate emissions. Korea was attempting to employ the FOD model at the current situation.

In addition to the subcategory "Managed Disposal Site" or "Unmanaged Disposal Site" used by all of the countries, Indonesia added the country-specific subcategory "EFB solid waste – CPO mills" as another subcategory.

1.2.3 Activity data

Only a few parties completed sufficient time series analysis of the amount of final disposal to estimate emissions using the FOD method. Korea has maintained waste statistics since 1990, China has maintained statistics since 2000, and China has estimate activity data prior to 2000 using several drivers.

In many cases, there are insufficient data about the amount of final disposal to estimate emissions from SWDSs, especially from unmanaged disposal sites. Due to the lack of data for unmanaged disposal site, for some parties, emission estimates from this category are incomplete.

To resolve such problems of data collection, the all parties have been in the process of conducting a study to look for solutions. As an example of ensuring time series consistency for the amount of waste disposal, they are planning on referring to population statistics and waste generation ratio per person.

Sharing the experience, information, and knowledge regarding data collection methodology at workshops, such as those given by SWGA and WGIA, Asian countries have to make an effort to improve the inventory compilation.

1.3 Preparation of a GHG inventory and national communications

The participating countries have finished the waste sector inventory compilation included in the National Communications (NC), and most countries have completed the Second NC (SNC) to be submitted to the UNFCCC secretariat by the end of 2010. Myanmar will submit their NC for the first time. Korea already submitted the SNC in 2003.

Compilation of inventory requires the completion of many processes, such as data collection, verification of the methodology, coordination among relevant agencies, conducting surveys, and so forth. Therefore, it requires the establishment of well-resourced inventory compiling and/or a confirmation agency and the participation of specialized agencies in the inventory compilation processes by category. In each participating country, a specific agency, such as a government agency, university, research institute, and/or temporal project team took charge of inventory compilation in the waste sector (Table 1). Also, each participating country has established a compilation system to support inventory confirmation.

For the current status of national systems, Japan, Korea, Malaysia, Philippines, and Thailand expressed that they would continuously prepare their inventories. Mongolia and Vietnam reported that temporary project teams had compiled SNCs. The remaining countries responded negatively because of the following problems:

- No legal obligation to compile inventories
- Lack of human resources
- Lack of budget
- Lack of an inventory calculation system
- Lack of time

2. Specific parameters for emission estimation from SWDSs in Asia

2.1 First-order decay (FOD) model and the waste degradation rate constant (k)

The main problem of modelling landfill gas (LFG) generation is not only forecasting the amount of LFG that will be produced, but also the rate and the duration of the production [Augenstein and Pacey, 2001]. Recently, some models have been introduced to estimate the LFG generation rate of landfills. Among them, the FOD model is generally recognized as being the most widely used approach, as it was recommended by the IPCC in the 2006 IPCC Waste Model and by the U.S. Environmental Protection Agency in the LandGEM Model for calculating methane emissions from landfills [IPCC, 2006; USEPA, 1998].

The k value determines the degradation rate of refuse in the landfill. The higher the value of k, the faster the total methane generation at a landfill increases (as long as the landfill is still receiving waste) and then declines over time after the landfill closes. The value of k is a

Countries	Responsible Organization or Agency			Compilation system
	Government or relevant agency	University or research institute	Temporary project team	
Cambodia	○			○
China		○		○
India	NA	NA	NA	NA
Indonesia	○			
Japan		○		○
Korea	○			○
Laos	NA	NA	NA	NA
Malaysia	○			○
Mongolia	○		○	○
Myanmar	NA	NA	NA	NA
Philippines		○		○
Singapore	NA	NA	NA	NA
Thailand	○	○		○
Vietnam	○		○	○

Table 1. Responsible agency

function of the following factors: (1) refuse moisture content, (2) availability of nutrients for methane-generating bacteria, (3) pH, (4) temperature, (5) composition of waste, (6) climatic conditions at the site where the disposal site is located, (7) structure of the SWDS, and (8) waste disposal practices [IPCC, 2006; Pierce, 2005].

In the U.S., regulations under the Clean Air Act suggest a default k value of 0.05 yr^{-1} for conventional MSW landfills, except for landfills in dry areas where the recommended default k is 0.02 yr^{-1}. An additional set of default values is provided based on emission factors in the U.S. EPA's AP-42, which are a k value of 0.04 yr^{-1} for developing estimates for emission inventories that are considered more representative of MSW landfills where no leachate recirculation is practiced [USEPA, 1997; Thorneloe, 1999]. However, in the case of wet landfill or bioreactor landfill, where leachate recirculation is applied, Faour et al. [2007] analyzed the available recovered landfill gas from wet landfills in order to estimate the gas emission parameters for wet landfills. They found that conservative LandGEM parameters for gas collection at wet landfills suggested a k value of 0.3 yr^{-1}. In Southeast Asia, there were some studies investigating the k value by using the pumping test and the surface flux measurement. The pumping test from a landfill gas recovery project in Thailand showed that the k value was 0.32 yr^{-1}, which was close to the obtained k value from the surface flux measurement (0.33 yr^{-1}) [Wang-Yao et al, 2004; 2010]. In Vietnam, by using surface flux measurement, it was found that the k value was 0.51 yr^{-1} [Ishigaki et al., 2008]. The high content of rapidly degradable organic carbon combined with high leachate levels in the waste body might be the main reason for the specifically high degradation rate in these reports [Wangyao et al., 2008].

2.2 Gasification ratio (DOC_f)

The gasification ratio is defined as a fraction of the biodegradable carbon to be gasified. At the first stage of degradation, biodegradable carbon in waste should be converted through biological degradation, and normally it will be sequestrated or solubilized. Solubilized carbon will be converted to gas, or discharged from the landfill as leachate. The current default DOC_f was determined to be half (50%) of the biodegradable carbon that will be gasified. The remaining half of the biodegradable carbon is considered to be stored in the SWDS for long term as lignin or humus. For more accurate estimation, separate DOC_f values should be defined for specific waste types [IPCC, 2006]; for instance DOC_f of wood would be different from that of food. Since the former default DOC_f was 66%, the DOC_f value is still under scientific discussion and will likely need revision to reduce the uncertainty. In regions with higher precipitation, anaerobic sanitary landfills should discharge larger amounts of carbons. Matsufuji et al. [1996] reported that SWDSs with a high penetration rate have been found by lysimeter study to leach sometimes more than 10 percent of the carbon in the SWDS. This suggests that DOC_f in countries with higher precipitation should account for both the carbon storage in the SWDSs and the carbon discharge through leachate.

2.3 Methane oxidation (OX)

Up to 50% of emission reduction of the methane oxidation observed at a landfill surface was achieved with an engineered cover soil structure [Bogner & Matthews, 2003]. Literature survey conducted by Chanton and Powelson [2009] revealed fraction of methane oxidized ranged from 11 to 89% with a mean value of 35%. However, the IPCC guidelines recommended a 10% emission reduction of methane oxidation for managed landfills and a negligible amount for unmanaged SWDSs [IPCC, 2006]. Since most Asian countries lack sufficient scientific proofs for setting country-specific OX values, 0-10% oxidation as a default value was widely adopted.

Tropical rainfall will affect the methane oxidation by the decrease of gas permeability, and higher temperature will enhance the activity of methanotrophs. Inherently, the percentage of methane oxidation, i.e., OX, will be determined by the balance of the metabolic rate of methanotrophs, methane generation, and oxygen supply into the surface layer of SWDSs. In other words, OX might be partially related to the change of amount of methane emission. This is why it is difficult to set the appropriate OX and is one of the limitations to applying the IPCC Waste Model to Asian SWDSs.

Recent research indicated that nitrous oxide, which is a well-known GHG, must be generated by the activity of methane oxidizing bacteria [Zhang et al., 2009]. Although nitrous oxide generation should be independent from the estimation of methane emission, the total reduction capacity of GHGs should be taken into consideration when introducing methane oxidation technology.

2.4 The methane correction factor (MCF) and manner of degradation

The original concept of the MCF was the expression of inhibition of anaerobic waste degradation by the structure and management of waste landfills. Well-managed sanitary landfills were considered to exist under anaerobic conditions, and unmanaged disposal sites were assumed to be partially aerobic because of their lack of covers and/or compaction. In the IPCC guidelines, SWDSs possessing deep layers or high water table were assigned to 20% inhibition of anaerobic degradation, that is, 20% aerobic degradation. SWDSs with shallow layers were assigned to 40% inhibition of anaerobic degradation, since the ratio of surface area to total volume of waste is higher in these SWDSs than in other categories of landfill.

Under current practices, semi-aerobical management of landfills will promote aerobic degradation of waste partially thorough passive ventilation. This provided 50% of inhibition of anaerobic degradation, based on the experimental results reported by Matsufuji et al [1996]. This is an overall estimation of methane emission in semi-aerobic condition compared to that in anaerobic conditions, though the estimation methodology was developed based on anaerobic waste degradation.

Semi-aerobic landfill management was developed in Japan in the 1970s, and many Asian countries have adopted this management concept for their landfills. At unmanaged disposal sites and semi-aerobic landfills, both aerobic and anaerobic degradation will occur simultaneously in a SWDS and should exhibit a specific degradation manner different from that of anaerobic-only degradation. At this moment there is no other good model to express this complicated waste degradation manner. This is a fundamental problem in current emission estimation from the SWDSs in Asia. Further detailed information on semi-aerobic landfill management can be found in later sections.

3. Emission estimation in new waste management schemes

3.1 3R activity

Usually, the reduction, reuse, and recycling (3R) activity in the MSW management treats valuable materials, such as cans, bottles, papers, and plastic packages, in developed countries. However, recycling of these materials by private sectors has already been established in societies in most developing countries, including those in Asia [Wilson, 2009]. Therefore, the target material for 3R in such countries will be garbage or food waste. The first incentive of 3R activity is the reduction of waste disposed in landfill sites. The resource saving and the pollution reduction are the preferred results from 3R activity. The 3R activity of food waste will also result in the reduction of landfilled waste, especially in Asia, where providing enough food to guests is a polite service and/or a symbol of wealth. Since the reduction of landfilled food waste will decrease the degradable organic carbon in landfills, this activity will be a methodology for GHG reduction and also be a part of projects of Clean Development Mechanism (CDM) [Bogner, 2007].

As noted above, 3R activity consists of reduction, reuse, and recycling. Key technologies for the recycling of food waste are composting (or aerobic digestion) and biogas production (or anaerobic digestion). The latter requires a substantial investment to build up the system, including facilities for implementing biogas production. The former, composting, will be the first choice for 3R activities in most Asian countries. However, it should be noted that some GHGs (methane and nitrous oxide) will be emitted from the process of composting and from farmlands applying the compost [IPCC, 2006].

In all types of waste, recycling is tied to the demand for products. The compost made from food waste should have a quality that meets requests by farmers. A key quality factor for the waste compost will be mixed trashes, such as plastics, metals, glasses, and the like. These materials don't alter the effect of the compost when it is used as fertilizer; however, farmers dislike spreading waste onto their farm land. When the quality of compost produced by food waste does not meet the requests of farmers, it will become waste, be relegated to the landfill, and emit GHGs from the residual biodegradable carbon in the compost. Separation of trashes from the food waste is a key technology for the quality control of food waste and compost. In addition to the mechanical biological treatment (MBT) in Europe [Pan, 2007], the segregation of food waste at the source (or home) is a key part of this process. For example, Hanoi city, Vietnam, has been introducing the segregation of food waste at the home into their waste management system to reduce landfilled waste.

The reduction of food waste before generation is the most important of the 3R activities, as well as other waste. This is challenging, however, because it means asking citizens to make drastic changes in their lifestyle, including changing habits performed historically as part of their culture. In conclusion, determining ways to raise public awareness about the importance of "saving food for the environment" remains an unsolved problem and is the ultimate question that must be answered for the establishment of a sustainable society and GHG reduction.

3.2 Leachate charge to water body through landfill gas to energy (LFGTE)

Landfill gas (LFG) is formed as a natural by-product of the anaerobic decomposition of wastes in landfills. Typically, LFG is composed of about 50% methane, 45% carbon dioxide, and 5% other gases, including hydrogen sulfides and volatile organic compounds. LFG is thought to be released from six months to two years after waste is placed in the landfill [U.S. Environmental Protection Agency, 1997]. Methane is a potent GHG, with 21 times the global warming potential of carbon dioxide. LFG can contribute to malodor and present health and safety hazards if it is not well controlled. Many landfill sites have installed LFG recovery and utilization systems or landfill gas to energy (LFGTE) systems to recover the energy value of LFG and to minimize its pollutant effects.

The two common ways to recover LFG are vertical extraction wells and horizontal collectors. The standard and most commonly used is the vertical extraction well. The wells are drilled into the landfill at spacing typically ranging from 45 to 90 m. Pipes 2 to 8 inches in diameter (typically PVC or HDPE) are placed in the holes, which are backfilled with 1-inch-diameter, or larger, stones. The pipe is perforated in the lower section where the LFG is collected. Horizontal extraction collectors or trenches may be installed instead of or in combination with vertical wells to collect LFG. They consist of excavated trenches (similar to a pipeline trench) that are backfilled with permeable gravel. Perforated, slotted, or alternating diameters of pipe are installed in the trench. Horizontal extraction collectors are less expensive than vertical extraction wells and are particularly suitable for installation in active filling areas. The advantages of a horizontal extraction collector are low effects from the high leachate level problem in landfill, less obstruction for landfill operations caused by collector headers, and easy installation. The disadvantages of a horizontal extraction collector are high effects from waste settlement and a low recovery efficiency rate per well [The World Bank, 2004].

In tropical countries, the LFG collection system should be used in concert with good leachate management practices. Leachate accumulation within the refuse can dramatically impact the rate of LFG recovery, because liquid in the extraction well and collection trenches effectively restricts their ability to collect and convey LFG [The World Bank, 2004]. In Thailand, field experiences indicate that horizontal extraction collectors are more suitable compared to vertical extraction wells [Eam-O-Ppas and Panpradit, 2003]. The main purpose of using the horizontal extraction well is the very high leachate level in tropical landfills. According to the results of geophysics surveys using the electrical resistivity tomography technique in Thai landfills, the moisture content of waste inside tropical landfills was very high. The distributions of high moisture content were found in all parts of the mass of waste, even in areas where the waste had been deposited 3 years previously. The level of leachate was found in the range of 3 to 5 m beneath the final cover (5 to 7 m above ground level) [Wangyao et al., 2008].

The high level of rapidly degradable organic carbon in the waste stream combined with the high moisture content in the waste body in tropical landfills can stimulate the anaerobic degradation and produce more LFG in a shorter time after the wastes have been deposited. This means that the methane generation rate constant (k) in tropical/wet landfill must be higher than that in dry landfill, which directly affects the LFGTE. Many studies in Asian countries have shown that the k values are about 0.32 to 0.51 yr^{-1} [Wang-Yao et al., 2004; Wangyao et al., 2010; Ishigaki et al., 2008]. The high k value also means that the projected period for LFGTE will be shorter than the period for conventional landfills in Europe and the U.S. Moreover, the LFGTE projects in small and medium scale landfills in Asian countries may not be cost effective.

3.3 Semiaerobic landfill management

Semiaerobic landfill systems were developed more than 30 years ago and have since then been introduced all over Japan. Nowadays, the characteristics of waste have been changed by the economical situation in many countries and also the technical situation of pretreatment systems of municipal solid waste such as incinerators, mechanical shredders, and so on. However, semiaerobic landfill systems are still being installed in new landfill sites as fundamental technology [Tachifuji, et al., 2009], and are again attracting attention due to the reduction of GHG emissions from lanfill sites in recent years [Matsufuji, et al., 2007].

The main structure of the semiaerobic landfill system is the leachate collection pipe, which is placed on and wrapped by pebbles on the bottom layer. These pipes are linked, with a wide cross-section of pipe ends opened to the air. The most important functions of this pipe are the leachate drainage from the waste layer, and to bring air into the waste layer. The biodegradation process of organic waste can produce heat energy and increase the temperature (50 °C to 70 °C) of the waste layer. As part of this phenomenon, the air can enter the landfill body naturally by heat recirculation. Both aerobic and anaerobic conditions can be created by the leachate collection pipe in the landfill, and thus both nitrification and denitrification from the leachate can occur.

This system has many advantages, as follows:

1. Cheaper construction and maintenance fee.
2. Less influence on the surrounding environment due the leachate treatment effect.
3. Acceleration of the waste decomposing process by biodegradation due to the increased aerobic bacterial activity.
4. Reduction of water pressure on the bottom liner and prevention of seepage because of rapid draining out of the leachate.
5. Reduction of GHG emissions because of the promotion of aerobic bacterial activity by the expansion of aerobic conditions inside the landfill site.

Recently many countries have started to install this type of landfill system, especially in Asia. This system is a candidate mitigation method for the CDM project, and the new methodology for estimating the emission reduction in semiaerobic landfill projects is waiting for approval by CDM Executive Board of IPCC.

GHG emissions from semi-aerobic landfill are described by using the structure coefficient, with the MCF estimated as being half as much as that in anaerobic landfills. This effect on the reduction of GHG emissions by semiaerobic landfills is greatly influenced by the amount of passive air introduction into the waste layer. Researchers are currently investigating which parameters have a strong relation to the air inflow rates for improving the aerobic condition in landfill sites. We hope this examination will provide valuable information that will lead to wide acceptance of the CDM project for semiaerobic landfill management.

3.4 Future trends in national communication and NAMAs

On a global scale, the waste management sector makes a relatively minor contribution to GHG emissions, estimated at approximately 3-5% of total anthropogenic emissions in 2005 [Bogner et al., 2007]. The waste sector is considered to be in a unique position to move from being a minor source of global emissions to becoming a major sink of emissions [UNEP, 2010]. While the prevention and recovery of wastes is aimed at avoiding emissions in all other sectors of the economy, the GHG emissions of developing nations are anticipated to increase significantly as better waste management practices lead to more anaerobic, methane-producing conditions in landfills. Therefore, nationally appropriate mitigation actions (NAMAs) have been planned under the specific circumstances of nations. In the present framework under the Kyoto Protocol, CDM had gained initial concerns about mitigating GHG emission. CDM activity in the waste sector has been mainly concentrated on landfill gas capture (where gas is flared or used to generate energy) due to the reduction in methane emissions that can be achieved.

However, it was recognized that under the LFGTE process, fugitive methane leaks from the system also contribute to total GHG emissions from landfills. The climate benefit of this energy generation is attractive in the initial stages though the duration of electricity supply is limited. Furthermore, since most LFGTE projects cannot provide the estimated emission reduction, Asian nations realized the limited possibility of mitigation effect on GHG reduction by insufficient capacity and resources [Ministry of Natural Resources and Environment [MONRE], 2010].

Although the country-specific situation will affect the choice of mitigation option and technologies, the energy production was attracted as the most perspective options on waste-related mitigation as using rice husks to electricity and using biogas to heat and/or electricity [MONRE, 2010; Office of Natural Resources and Environmental Policy and Planning, 2011]. Substitution of raw material by the utilization of industrial or agricultural waste should also be considered, such as using molasses urea to feed dairy cattle [MONRE, 2010]. These mitigation options are focused on the main/important industries in each nation; however, the ripple effect in scale of these mitigations cannot be expected. In contrast, direct measures to improve the waste management should be the fundamental solution to achieve the co-benefit philosophy [Jochem & Madlener, 2003], such as prohibition of open dumping by 2013 in Indonesia [Hilman, 2010] and solidified fuel production from the refuse [Ministry of Nature, Environment and Tourism, 2010]. In addition, waste management provided also socioeconomic and environmental co-benefit in term of employments and imcomes as well as raising the environmental awareness and standard. In many developing countries proper waste managements the campaign to reduce GHG. In Singapore, limitation of disposal land drove to reduce the waste volume by incineration, simultaneously producing energy (Waste to Energy; WtE). Currently a total of

four WtE plants in Singapore contribute 3-4% of the country's electricity supply [National Environmental Agency, 2010].
Mitigation options in the waste sector must be determined based on each country's situation and development policies. The future planning of a nation's energy, primary industry, and manufacturing industry will be key factors when selecting the mitigation actions. The plans must be appropriate, and the technical support by developed countries must also be appropriate with regard to the nation's and world's future.

4. Conclusion – needs for specific estimation methodology for Asian nations

Disposal of organic waste is a major source of GHG emissions from the waste sector in Asia. Current estimation schemes for GHG emissions and mitigation at SWDSs were developed in and for Western countries with temperate climates and lower precipitation zones. There are several barriers to applying these to Asian countries with tropical climates and higher precipitation zones. In particular, the basic design of the IPCC Waste Model doesn't fit the unmanaged and managed SWDSs in Asia with their higher water flux, permeable cover, and semi-aerobic configuration. Available measures for the GHG mitigations at SWDSs, including LFGTE and WtE, have also emerged from Western countries, where the social and economic background is quite different from that in Asia. For example, in Asia the higher moisture content of waste, mainly caused by food waste, makes the separation and processing of food waste difficult, and the higher k value leads to failures of CDM projects of LFGTE. It is need for the Asian countries to establish appropriate estimation schemes for GHG emissions and mitigation that reflect their own situations. CDM and other mechanisms for GHG reduction actively promote several researches, development and projects for GHG mitigation in the waste sector of Asia. These projects, if successful, will release Asia from situations of being "unable to comply because of insufficient information" and reveal measures that are specific and appropriate in Asia. Naturally, appropriate mitigation of GHG emission from organic waste will achieve local environmental protection and 3R, that is expressing as the "co-benefit".

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