WASTE MANAGEMENT AND ECONOMY

ENVIRONMENTAL AND SOCIO-ECONOMIC IMPACTS OF LANDFILLS

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ABSTRACT

A modern landfill is an engineered method for depositing waste in specially constructed and protected cells on the land surface or in excavations into the land surface. Despite the fact that an increasing amount of waste is reused, recycled or energetically valorized, landfills still play an important role in waste management strategies. The degradation of wastes in the landfill results in the production of leachate and gases. These emissions are potential threats to human health and to the quality of the environment. Landfill gas consists mainly of methane and carbon dioxide, both important greenhouse gases. Landfill sites contribute 20% of the global anthropogenic methane emissions. Furthermore, it usually contains a large number of other gases at low concentrations, some of which are toxic. Leachate can migrate to groundwater or even to surface water through the flaws in the liners and this poses a serious problem as aquifers require extensive time for rehabilitation. Construction and management of landfills have ecological effects that may lead to landscape changes, loss of habitats and displacement of fauna. Socio-economic impacts of landfills include risks for public health derived from surface or groundwater contamination by leachate, the diffusion of litter into the wider environment and inadequate on-site recycling activities. Nuisances such as flies, odors, smoke and noise are frequently cited among the reasons why people do not want to reside close to landfills. Various researches conclude that landfills likely have an adverse negative impact upon housing values depending upon the actual distance from the landfill. The present paper reviews the environmental and socio-economic impacts related to landfills and presents existing modeling approaches to assess these impacts. Furthermore, this review is complemented with suggestions to minimize the environmental burden of landfills and to re-introduce the buried resources to the material cycle.

KEYWORDS

Landfill, Landfill gas, Leachate, Environmental impacts, Socio-economic impacts, Bio reactor, ELFM.

1 INTRODUCTION

Despite the fact that the EU waste hierarchy, as set by the Waste Framework Directive (2008/98/EC) establishes the preference of reuse, recycling and recovery of waste above landfilling, a significant amount of waste is still landfilled. It is a well-known fact that landfilling has environmental effects, mainly due to the long term methane emission and leachate production. Landfill gas consists mainly of methane and carbon dioxide and it can also contain a large number of other gases at low concentrations some of which are toxic[1]. The substances that are present in landfill gas are known to contribute to several environmental problems such as global warming, acidification, depletion of the quality of ecosystem as well as social issues like human health [2-7]. Leachate production is also a major concern as leachate can migrate to surface and groundwater. This is more serious than river pollution because aquifers require extensive time for rehabilitation [1]. Landfill leachate may present significant concentrations of trace metals, nutrients such as nitrate and phosphate, ammonia and chlorides. Apart from the environmental burdens, occupation and requirement of the enormous space for landfills generates the issue of land scarcity for the development of human society and eco systems. Moreover, landfills decrease the market value of the surrounding area [4, 5]. Different modeling approaches to quantify landfill emissions have been developed. Most of the models concentrate on landfill gas and leachate and a few of them address nuisances like odor, dust, noise and etc. In addition to the generation models, a few studies have been performed to model the impacts of landfills. Landfill modeling in life cycle analysis (LCA) is the most common approach.
The purpose of our research is to review the existing literature on environmental and socio-economic impacts of landfills. An attention has been given to the available modeling approaches to assess the landfill emissions and their impacts. Furthermore, this paper highlights evolving landfill concepts such as landfill bio reactors and enhanced landfill mining as to minimize the risk and environmental burdens of landfills and to re-introduce the disposed resources to the material cycle.

2 LANDFILLS AND LANDFILL EMISSIONS

A modern landfill is an engineered method for depositing waste in specially constructed and protected cells on the land surface or in excavations into the land surface. Within the landfill, biological, chemical and physical processes occur and they promote the degradation of wastes and result in the production of leachate and gases. The landfill ecosystem is quite diverse due to the heterogeneous nature of waste and the variety of landfill operating characteristics. The diversity of the ecosystem promotes stability; however the system is strongly influenced by environmental conditions such as temperature, pH, the presence of toxins, moisture content and the oxidation reduction potential. The stabilization of wastes proceeds in five sequential and distinct phases [8]:

1) Initial adjustment phase: This phase is associated with initial deposition of solid waste and accumulation of moisture within landfills. An acclimatization period is observed until sufficient moisture develops to support an active microbial community.
2) Transition phase: In the transition phase transformation from aerobic to anaerobic environment occurs.
3) Acid formation phase: The continuous hydrolysis of solid waste followed by the microbial conversion of biodegradable organic content results in the production of intermediate volatile organic acids at high concentrations throughout this phase.
4) Methane fermentation phase: Intermediate acids are consumed by methanogenic bacteria and converted into methane and carbon dioxide.

5) Maturation phase: During the final state of landfill stabilization, nutrients and available substrate become limiting, gas production dramatically drops and leachate strength stays steady at much lower concentrations.

Apart from landfill gas and leachate emission, wind-blown litter, vermin and insects are also identified as the minor emissions of the landfills. But the following discussion is limited to the landfill gas and leachate as they are the most important causes for number of environmental and socio economic impacts.

2.1 Landfill Gas

In theory the biological decomposition of one ton of municipal solid waste produces 442 m^3 of landfill gas containing 55% methane and a calorific value of 15 - 21 MJ/m^3 [9], which is approximately half that of natural gas. The major components of landfill gas are methane (CH_4) and carbon dioxide (CO_2), with a large number of other constituents at low concentrations such as ammonia, sulfide and non-methane volatile organic compounds (VOCs) [1]. Chemical and biochemical transformations within the landfill create new organic or inorganic substances; e.g. tri- and per-chlorethylene to vinylchloride; amino acids to methyl- and ethyl-mercaptans; or sulphur compounds to hydrogen sulphide (H_2S). For these reasons, inclusion of large amounts of particular types of industrial waste in a landfill can generate high quantities of other gaseous compounds. For example, a very large proportion of plasterboard (i.e. gypsum, $CaSO_4$) may cause the emission of H_2S [10]. The US EPA [11] listed 94 non-methane organic compounds found in air emissions from municipal solid waste landfills, which included benzene, toluene, chloroform, vinyl chloride, carbon tetrachloride, and 1,1,1trichloroethane. Forty-one are halogenated compounds. Toluene, xylenes, propylbenzenes, vinyl chloride, tetrachloroethylene, methanethiol and methanol have been reported from landfills that received both municipal and industrial wastes [12]. CH_4 and CO_2 are greenhouse gases which were the main focus of the 1997 Kyoto Agreement and of subsequent efforts at world-wide emission reduction. Landfill sites contribute 20% of the total global anthropogenic methane emission [13].

Landfill gas is generally controlled by installing vertical or horizontal wells within the landfill. These wells are either vented to the atmosphere or connected to a central blower system that pulls gas to a flare or treatment process. Intergovernmental Panel on Climate Change (IPCC) report that the landfill gas collection efficiencies ranging from 9-90% and estimates an average of 20% [14]. The uncaptured gas can pose an environmental threat because methane is a greenhouse gas and many of the VOCs are odorous and toxic. This issue is discussed in the other sections of this paper.

2.2 Leachate

Leachate is defined as any liquid percolating through the deposited waste and emitted from or contained within a landfill. As it percolates through the waste it picks up suspended and soluble materials that originate from, or are products of the degradation of the waste. The principal organic contents of leachate are formed during the breakdown process described above and its organic strength is normally measured in terms of biochemical oxygen demand (BOD), chemical oxygen demand (COD), or total organic carbon (TOC) [1]. The municipal solid waste leachate contains a wide variety of hazardous, toxic or carcinogenic chemical contaminants [6]. Moreover, mining wastes, sewage sludge and residual solids from air pollution control equipment contain high concentrations of trace metals, a range of acids and even radioactive material. Under the acidic conditions hazardous trace metals such as copper, cadmium, zinc and lead dissolve and travel with leachate [1].The characteristics of leachate produced are highly variable depending on the composition of the waste, precipitation rates,

site hydrology, compaction, cover design, waste age, sampling procedures and interaction of leachate with the environment and landfill design and operation.
It is important to control and manage the leachate production and discharge due to the potential threat of it to both the environment, particularly groundwater, and human health. An effective leachate collection and removal system is a prerequisite for all non-hazardous and hazardous landfill sites and it must function over the landfill's design lifetime.

3 MODELING APPROACHES TO ASSESS THE LANDFILL EMISSION AND THEIR IMPACTS

Modeling landfill emissions and their impacts already exists for several decades. Many researchers have conducted studies to evaluate the landfill emission management. Most of the studies are mainly about landfill gas and leachate and a few of them address nuisances like odor, dust and noise. This section summarizes the different modeling approaches available to evaluate and quantify the landfill emission and their environmental and socio-economic impacts.
Attempts to model landfill gas formation stem from the early '80's. The first landfill gas formation models were made to help determine the size of landfill gas recovery projects. They estimate the amount of formation and including future expectation and gas recovery. More recent models quantify methane emission. As described in the review of Oonk H. [15], modeling of methane emission generally requires modeling of methane generation, measuring landfill gas recovery and assuming some methane oxidation. The emission equals the gas generation minus the gas recovery minus the gas oxidation.
According to Oonk, the major issue when modeling methane emissions is the modeling of the methane or landfill gas formation. Most of the models are based on a first order decay model (a first order decay models have one half-time of biodegradation) or a multi-phase model (multi-phase models consider 3 fractions: fast, moderate and slow degradation of waste, each with their own half-time of biodegradation). Modeling oxidation has received less attention: in most cases 10% of the methane flux through the top layer simply is assumed to be oxidized. Nevertheless, more recent models are being developed for the evaluation of methane oxidation as well. The most widely applied generation models are the IPCC model, the TNO model, GasSim Lite, Landgem, the Afvalzorg-model, the French E-PRTR-model and the Finnish E-PRTR-model [15]. The IPCC model is intended to give guidance to national authorities in the quantification of methane emissions from all landfills in a country. But the model itself can also be used for individual landfills. The choices exist between a first order decay model and a multi-phase model. The IPCC model accommodates for 4 different climate regions [16]. TNO is the first model in which model parameters were based on real data of landfill gas generation in a larger group of landfills. Both a first order and a multi-phase model were made, that describe landfill gas generation as a function of amount of waste deposited from different origin ([17, 18]. GasSim Lite quantifies all landfill gas problems of a landfill, ranging from methane emissions, effects of utilization of landfill gas on local air quality to landfill gas migration via the subsoil to adjacent buildings [15]. Landgem is a first order decay model, with separate default values for the rate constant of biodegradation for conventional and arid regions [19]. The Afvalzorg model itself is a multi-phase model and is intended to give a more realistic prognosis of methane generation at landfills with little or no household waste deposited. The French E-PRTR-model is a simplified first order decay model and the Finnish E-PRTR-model is a multi-phase model with model parameters for different climatic regions [15]. In addition to these models, three dimensional models have been developed for transport and reaction of gaseous mixtures in a landfill [20-24].

Successful prediction of the amount of landfill leachate generated and its composition is a highly complex and difficult task. As discussed in previous sections, the amount of leachate generated is primarily a function of water availability, waste characteristics and landfill surface conditions. Similar to landfill gas, numerous leachate generation and transport models have been developed. These models can be classified into two types: (1) models that emphasized only the quantity of leachate generated; and (2) models that combined both quantity and composition [25]. Among these models that can estimate the volume of leachate generated from a landfill, the Water Balance Method (WBM) is the most commonly used [25-27]. The WBM simply states that water infiltrating through the landfill cover and past the depth influenced by evapotranspiration will eventually emanate from the landfill as leachate. This is valid after the solid waste reaches absorptive capacity for holding water, which may take several years. Although this method is theoretically correct and simple, a great degree of uncertainty is associated with estimating its variables [28]. Demetracopoulos, Sehayek et al. [29] built up a mathematical model for the generation and transport of solute contaminants through a solid waste landfill. A three dimensional mathematical model has been developed by Demirekler, Rowe et al. [30] to estimate the quality and quantity of the leachate produced. The model takes the effects of changing hydraulic conductivity with overburden pressure and time dependent landfill development into consideration. Laner, Fellner et al.[31] suggested a methodology to estimate future emission levels, mainly leachate, for a closed municipal solid waste landfill. The approach is based on an assessment of the state of the landfill including detailed analysis of landfill monitoring data, investigations of the landfill waste and an evaluation of engineered landfill facilities.
Apart from these gas and leachate generation models, many modeling approaches have been developed for assessing the environmental and socio economic impact of the landfills. Landfill modeling in life cycle analysis (LCA) is the most common approach. Obersteiner, Binner et al. [32] introduce and discuss the different approaches concerning time horizon and life cycle inventory data for landfills in Central Europe. Damgaard, Manfredi et al. [3] performed an economic and environmental evaluation of landfill leachate and gas technologies by using waste LCA model EASEWASTE. A methodology to estimate future emission rates and evaluate the response of the affected environment based on the current state of the landfill and its surroundings has been introduced by Laner, Fellner et al. [31]. They present a modeling approach to evaluate residual environmental impacts in view of different post closure management strategies. In addition to that numerous LCA studies have been conducted to compare the environmental impact of landfills with that of other waste treatment technologies [33-35]. Furthermore, Úbeda, Ferrer et al.[36] developed a Gaussian dispersion model to evaluate the odor impact from a landfill area. Apart from environmental modeling a few studies report for economic models of landfills. Similar to the environmental modeling, landfilling has been compared with the other waste management systems from an economic point of view [7, 37]. Some studies have been performed to assess the social impacts of landfills. Assessing the impact of landfills on residential property values is an example [4, 5, 38].

4 ENVIRONMENTAL IMPACT OF LANDFILLS

As with any waste management activity, landfilling is also a potential threat to the quality of the environment due to its gaseous and leachate emissions as well as wind-blown litter and dust. There are also substantial environmental effects associated with waste transport and collection. In this section the environmental effects of landfilling are discussed, making use of the results of above mentioned modeling approaches towards landfill emission and their impacts. Three major categories of environmental impacts are considered: (1) Landfill construction (2) Landfill gas (3) Leachate.

4.1 Impact of landfill construction

Site selection of waste management facilities can be a major issue as all infrastructural projects have the capacity to damage the ecology of the site on which they are developed, causing landscape changes, loss of habitats and displacement of fauna. Such impacts are generally site specific and need to be assessed on a case by case basis [1, 39-41].
The soils on selected sites tend to suffer from high levels of disturbances and their chemical and physical properties differ from those of the surrounding areas due to the general removal of topsoil as well as specific process related changes. Soil is an important resource which supports a variety of ecological, economic and cultural functions. The factors like porosity, density, water holding capacity and aggregate strength that operates the soil quality are best developed in the top soil fraction, subsoil being more poorly developed and having a lower ability to support plant growth. This quality can be disturbed during the construction activities. The movements of heavy machinery can lead to excessive compaction of topsoil and subsoil, and in deeper soil this may only be reversible over relatively longer time periods.
There is a considerable impact on flora and fauna during the construction phase of landfills due to the removal of existing vegetation. But this damage could be recovered after the closing phase of the landfills. The studies have shown that landfills are capable of supporting a rich and varied fauna including exotic species during the operational and closing phase of landfills [42].

4.2 Impact of landfill gas

The environmental impact of gaseous emission from landfills, which are of global or regional significance, can be mainly grouped as contribution to the greenhouse effect and damage to the eco system. Apart from that, risk of explosion and odor problem due to some trace gases can also be identified as significant impacts.
As described in earlier sections of this paper, CO_2 and CH_4 are the primary constituents of environmental importance in landfill gas. They act as greenhouse gases of global significance, with CH_4 being the most active but CO_2 being produced in the greatest quantities [2]. The LCA modeling performed by Damgaard, Manfredi et al. [3] shows that landfills are main contributors for global warming and photochemical and stratospheric ozone formation. According to Clarke [43], O'Neill [44] and Wellburn [45], CH_4 reacts with hydroxyl radicals and oxygen in the atmosphere to generate CO_2 within a period of days to a few years, thereby losing some of their greenhouse gas potential. Small amounts of methane are also consumed after absorption by soil [46]. Nevertheless, control of these emissions at the source is necessary from an environmental protection viewpoint and to address the obligations under the Kyoto protocol.
Gaseous pollutants have significant effects on plants, animals and entire eco systems. The lateral migration of gas through soil beyond landfill boundaries causes the displacement of oxygen from soil. This results in a decline in soil faunal populations and burrowing animals and causes vegetation dieback. Mainly the vegetation around the landfill and the newly planted vegetation on a closed landfill can be damaged due to the suppression of air around the roots by migrated landfill gas [1]. The acidic gaseous constituents contribute to the phenomenon of acid rains and its secondary effects on the acidification of soils and ecosystems. Ammonia is a major acidic constituent which can be found in the landfill gas. It is a secondary acidifying agent following its atmospheric oxidation to nitric acid. It has effects on plants, causing a loss of stomatal control, a reduction in photosynthesis, enzyme

inhibition, changes in synthetic pathways and depressed growth and yield. Hydrogen sulfide is also having a considerable impact on ecosystem. It is an extremely biotoxic gas, effective at a few parts per billion in mammals. Plants are far less sensitive to direct toxicity effects but have a threshold of 1µg/g [45, 47]. The most severe impact on plants is inhibition and destruction of root growth and vegetation cover due to the anaerobic soil conditions created by high concentration of sulfides which laterally seepage from landfill sites. VOCs play a significant role in formation of ground level ozone. High concentrations of ground level ozone tend to inhibit the photosynthesis, reduce growth and depress the agricultural yields [48, 49].

Gendebien, Pauwels et al.[50] say that the lateral migration of gas through soil has been the cause of a number of hazardous explosions as methane is inflammable and explosive when it mix with sufficient amount of air. Moreover, an unpleasant odor can be caused by the series of trace elements present in the landfill gas especially organic fatty acids from the acid phase and H_2S and other sulfur containing compounds. These impacts are discussed further in this paper under the section of socio- economic impacts of landfills.

4.3 Impact of leachate

The leachate production decreases very slowly and some parameters might be of environmental relevance for many decades to centuries. The main constituents of landfill leachate are dissolved methane, fatty acids, sulfate, nitrate, nitrite, phosphates, calcium, sodium, chloride, magnesium, potassium and trace metals like chromium, manganese, iron, nickel, copper, zinc, cadmium, mercury and lead. Leachate can migrate through the soil to groundwater or even to surface water due to the absence of proper liner system or damages of the liners and this results a serious problem as aquifers require extensive time periods for rehabilitation. Moreover, soil can retain the constituents of the leachate like metals and nutrients and can cause adverse impacts on the eco system.

The metals retained by the soil uptake by plants and thereby provide a key route for entry of metals into the food chain. Deposition of trace metals in the plants can affect crop growth and productivity and also pose a greater threat to animal health. Those metals such as lead, zinc and cadmium show differential mobility through the vegetation and invertebrate trophic levels and must be assessed by case by case basis [1]. Uptake by plants is affected by soil pH and salinity and also cadmium and lead uptake is enhanced by the chloride complexation of the metals present in the leachate [51]. Eutrophication is the most extensive threat when the leachate is mixed with the surface water with higher concentrations of nitrate and phosphates [52]. Eutrophic conditions invariably cause excessive production of planktonic algae and cyanobacteria in the open sectors of the lakes. This excessive production of algae results adverse impacts on fish species in the lake by limiting the light penetration into the lake. Ammonia generated from leachate within landfills will migrate through the soil horizons where it is progressively nitrified to nitrite and nitrate and cause eutrophication problem. A number of chemicals can disrupt the reproductive behavior in a range of species by acting as oestrogen mimics. Dempsey and Costello [53] found the landfill leachate as a potential source for these substances.

Above mentioned metals can be present in the leachate either in large or small concentrations depending on the waste categories deposed in the landfills. Mercury is one of the best studied contaminant. It is one of the most toxic metals within the food chain, being readily absorbed by animals, fish and shellfish. Landfills are potential mercury emitters to the eco system due to the disposal of batteries and paint residues in the landfills. Alloway [51] revealed that the chromide to chromate conversion in the landfills is environmentally significant as chromate is more toxic to plants than chromide.

5 THE HEALTH AND SOCIAL IMPACTS OF LANDFILLS

Apart from the environmental impacts, landfills are sources for several socio-economic impacts like public health issues due to the exposure to landfill gas and to the ground and surface water contaminated by landfill leachate. Although modern landfill sites are well designed to reduce emissions, the emissions from landfills continue to give rise to concerns about the health effects of living and working near these sites, both new and old. The exposure to contaminants and emissions can be via direct contact, inhalation or ingestion of contaminated food and water. Drinking water contamination has been identified as the source of exposure to harmful substances in many studies [54-56]. Those studies revealed that congenital malformations, birth weight, prematurity and child growth and cancers have a significant impact on landfill emissions. In a multi- site study of residents of New York State, a 12% increased risk of congenital malformations in children born to families within one mile of hazardous waste sites were reported [57]. Fielder, Poon-King et al.[58] and Vrijheid, Dolk et al. [59] also found an increased risk of congenital malformations in populations live near landfill sites. A multi-site European study called EUROHAZCON discovered a 33% increase in non- chromosomal birth defects among the residents living within 3 km of the 21 hazardous waste landfill sites studied [60]. This conclusion was confirmed by the study conducted by Elliott, Briggs et al.[61]. A number of studies revealed that there is a higher risk of developing cancer among the people near landfill sites and the elevated risks were observed for cancers of the stomach, liver and intrahepatic bile ducts and trachea, bronchus, lung, cervix and prostate [62, 63]

In addition to the health issues, landfills create considerable impacts on land value, land degradation and land availability. Various researches conclude that landfills likely have an adverse negative impact upon housing values depending upon the actual distance from the landfill [4, 5, 38]. Potential hazards such as flies, odor, smoke, noise and threat to water supplies are cited as reasons why the public do not want to reside close to the landfills. Reichert, Small et al. [38] revealed that 40% of participants to their survey reported odor and unattractiveness as the most severe nuisance while 35 % reported about the toxic water runoff and methane gas emission. Their study concluded that landfills have a negative impact of 5.5-7.3% of market value depending on the distance to landfills. Akinjare, Ayedun et al.[5] found that all residential property values increased with the distance away from landfill sites at an average of 6%. Ready [4] performed a meta-analysis that included all available hedonic price studies of the impact of landfills on nearby property values. It showed that landfills that accept high volumes of waste (500 tons per day or more) depresses the value of an adjacent property by 12.9% while a low volume landfill depresses this value only by 2.5%. Furthermore, occupation and requirement of the enormous space for landfills contribute to land scarcity for the development of human society and eco systems.

6 EVOLVING LANDFILL CONCEPTS

Despite the landfilling has become the final option of the waste hierarchy defined by the EU waste directive (2008/98/EC), it is still expected to be applied in several cases because of the growing amount of solid wastes and a lack of suitable techniques to treat all kinds of wastes. But it is very clear that the landfill concept should evolve to minimize the potential risks and environmental burden of landfills and on the other hand to re-introduce the buried resources to the material cycle. One approach is engineered bioreactor landfills in which a controlled degradation is allowed in order to guarantee the long term stability of the landfill [64]. Another approach is the concept of enhanced landfill mining (ELFM) that reduces the emission and potential hazard of landfills and valorize the resources contained in it. Several studies have been conducted on ELFM both in environmental and economic point of view [65-68].

6.1 Landfill as a reactor

Waste decomposing period of a MSW landfill is estimated as over fifty years. There is considerable interest in techniques for shortening this time because it has the potential of reducing overall costs and risks. One method is considering a landfill as a bio-reactor in which the degradation processes is provocatively accelerated [1]. A bioreactor landfill is a sanitary landfill site that uses enhanced microbiological processes to transform and stabilize the readily and moderately decomposable organic waste constituents within 5 to 8 years of bioreactor process implementation [64] . According to Warith's study, the bioreactor landfill significantly increases the extent of organic waste decomposition, conversion rates and process effectiveness over those occur within the traditional landfill sites. The environmental performance measurement parameters (landfill gas composition and generation rate, and leachate constituent concentrations) remain at steady levels. A bioreactor landfill site requires effective operation of liquid addition and management. Other than that waste shredding, pH adjustment, nutrient addition and balance, waste pre-disposal and post-disposal conditioning, and temperature management may also serve to optimize the bioreactor process. The advantages of bio reactor landfills are: enhancement the landfill gas generation rates, reduction of environmental impact, production of end product that does not need land filling, overall reduction of land filling cost, reduction of leachate treatment operational cost, reduction in post-closure care, maintenance and overall reduction of contaminating life span of the landfill due to a decrease in contaminant concentrations during the operating period of the bioreactor landfills.

6.2 Enhanced landfill mining (ELFM)

The previous sections of this paper highlighted that landfills have related implications such as long term methane emissions, local pollution concerns, settling issues and limitation on urban development. Landfill mining consisting of excavation, processing, treatment and/or recycling of deposited materials has been suggested as a strategy to address such problems [67]. ELFM includes the combined valorization of the historic waste streams as both materials and energy. As mentioned in the review of Krook, Svensson et al. [67] massive amounts of important materials such as metals have accumulated in landfills. On a global level, the amount of copper situated in such deposits (393 million metric tons) has been estimated as comparable in size to the present stock in use within the technosphere (330 milion metric tons). The same study revealed that apart from metals, the amount of potential waste fuel situated in municipal waste landfills is enough to cover the district heating demand in the country for 10 years. Apart from old landfills, ELFM is also applicable to new landfills by considering them as temporary storages. In that approach landfills become future mines for materials which could not be recycled with existing technologies or show a clear potential to be recycled in a more effective way in near future [69, 70]. Recently, Van Passel, Dubois et al.[66] address the economics of ELFM both from private point of view as well as from a societal perspective. Their analysis shows that there is a substantial economic potential for ELFM projects on the wider regional level. Furthermore, the feasibility of ELFM is studied by synthesizing the research on the Closing the Circle project, the first ELFM project targeting the 18 million metric ton landfill in Houthalen-Helchteren in the East of Belgium [71]. They highlighted the worldwide potential of ELFM in terms of climate gains, materials and energy utilization, job creation and land reclamation. Nevertheless, for ELFM to reach its full potential, developing standardized frameworks for evaluating critical factors for environmental and economic performance is necessary. Moreover, strategic policy decisions and tailored support systems, including combined incentives for material recycling, energy utilization and nature restoration, are also required [67, 71].

7 CONCLUSIONS

Landfills mainly emit gas and contaminated water as well as wind-blown litter and dust. Landfills are potential threat to the quality of the environment, although the full extent of this threat has not always been scientifically validated. The main potential impacts are due to landfill gas and leachate. Both are highly complex mixtures and vary from site to site and with waste composition and age of the landfill. It is clear that enough attention has been given to modeling of landfill emission in order to quantify the landfill gas and leachate production. But on the other hand, studies that model the impacts of landfill emission are scarce. A few LCA studies have been performed to compare landfilling with other waste management technologies but by our knowledge an integrated assessment of the impacts of landfills has not been addressed yet. Nevertheless, available literature highlights that the landfills create significant impacts on global warming, eco system, ground and surface water, human health, land value and land availability. In order to minimize the potential risk and environmental burden of landfills and on the other hand to re-introduce the buried resources to the material cycle the landfill concepts should be made operational in the future. Further development of the concepts of landfill bioreactors and enhanced landfill mining can be seen as a promising approach to reduce the environmental impact and the negative socio-economic impacts.

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ASH PRODUCTS AND THEIR ECONOMIC PROFITABILITY

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ABSTRACT

Sustainable whole-tree harvesting practice requires that nutrient removal from the forest is compensated. Wood ashes contain all the nutrients, except for nitrogen, that are found in unburned fuel and can also increase soil pH, which makes ash recycling a natural way to stabilize the nutrient balance and counteract the acidification of forest soils that occurs due to intensive forest management. Several methods for processing ashes into spreadable products have been developed. The aim of this paper is to compare these methods. The study mainly focused on an economic evaluation of production, transportation and the spreading of self-hardened ash, ash pellets and ash granules. Self-hardened ash is generally considered to be the cheapest alternative to manufactured ash products, but these results imply that the most cost effective alternative is ash pellets. Around 27 % of total costs could be earned from recycling the ash by producing pellets and 8 % if granules are produced instead of self-hardened ash. This partly depends on the higher density of the pellets and granules and a significant reduction in the number of transportation operations. The reduction in transportation operations and diesel consumption also has major environmental benefits. Furthermore, it is more efficient to produce granules and pellets than it is to produce self-hardened ash and it is also easier to produce a reliable product of an appropriate size, shape and texture for a market that has well defined requirements.

KEYWORDS

Ash recycling, Ash products, Economic valuation.

1 INTRODUCTION

Whole-tree harvesting has increased significantly (removal of whole trees, not only the stem wood) leading to a loss of nutrients and acid-buffering substances in forest soils. The shortage of nutrients may, in turn, lead to reduced tree growth and have negative effects on runoff water (Wall 2012; Saarsalmi et al. 2001). Wood ash is a concentrate of the nutrients found in unburned trees, except for nitrogen (N). The ash also has a high pH, resulting in a liming effect when spread on soil (Holmberg 2000). The whole concept of ash recycling might seem clear and logical, an environmental disposal solution and the missing link in an ecological cycle. However, the pH of untreated ash is very high and can severely damage the soil and vegetation (Eriksson 1998). Therefore, the ash must be treated by mixing it with water so that the calcium oxide (CaO) reacts with the water and forms the more stable compound, calcium hydroxide ($Ca(OH)_2$), which has a lower pH of, around 10 (Equation 1). The next reaction in the process is where $CaCO_3$ precipitates from $Ca(OH)_2$ in the presence of CO_2 (Equation 2; Holmberg 2001).

$$CaO + H_2O \leftrightarrow Ca(OH)_2 \quad (1)$$
$$Ca(OH)_2 + CO_2 \leftrightarrow CaCO_3 + H_2O \quad (2)$$

In addition to chemical treatment, the ash has to be physically converted into a material that is easy to load, transport and spread. Three common techniques for converting ashes are: self-hardening, pelletisation and granulation. Self-hardening is a process whereby the ash is mixed with water by a mixing screw or pan mixer in the ash silo at a heating plant/or an industry, which creates excess heat. It is then transported to and spread on a paved surface in loaf shaped piles where it is allowed to self-harden for 3-6 months. To improve the carbonation and agglomeration process, the pile is compacted and mixed by wheel loaders. This procedure is repeated several times so that carbonation can take place throughout the whole pile (Steenari & Lindqvist 1997). The ash is then crushed and sieved, resulting in a product with a size ranging from fine dust to agglomerates of several cm in size (*Figure 1*) with a density of approximately 0.75 kg/l (Väätäinen et al. 2010). The disadvantage of this technique is the varying degree of carbonation within the pile, which partly depends on the difficulties surrounding the management of the mixing operations. This might result in ash with a high reactivity and high pH, which can cause injury to vegetation if spread on forest soils (Lindström & Nilsson 1998). The carbonation process is also temperature dependent, so storing the pile outdoors runs the risk of such reactions not to proceeding to completion and there is also the risk that the leakage of easily soluble compounds may leak into surrounding soil and water (Lindström 1996).

Figure 1. Self-hardened ash.

Figure 2. Ash pellets.

(Photos: T. Claesson)

Ash pellets can be produced by compaction where the wetted ash paste is compacted into strings by a press cylinder with grooves of a certain width. The strings are then cut to the desired length (*Figure 2*). The pelletisation equipment can be built-into a container equipped with a set of controls as shown by Windelhed (2000). The apparatus should be able to produce about 5-10 tons of pellets/day. When the pelletisation process is finished, the pellets have to dry for about 1 month if self-dried at room temperature (Windelhed, 2000). Sarenbo et al.

(2009) studied four different pellet techniques: drying at room temperature, drying by hot air (60 and 130 ^{o}C), and drying by flue gas. Drying granules using flue gas has also been tested in a flue gas simulator (Holmberg et al., 2003). The flue gas drying resulted in the lowest pH and the lowest electric conductivity of the pellets, which is considered advantageous when the product is to be spread in the forest. Ash pellets also have a slow leaching rate and a density of 0.98 kg/l (Svantesson 2002).
Ash granules are formed by rolling moistened ash in a drum, disc or mixer. Because granulation in a drum- or a disc produces quite large granules, the material has to be sieved in order to produce a desired particle size distribution with granules having a density of 1.0 kg/l. A binder may be needed to strengthen the granules, usually limestone, dolomite (Holmberg et al. 2000), green liquor sludge (Österås et al. 2005) or cement (Högbom & Nohrstedt 2001). Granules have the slowest leaching rate of all the ash products, which is beneficial because it provides a more even and continuous supply of nutrients to the vegetation over a long period of time, *Figure 3* (Nilsson & Steenari 1996).

Figure 3. Ash granules found in the forest soil 5 years after spreading (Photo: S. Sarenbo).

Figure 4. Ash granules (Photo: T. Claesson).

Granule production can be performed using cement mixers, with or without knives built-in. They are readily available machines, simple to use and time effective. Alternative, cylindrical containers with shovels anchored to a rotating axel can be used to achieve a homogenous result or rotating disc/drum granulators can be used (Nilsson 1993). Intensive mixers, for example the Eirich mixer from Germany, are filled with ash and water and a high speed drum rotates in order to form perfectly round granules (*Figure 4*). There is also a Swedish variant of this type of mixer called Lödigblandare, which works in a similar manner (Rhodes, 1990; Svantesson, 2002).
There are no specific regulations concerning the particle size distribution of ash products that are destined for recycling in Sweden, but the Swedish Forest Agency recommends that their reactivity test should be performed on ash products < 4 mm in size. Practical spreading experiments in Finland revealed that > 6 mm sized particles were spread most evenly when applied to forest soils (Mikko Räisänen, personal communication).
Kalmar Energi AB is a Swedish company that runs two heating plants; “Draken” (41. 64 GWh heat production in 2011) and “Moskogen” (136. 4 GWh electricity and 384. 1 GWh heat

production in 2011) producing nearly 3000 tons of wood fly ash annually as a byproduct of combustion. The Moskogen heat and power plant is new and became fully optional in 2009. Several studies concerning combustion residues from "Draken" have been performed previously (Holmberg et al. 2000, Svantesson et al. 2000, Holmberg et al. 2001, Svantesson et al. 2002, Holmberg et al. 2003, Holmberg et al. 2004, Mellbo et al. 2008, Stålnacke et al. 2008, Sarenbo et al. 2009). There is also fully automated equipment to pelletize the fly ash, but the equipment is currently not in use. The fly ash from the Moskogen heat and power plant is, instead, processed by self-hardening and the final product is spread onto the forest soil. The ash is transported and processed by an entrepreneur responsible for spreading the ash.
The objective of this study was to investigate whether self-hardening is the most economical way to manage the byproducts of combustion for recycling. Every step of the ash recycling chain, from the production of the ash at the heating plant until the ash is spread on the forest floor, was considered.

2 MATERIALS AND METHODS

The Moskogen heat and power plant was chosen as a model for this economic study were we calculated the production costs of 1500 tons of self-hardened ash, pellets and granules during a six-month period. Information concerning the production costs of ash processing and spreading has been gathered from the literature and by interviewing the personnel at the Moskogen heat and power plant and the entrepreneur handling the ashes. Both quantitative and qualitative aspects of the recycling processes were considered. The economic calculations were performed by using Microsoft Excel software.

3. RESULTS

3.1 Ash processing costs

The production costs of approximately 1500 tons of self-hardened ash during six months at the Moskogen heat and power plant includes the costs for the 1300 m^2 paved surface, renting and use of the wheel loaders (105 €/h), as well as the crushing and sieving equipment. Management of the ash pile over 3-6 months, with mixing/compaction of the ash twice, and the administration fees, were included. The ash has a moisture content of approximately 25 % when it is put in the 3 m wide and 3 m high piles for self-hardening. At the Moskogen heat and power plant, the fly ash is collected by an entrepreneur and transported to a disposal site outside the city of Nybro where self-hardening is undertaken (distance 28 km). The entrepreneur outsources the ash transportation operation at a cost of 107.1 €/h. Calculated on the basis of 300 tons of ash and 6. 75 tons of ash/truckload, then 45 trucks are required. This means that the transport costs are 75 € and the cost per ton is 11.2 € (*Table 1).*

Table 1. Production costs for 1500 tons of self-hardened ash (€/ton DM).

Costs related to manufacturing of self-hardened ash	€/ton DM
Costs associated with the paved surface	2.3
Management of the ash pile during the self-hardening phase	3.5
Crushing and sieving	5.8
Administration fee	5.8
Sum A: Ash production exclusively handled at the heat and power plant	17.4
Transportation costs to the storage site; Moskogen-Nybro (28 km)	6.6
Sum B: Sum A + Transportation costs to storage place (28 km)	24

The costs for producing 1500 tons of pellets during six months using a pelletisation machine with a production rate of 5-10 tons/hour included the cost of a paved surface of about 375 m^2, purchase and maintenance of equipment, operation costs and administration fees. A self-drying system was assumed (*Table 2).*

Table 2: Production costs for 1500 tons of ash pellets (€/ton DM).

Costs related to manufacturing of ash pellets	€/ton DM
Costs associated with the paved surface	0.6
Equipment (Depreciation period: 10 years)	2
Operating cost	2.3
Administration fee	5.8
Total production costs for ash pellets	10.7

The production of 1500 tons of granules during six months using an Eirich intensive mixer running at a production rate of 2 tons/hour results in costs for a paved surface (375 m^2), purchase and maintenance of equipment (dosage system with weighing scales, mixers, dispensers and control systems), operational costs and administration fees. If granules larger than 3-4 mm are produced, a plate is needed. A self-drying system is assumed (*Table 3)*.

Table 3: Production costs for 1500 tons ash granules (€/ton DM).

Costs related to the manufacturing of ash granules	€/ton DM
Costs associated with the paved surface	0.6
Equipment (Depreciation period: 10 years)	11.7
Operating Cost	2.3
Administration fee	5.8
Total production costs for ash granules (excluding the plate)	20.4

3.2 Transport costs

Regardless of the ash processing technique, the ash product must be transported from the heating plant to the forest and that distance was assumed to be 50-52 km for the purposes of this study. At the Nybrogrus AB plant, the transport cost over 50-52 km with a boogie car and trailer was 8.4 €/ton + 25 % sales tax. The maximum load was 30 tons or 9 m^3, which meant that 6.75 tons of self-hardened ash can be loaded per truck. Therefor 45 trucks are required to transport of 300 tons of ash product. To transport the same amount of pellets, 34 trucks are required and for granules, 33 trucks are needed. *Table 4* presents the transport costs for each respective ash product.

3.3 Spreading costs

An ordinary forwarder, commonly used for ground spreading of ash is usually loaded with 8 tons of ash at a time (note that this is not the maximum load capacity). The self-hardened ash has a density of 0.75 tons/m^3, so the amount of ash loaded in the forwarder is 11 m^3. The density of granules is 1.0 ton/m^3 and the density of pellets is 0.98 tons/m^3. The amount of granules or pellets loaded are therefore roughly 8.0 m^3 which means that 3 tons more ash can be loaded if granules or pellets are used instead of self-hardened ash. The number of rounds needed to spread 300 tons of self-hardened ash is 37.5, while only 27 rounds are needed to spread granules or pellets. This means that 28 % fewer rounds are needed to spread granules or pellets. A forwarder consumes about 0.8 liters fuel/km or around 120 liters/day so, 28 % less diesel is consumed by driving 10-11 fewer rounds, which means there is a 58 € reduction in diesel costs/day, calculated using the diesel price during Sweden of October 2012 (1.7 €/litre).

3.4 Summary of costs

The differences in total costs between self-hardened ash, granules and pellets are presented in €/ton DM and a cost comparison for different amounts of each ash product are illustrated in Figure 5.All relevant economic factors are compiled in *Table 4*. It is 27 % cheaper to use pellets instead of self-hardened ash and 8 % cheaper to use granules than self-hardened ash, if the total costs involved in the ash recycling process are taken into account.

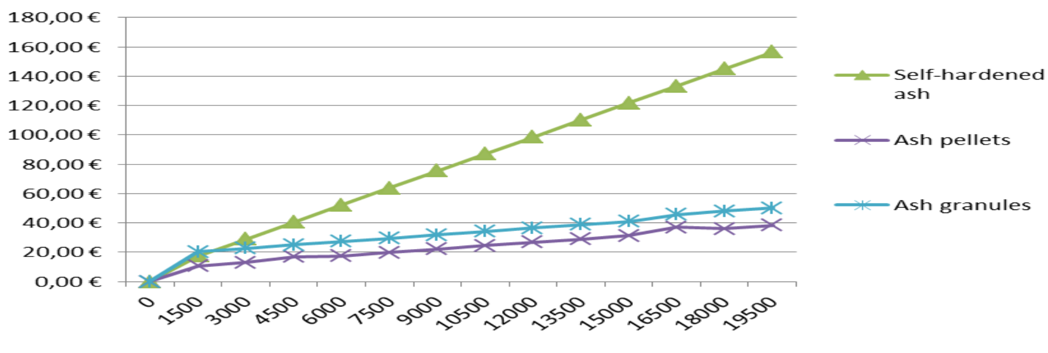

Figure 5. The production costs for self-hardened ash, pellets and granules relative to the amount of ash produced.

Table 4. The costs of the ash products are compared at every part of the ash recycling chain and then summed and presented in €/ton DM and €/ha for the production of 1500 tons of ash.

Costs related to ash recycling	Self-hardened ash	Ash pellets	Ash granules
Processing costs for each ash product (€/ton DM)	17.4	10.7	20.4
Moisture content (%)	25%	25%	25%
Density (kg/l)	0.75	0.98	1.0
Ton DM ash/ha	3	3	3
Ton ash/ha	4	4	4
Costs/ha (€/ha)	69.6	42.8	81.6
Transport cost T_1: Heating plant-spreading site; 50-52 km (€/ton DM)	10.6	10.5	10.4
Transport cost T_2: Heating plant-spreading site; 50-52 km (€/ha)	42.5	41.9	41.6
Spreading cost €/ton DM	22.6	16.2	16.2
Spreading cost €/ha	90.3	64.8	64.6
Sum €/ton DM	50.6	37.4	47
Sum €/ha	202.3	149.5	187.8
Swedish landfill cost 115.7 €/ton			

4 DISCUSSION

The most common method used for processing ash, self-hardening, is not the most economically beneficial choice for the purposes of ash recycling. By producing pellets, the total costs for ash recycling could be lowered by 27 % if 1500 tons of ash is produced during six months. If granules are produced, then the total costs could be reduced by 8 % compared to self-hardened ash, if the same amount of ash were produced. The production of self-hardened ash is cheaper than the granules during the first phase because of the need to purchase expensive granulation equipment. However when it comes to the transportation operations and the spreading phase, granules are the cheapest alternative compared to both self-hardened ash and pellets. The total granule production costs depend on what kind of technique is used. There are several techniques available and other simpler and cheaper granulation techniques exist. The costs also depend on the quantity of ash product produced. As is illustrated in *Figure 5,* the production of self-hardened ash increases exponentially relative to the amount of ash product produced. The large increase in costs is due to the need for a large paved surface in order to produce self-hardened ash. If the amount of self-hardened ash produced is doubled, then the area required to handle the self-hardening is doubled, as is the number of wheel loader operations. This is not the case with pellet or granule production. The surface area needed is not dependent on the amount of pellets or granules produced. Even the transportation operation costs can differ depending on the circumstances. These calculations have been performed from the starting point that all production takes place in the heat and power plant at Moskogen in Kalmar, but currently self-hardening takes place at another site away from the Moskogen heat and power plant, which means that there are large costs for extra transportation operations. The lower costs of using granules or pellets instead of self-hardened ash depend, to a great extent, on the difference in density between the products. On average, 30 % more granules and about 28 % more pellets can be packed for a given weight compared to the self-hardened ash, meaning that the transportation operations and spreading of granules or pellets requires less diesel, which significantly reduces the costs. Based on these results, there does not seem to be any reason not to start producing granules or pellets on a larger scale. In the end, this is just a matter of development and technical improvement.

5 CONCLUSIONS

Unlike what is generally stated about ashes, this study shows that ash pellets are the most cost effective alternative for ash recycling, not the production of self-hardened ash. Even the granules are a cheaper alternative to self-hardened ash and the gap increases relative to the amount of ash produced. With regards to the practical aspects, the granules and pellets have advantages over self-hardened ash because of the lower number of transportation operations that comes as a result of the differences in material density. The reduction in diesel consumption also has considerable environmental advantages. The conclusion of this study is that pellets are the most cost effective option but even the granules are a beneficial choice for a more effective, economic and environmentally friendly solution to the ash recycling issue.

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TOWARDS BETTER UNDERSTANDING OF SUSTAINABLE LIVING IN SPARSELY POPULATED AREAS – A CASE STUDY OF NORDERÖN ISLAND

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ABSTRACT

Much effort presently goes into research regarding ‘sustainable cities’. This is reasonable in light of the globally rapidly increasing population in urban areas. Much less research is directed to understand how to achieve a more sustainable situation in rural areas, but a substantial number of the world population will live in sparsely populated areas also in future. This study contributes to the understanding of environmental impacts and sustainability issues in sparsely populated areas. The island of Norderön in Jämtland, Sweden, is used as a geographically well-defined case study object. Activities on the island has been screened and green house gas emissions from these activities been estimated using LCA methodology. Special interest was given transportation, since it is often argued that this is a significant sustainability issues in rural areas. It is concluded that solutions adapted to each location are needed to develop sparsely populated or rural areas in a more sustainable direction. Solutions developed for urban areas will often not be applicable or efficient if directly transferred to sparsely populated areas.

KEYWORDS

Rural areas, Screening life cycle assessment, Transportation, Green house gases.

1 INTRODUCTION

Much effort presently goes into research regarding ‘sustainable cities’ [1] [2]. This is reasonable in light of the rapidly increasing population in urban areas globally. Much less research is directed to understand how to achieve a more sustainable situation in rural areas, but a substantial number of the world population will live in sparsely populated areas also in future, which is prospected to be one third of world’s population in 2050 [3]. This study contributes to the understanding of environmental impacts and sustainability issues in sparsely populated areas. The island of Norderön in Jämtland, Sweden, is used as a geographically well-defined case study object. Activities on the island has been screened and green house gas emissions from these activities been estimated using LCA methodology [4]. Special interest was given transportation, since it is often argued that this is a significant sustainability issues in rural areas. Rural communities are characterized by dispersed population, low public transportation demand per area, long distances, low accessibility, and high dependence on fossil-fuelled automobile transportation [5] [6] [7].

2 NORDERÖN – THE CASE STUDY AREA

Storsjöbygden, the area surrounding Lake Storsjön and including the island of Norderön, is well suited for agriculture and has hosted the majority of the population of Jämtland County through history. The island, see Figure 1, is classified as area of national interest for culture conservation. Norderön measures approximately 773 hectares and had in 2011 a population of 117 people [8]. Administratively the island is part of the municipality of Östersund and is located about 20 kilometers west of the city. The island has 70-80 permanent households, and 16 summer houses hosting about 100 temporary residents during the summer. To the west of Norderön one finds the Håkansta ferry port and the island of Verkön that is owned by three Norderön islanders but has no permanent residents. To the east of Norderön one finds the island of Isön with its ferry port. There is no public transportation servicing Norderön, except for school busses. The closest bus stop is located about two kilometers from the Isö ferry port on the mainland side. Norderön has a long tradition of agriculture and today one dairy farm [9], one organic grain farm and two potato farms are in operation.

Figure 1. The region of Storsjöbygden. The two ferry routes are marked by red dotted lines, the city of Östersund is marked by grey color and the islands of Norderön and Verkön by dark blue.

Several visitor attractions are located on the island:
• Tivarsgård Dairy and Restaurant: 7,600 visitors, opened in summer of 2011
• Norderö church: one of the oldest in the province
• Norderö golf course: 2,500 visitors per year
• EFS Church's Wilhelmsberg conference centre: 2,000 guest nights per year
• Verkö slott hotel and conference centre: 3,000 visitors ferried by shuttle boat from Norderön, shut down in late 2011

3 GOAL AND SCOPE

3.1 Goal

The goal of this study is to contribute to the understanding of environmental issues in the context of sparsely populated or rural areas, using the community of Norderön as a case study object.

3.2 Functional unit

The functional unit is activities on the Norderön island and activities related to inhabitants of the island during one year.

3.3 Scope

This study investigates the environmental impacts from the community on Norderön, within the municipality of Östersund, Sweden, during one year, using attributional life cycle assessment methodology. The year of study is 2011. The study differs in scope from many LCA studies, since our main interest is studying activities based in a geographic area and relates this to persons living in this area basically for benchmarking purposes to be able to better understand the findings. This means, for example, that we have not made any system expansion for milk or potatoes produced on the island but transported off the island for consumption elsewhere. Since this is an initial screening study we have focused only on global warming potential (GWP_{100}). Emissions included are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). A special focus is put on a better understanding of rural transport and mobility needs.

3.4 System Boundaries

The calculations are made for the year 2011. The geographical boundary includes the island and the ferry routes (shown in Figure 1). There nearby island Verkön has no permanent population but goods to and waste from Verkön is passing through Norderön. Other activities on Verkön are excluded in this study. The church, the golf course, Midgården and Wilhelmsberg have been included regarding their heating demand and electricity use. An overview of activities considered in the screening is presented in Figure 2.

Annual transport work and diesel consumption are the only parameters regarding the ferry routes that was included. For example the energy use for ice breaking to keep the Isö ferry route open during winter has been excluded. Regarding fuels for heating and vehicle fuels, both use (combustion) and production are included. However, such things as car production and maintenance are excluded, as well as building construction, reconstruction (restoration) and maintenance.

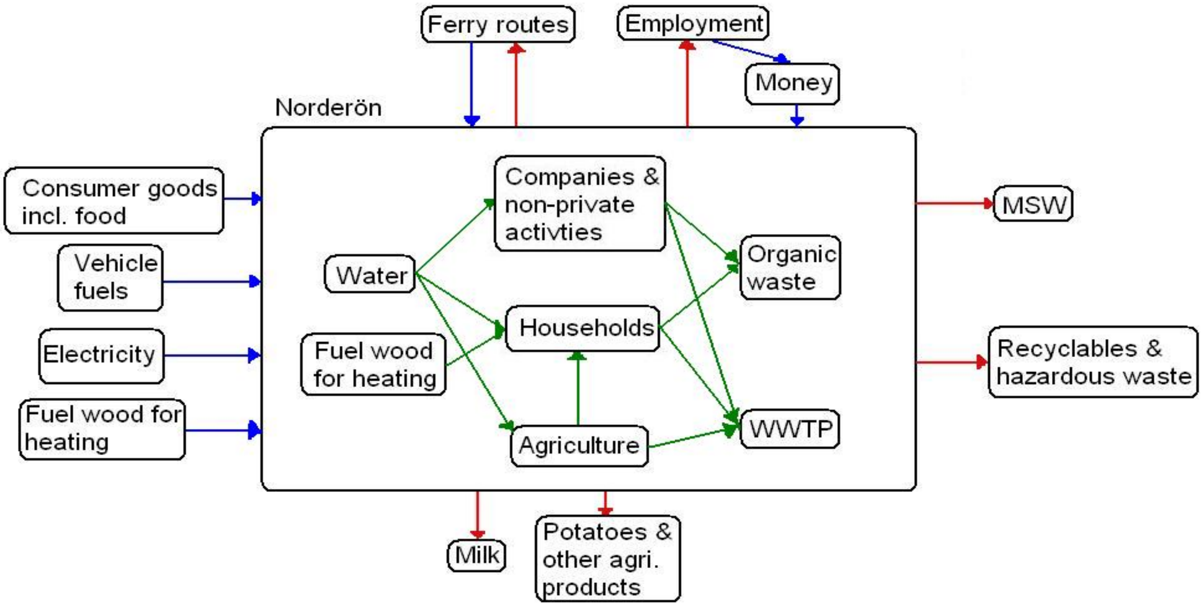

Figure 2. Overview of activities discussed in this study of the Island of Norderön.

3.5 Data collection

The main primary information where gathered by interviewing inhabitants of the island and from companies and authorities. For upstream and downstream activities, generic LCI data from the ELCD database from the Joint Research Center of the European Commission [10] was used. Additional datasets were collected from Probas [11], run by the German Federal Environmental Agency.
The consumptions of households were assumed to be similar to Swedish average consumption. For this screening study we have also assumed that all vehicles run on petrol and that the electricity used on the island is Swedish electricity mix.
Most data used in this study is newer than 2000.

4 INVENTORY

4.1 Electricity

The island of Norderön is supplied with electricity by one cable from the mainland. According to the local power company owning the cable, Jämtkraft AB, and energy balance calculations, the electricity consumption on the island was found to be 1,7 GWh in the year 2011 [12]. The electricity used is described as Swedish average electricity mix (45 % hydro power and 45 % nuclear power) plants, causing 55 g CO_2eq. per kWh [13].

4.2 Heating

There are about 80 houses that are heated all year. The houses were built during several hundreds of years, but in average they were built around the beginning of the 20th century [14]. The most common heating systems are heat pumps (air or geothermal, using electricity) and fuel wood heating by stoves and boilers. Based on interviews, it was estimated that 54 houses on the island are heated using electricity (covered by the electricity use described above) and 26 houses are heated with different kinds of locally produced wood fuels (fuel wood, wood chips etc.).

The average heating consumption of a Swedish one family house is about 20 MWh/year [15] and this figure has been used for estimate wood fuel consumption. Wood burning was assumed to have an efficiency of 70% using pinewood with 70% dry content and heating value of 3,44 kWh per kg. To heat 26 houses 216 tonnes of wood are needed per year. Assuming the carbon dioxide balance between tree growth and combustion is zero, because the wood binds as much carbon dioxide as it emits. The fuel wood production is based on a study producing reforested pine wood with a water content of 44% [10]. This gives a per capita emission of 12 kg carbon dioxide per year for wood fuelled heating. The rest of the emission due to heating is covered by the electricity delivered to the island.

4.3 Transportation and fuels for vehicles, ferries etc

Transport demand on the island can be roughly divided into three key groups: islanders, visitors and through traffic. A large quantity of the through traffic on Norderön is skiers going to Bydalsfjällen ski resort. Personal cars, tractors, snowmobiles motorcycles and other fuel based transportation units are the types covered by this study. The amounts and use of different vehicles are based on a study by Lanker [16]. Long distance vacation travel is not included in the study. Two different methods were used to estimate transport fuel use on Norderön: one based the data on the traffic on the ferries and the other based it on the questionnaire and oral sources, leading to similar results. It is assumed that all vehicles run on petrol except the tractors. Regarding fuel both use (combustion) and production are included.

Only the fuel used by the people living on the island and visitors is seen as emissions allocated to the functional unit. Emissions caused by through traffic are excluded. The amount of diesel used to run the ferry is included.

The consumption of diesel and petrol fuel from personal cars, tractors and heavy machinery, snowmobiles and similar equipment and the ferries is based on the inventory described in the thesis by Lanker [16]. The amount of fuel consumed on the island is 17 TJ (473 m^3) of diesel and 3,6 TJ (109 m^3) of petrol, respectively. The emissions from diesel and petrol production are based on data sets in the Probas database for European production in the year 2010 [11]. Only CO_2 is included regarding the combustion of the fuels.

4.5 Agriculture

Agriculture is rather prominent on the island: two farmers are working together with milk farming, two farmers are having potatoes, one farmer is utilizing low-intensive farming with different animals and one farmer has small-scale meat production with Highland cattle and sheep. A number of households are leasing out fields for grazing to the active farmers [14]. The report considers the milk and the potato farms, but not impacts from other minor farming or private gardening activities (except as a generic deduction in amount of store bought food per person compared to the Swedish average, see below).

Tivarsgård milk farm is the largest farming activity on the island. The farm has 140 ha crop fields and pastures. The crew is 135-140 cows, including recruitment. During one year 580 m^3 milk is produced [9]. A LCA study on milk farms in Northern Sweden [17] was assumed to sufficiently describe the Norderön milk farming activities resulting in 600 tonnes CO_2 eq. per year. Similarly, potato farming producing 175 tonnes of potatoes annually was described using data on potatoes from the Probas data base [11] resulting in 16 tonnes CO_2 eq. per year.

4.5 Consumer goods and food

The consumption of consumer goods and food was assumed to be similar to Swedish average. A discounting factor of 5% was introduced for the GHG emissions caused by food consumption, to take food production on the island into account. In 2003 the Swedish average emissions caused by private consumption of food was about 2,3 tonnes per capita and year and for consumer goods about 1,3 tonnes [18].

4.6 Municipal solid waste

The islanders generate approximately 208 m^3 (36,4 tonnes) of municipal solid waste (MSW) and 20 m^3 organic waste every year [19]. Most of the municipal solid waste is transported via Gräftåsen waste disposal plant in Östersund to Korstaverket waste incineration plant in Sundsvall, located 200 km from Östersund. The organic waste is separated from the MSW at collection and treated locally, without transportation off the island. The reason for the local treatment is to prevent transport to the mainland. This practice started in 2010 [20]. Recyclable materials and hazardous waste was disregarded in this screening study.
The emissions caused by the transportation of the waste are included in the diesel consumption. The GHG emissions from waste incineration in Sundsvall are based on emission data in the ELCD database [10]. As a result, 12 tonnes of carbon dioxide equivalent or 0,1 tons per capita and year were emitted.

4.7 Wastewater

Wastewater treatment is done on the island by wet composting [9]. Impacts included in this study were caused by transportations needed and electricity used by the facility and are included in the vehicle fuels and electricity estimates.

5 RESULTS

This report only considers global warming potential.

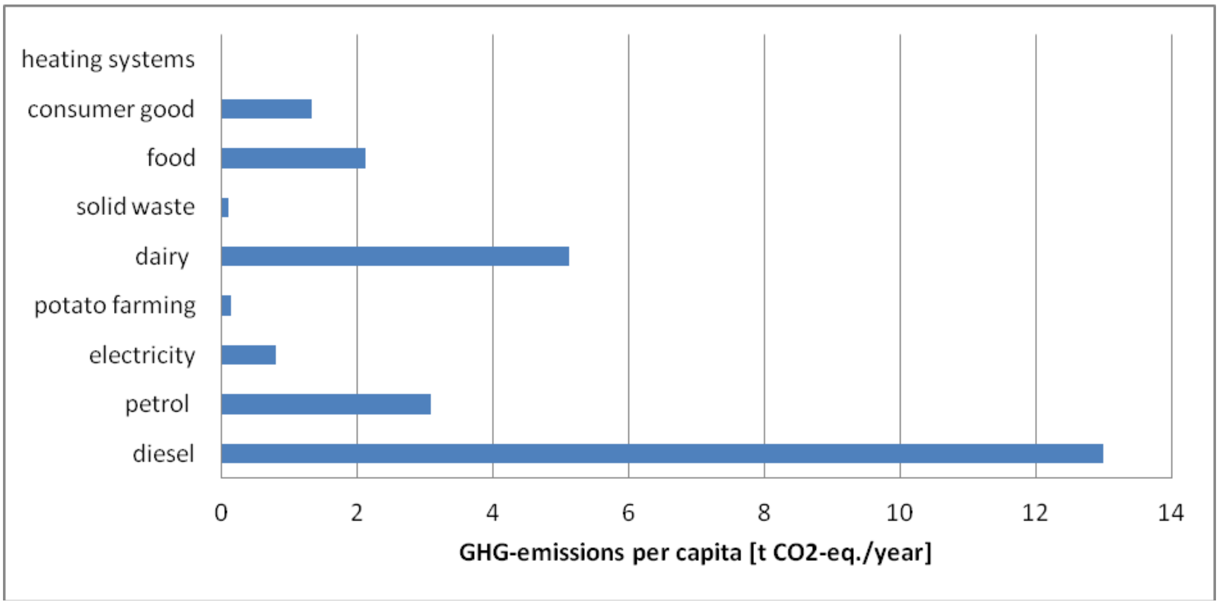

Figure 3. GHG emissions from different source.

Total emission of green house gases from activities connectd to Norderön in 2011 was estimated to 3011 tonnes of carbon dioxode equivalents. In Figure 3 the shares from diffrent sources reported per capita of the 117 inahbitants can be seen. The two largest sources of climate impacts are the vehicle fuels used (diesel and petrol) and the milk farming activites sector with 70% of the total carbon dioxide emission on Norderön. The running of the ferries result in about 70% of the consumption of vehicle fuels and and their connected environmental impacts.
In Figure 4 the Norderön results are benchmarked to Swedish average, when including and excluding the dairy and ferry.

6 DISCUSSION

A large part of the green house gas emissions are as expected connected to transportation activities, but it is also obvious that the magnitude of such emissions can be related to very local situations. The emissions caused by the Norderön related activities per capita are almost three times the Swedish average for the private consumption according to the Swedish EPA of about 9 tonnes per capita and year [18].
The two ferries and the dairy farming activities give significant contributions. If the ferries and the dairy farming activities are excluded, the carbon dioxide emissions are around 10 tons per capita and year, which is slightly higher than the Swedish average.

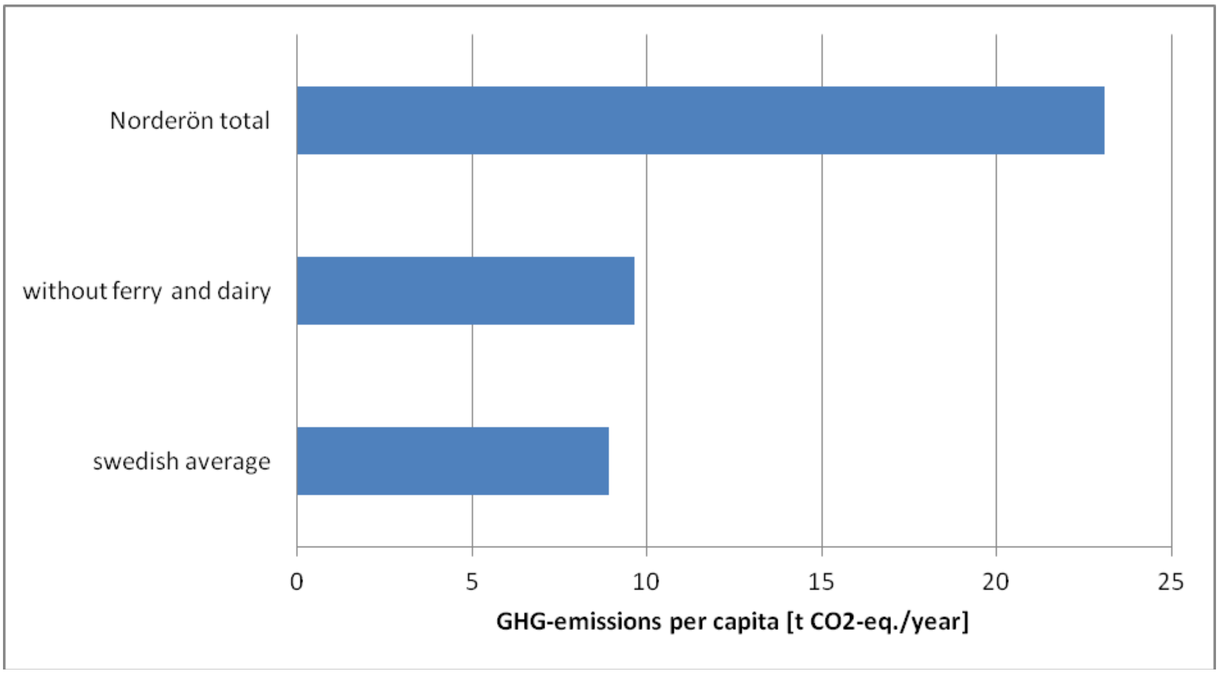

Figure 4. Emissions of Green house gases from the studied activities on Norderön Island benchmarked toward total carbon dioxide equivalents of Swedish average consumption.

In a narrow interpretation of LCA type results, this can be seen as an indication that the ferries should e.g. be replaced by a bridge. However, it should be kept in mind that such solutions are in themselves not without environmental impacts or consequences (e.g. concrete use, increased traffic). These types of interpretation also raise the question if we should have people living in rural areas at all. In a more holistic type interpretation, the same result could be seen as an indication that we will need a multitude of approaches adapted to each local situation, and that proposed solutions based in the idea of many people sharing the use of an artifact will not be applicable to less densely populated or rural areas. Another example could be the idea that transportation emissions should be curbed by public transportation. This is often feasible in a city, and the question then becomes how to get enough people to use it, but seldom in rural areas, where the density of people to utilize the public transportation is so low that it becomes an exceedingly expensive solution.

The second large contributor to total green house gas emissions in the geographical area Norderön is the agricultural activities. This highlights a problematic side of the 'simple solution' that we to curb green house gas emissions should just concentrate all population to large cities. However, since food production, as well as other area demanding activities like e.g. hydro power or forestry will still be needed in future, we will still be needing ideas for how people working with such activities should be able to live with good quality of life in a sustainable manner in rural areas where such production will take place. To have social sustainability in rural areas in general the population cannot be too low, e.g. because different services cannot be supported with a too low population (like schools, health care etc.) and vitality in general might diminish.

Fossil fuels are not only used for transportation of people but also for tractors, working machines etc. in agriculture and forestry, and for snowmobiles et c. Such fuel use is even less likely to be reduced by 'public' solutions than transportation of people, and is also less easy to change by switching from liquid fuels into electricity. These areas have so far not gained as much interest regarding R&D as fossil free private car solutions. Further discussion on issues regarding more sustainable transportation in sparsely populated areas, with Norderön as a specific case study, can be found in Lanker 2012 [16].

More research is needed in the field of solutions for how to increase sustainability in sparsely populated and rural areas. We can see this regarding the developed world from this case study, but it is also obvious that we urgently need sustainable solutions for better lives for millions of poor people in rural areas in developing countries.

7 CONCLUSION

To develop sparsely populated or rural areas in a more sustainable direction, solutions adapted to each location are needed. Solutions developed for urban areas will often be not applicable or efficient if directly transferred to sparsely populated areas. More research is needed in the field of solutions for how to increase sustainability in sparsely populated and rural areas.

ACKNOWLEDGMENTS

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SEQUESTRATION OF ORGANIC MATTER IN MSW LANDFILLS – A PROCESS TO BALANCE ANTHROPOGENIC CO_2 EMISSIONS

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ABSTRACT

The annual, global emissions of greenhouse gasses are estimated at about 49 billion tonnes of carbon dioxide equivalents. In Sweden domestic road transports account for the most pronounced share of emissions, approx. 34%, while only 3 percent come from the waste sector (uncontrolled landfilling, waste incineration, sewage treatment, a.s.o.). The annual accumulation of organic carbon in the World´s landfills has been estimated to be around 100 x 10^6 metric tons of C. During landfilling most of the carbon in lignin (from paper or wood) and all organic carbon in fossil derived products, like plastics, synthetic rubber, synthetic textiles, a.s.o., will be brought back to long-term accumulation. If more than about 60 % of the produced landfill gas is collected, the sequestration of resistant organic matter in landfills has a net positive effect to counteract global warming. In a well-controlled bioreactor landfill around 90 % of produced biogas can be collected.

Today in Sweden, landfill gas is extracted from about 50 of the landfills in operation. This generated over 310 GWh of which 24 GWh in the form of electricity. This clean and non-polluting bio-fuel can substitute fossil fuels and thus counteracts emissions of fossil derived carbon dioxide.

A landfill reactor cell, treating approximately 100 000 tons of waste per year, and where the fermentation residues are left in the landfill, a persistent organic fraction corresponding to about 45 000 tons of carbon dioxide remains long-term accumulated each year. This compensates for the annual carbon dioxide emissions from 15 000 cars, provided that each car runs 15 000 km per year with fossil fuel. With another comparison, the deposition of organic matter in a medium sized controlled landfill (100 000 tons per year) equals the total amount of carbon in approximately 65 hectares of grown-up spruce forest, or approximately 45 hectares of deciduous forest.

Energy forests within a landfill area, planted as receptors for leachates, also immobilize organic carbon in standing biomass, corresponding to about 10 metric tonnes /ha which can be stored in plant biomass or soil organic matter each time unit.

Reliable techniques to measure actual emissions of methane from landfills must be introduced and limits for uncontrolled emissions must be introduced instead of e.g. bans on landfilling of organic matter.

KEYWORDS

Reactor landfill, Bioreactor cell, Biogas, Climate change, CO_2, Carbon sink, Carbon sequestrating, Leachate.

1 INTRODUCTION

Since the 1970´s, the global CO_2 emissions have increased by 70 percent, and the World's total annual emissions of greenhouse gases is estimated at about 49 billion tonnes of carbon dioxide. From 1990 the increase has been approximately 24 percent. In Sweden it is mainly domestic road transports which account for the most pronounced share of emissions, approximately 34%. This is followed by emissions from industry, electricity and heat production and agriculture. Emissions from residential and commercial buildings account for 6 percent of the emissions, while only 3 percent of the emissions come from the waste sector. This sector reported methane emissions from uncontrolled landfills, carbon dioxide emissions from waste incineration, and nitrous oxide emissions from waste incineration and from wastewater management. Total greenhouse gas emissions in Sweden corresponded to about 59.8 million tons in 2009 converted to carbon dioxide units, while the Swedish emissions of the gas carbon dioxide itself were 46.6 million tonnes during the same year. EU has declared that greenhouse gas emissions shall decrease by at least 20 percent by 2020, the share of renewable energy should meet 20 percent of all energy in the EU 2020, biofuels should represent at least 10 percent of total fuel use in transport sector by 2020 and energy use will decrease by 20 percent in 2020 [1].

Production of a renewable fuel in the form of biogas through anaerobic digestion offers significant advantages. It has been evaluated as one of the most energy-efficient and environmentally beneficial technologies for bioenergy production. It can drastically reduce greenhouse gas emissions compared to fossil fuels by utilization of locally available resources [2]. Techniques for biogas production thus are promising means of achieving multiple environmental benefits and producing an energy carrier from renewable resources. Replacing fossil fuels with biogas normally reduces the emission, not only of greenhouse gases, but also of nitrogen oxides, hydrocarbons, and particles [3]. Anaerobic digestion and the production of biogas can provide efficient means of meeting several objectives concerning energy, environmental and waste management policies. Interest in biogas is increasing, and new facilities are being built. There is a wide range of potential raw material, and both the biogas and digestates can be used in many different applications. The variation in raw materials and digestion processes contributes to the flexibility of biogas production systems, but at the same time makes their analysis and comparison more complicated [4].

A goal has been set by the Swedish Environmental Protection Agency that at least 35% of food waste from households, restaurants and stores must be recycled through biological treatment. Assumptions are made that at least 60% of all food waste in Sweden can be available for biogas production. This amount corresponds to approximately 760 GWh annually, and represents 7% of the total biogas potential. The total biogas potential from all food waste in Sweden amounts to 1346 GWh / year [5].

Collection of biogas from landfills and reactor cells also helps to solve a waste problem in an environmentally feasible way. Today landfill gas is extracted from most of the major landfills in operation in Sweden (about 50). These landfills generated over 310 GWh of which 24 GWh in the form of electricity. In addition to this, about 65 GWh is flared away. Provided that a reliable and efficient biogas collection system is installed, a strictly controlled landfill, where at least 60-62 % of the biogas is collected, represents a technology to combat global warming. New reactor-landfill technologies, in e.g. Sweden, have shown promising results in collecting up to over 90 % of the produced biogas in the landfill cell. According to experiments in Sweden, approximately 150-250 m^3 of biogas per tonne of waste can be extracted from a landfill reactor cell over a 10 year period [6].

Major waste companies in Sweden have invested significantly in biogas production through efficient digestion of waste in various forms of biological treatment processes (reactor fermentation, landfill reactor cell fermentation and from the original landfill). The production of landfill gas at the Northwest Scania Recycling Company in Helsingborg, South Sweden, is enough from residual solid wastes to warm up about 3500 homes, which means 70 000 MWh of landfill gas [7]. Due to a rapidly increasing demand for renewable motor gas, landfill gas is also refined to motor fuel, and a new plant is recently being built in to liquify refined biogas for optimized transportation to consumers outside the grid network (e.g. bus companies). A similar trend is seen in many countries. At the same time as the residual waste is used to produce an environmentally friendly motor fuel, this creates economy for a reliable collection of produced gas in the landfill, and thus eliminated diffuse gas emissions to the atmosphere. At the same time nutrients can be extracted through the leachates to be used in e.g. energy crop production.

It is important to establish reliable technologies for extraction of landfill gas, to minimize the risk for diffuse emissions. In order to prevent the release of the methane from landfills, many countries, like eg the US, are introducing a system for trading with carbon emission units related to landfill gas. This stimulates the construction of reliable biogas collection systems in landfills, both for operating landfills and closed ones. Some waste companies in the US also includes carbon sequestration in the landfill in these calculations.

Some European countries, however, instead have introduced a ban on landfilling of organic matter, but at the same time have no regulations for emissions from old abandoned landfills. This will not reduce the emissions from existing landfills, and it will lead to a retarded technical development for safe extraction of biogas from residual solid wastes in landfills. Also with a ban on landfilling of organic waste, approximately 10 % organic matter will still be present in the landfilled waste and can act as a source for methane emissions. Storage of waste, as a result from these regulations also cause other environmental problems in the form of increased risks for frequent wild fires in waste storages, which can give significant air pollution problems.

A ban on landfilling of organic matter will also in the long-run lead to shortage of available biogas for energy production and for the production of motor fuels. Much of the organic matter in the residual waste, rich in e.g. cellulose needs a longer turn-over time than normally is provided from faster fermentation techniques. Instead of a ban, an upper limit for CO_2 emissions from landfills should be set up e.g. in the EU legislation, to be implemented by the member countries. Modern techniques today allow reliable measurements of gas emissions over large surfaces.

2 ACCUMULATION OF LONG-LIVED ORGANIC CARBON IN LANDFILLS

Accumulation of organic carbon in the World's landfills in the 1980´s was estimated to around 100 x 10^6 metric tons of C per year [8,9]. However due to increased material recovery and increased incineration the resent figure is probably somewhat lower, in spite of increased waste volumes. The size of the long-term accumulated fraction depends on the conditions for decomposition in the landfill. Estimates by Bogner and Spokas [10,11] indicate that the fraction left for long-term accumulation in the World's landfills amounts to approximately 30 x 10^{12} g C per year.

The total annual amount of mixed municipal and industrial waste that was landfilled in Sweden before the ban on landfilling of organic matter was introduced in Sweden in 2002 could be estimated to around 3 x 10^6 metric tons [12], corresponding to approximately 0.75 x 10^{12} g C. Of this fraction about 0.40 x10^{12} g organic carbon can be regarded as resistant to degradation under normal landfill conditions. At the same time about 2 x 10^6 ton of waste was incinerated each year in Sweden, resulting in a release of about 0.50 x 10^{12} g C. A fraction of about 0.30 x 10^{12} g C would have been more or less long-term accumulated if this amount of waste instead would have been landfilled or treated in landfill reactor cells. Today most of this long-lived carbon fraction is rapidly released to the atmosphere through incineration.

With optimized landfill gas recovery and reactor cell fermentation techniques approximately 150-200 m^3 of biogas, with 50-55 % methane gas concentration, can be extracted per ton treated waste with the first generation of a reactor cell fermentation technique. With 25 % organic carbon per ton household and light industrial waste, and with a total carbon content of 500 g C per m^3 of biogas, this would mean that around 130 x 10^3 g C per ton originally landfilled waste should remain in the landfill after approximately 15-20 years. Of this amount lignin contributes to approximately 40 - 50 x 10^3 g C and plastics to about 20-30 x 10^3 g C. The remaining fraction, about 40 - 50 x 10^3 g could theoretically be converted to biogas after improving and optimizing the fermentation technique in e.g. landfill reactor-cells (biocells). The amount of total organic carbon discharged from landfill reactor-cells through the leachates is small in comparison to the losses through the biogas, around 3-6 % [6]. Accordingly, approximately 70 - 90 x 10^3 g organic carbon per ton household waste would remain in the fermentation residue after advanced fermentation in landfill bioreactors, compared to about 130 x 10^3 g C with the present technique for optimized biogas production. In a normal well-managed landfill with biogas extraction probably around 150 x 10^3 g C is long term accumulated per ton landfilled waste. Household waste normally contains approximately 23-25 % organic carbon, while industrial waste, with lower water content and normally a higher proportion of paper, wood and plastics, contains somewhat higher proportions of organic carbon. Plastics and rubber are rather unaffected by biological degradation and is left in the landfill. Under anaerobic conditions also lignin is resistant to degradation and will remain in the fermentation residue. Thus, around 60 % of the organic carbon in the solid waste remain in the fermentation residue after normal landfilling, while about 30-50 % of the original carbon content would remain after optimized landfill reactor-cell (biocell) treatment. These figures are somewhat lower compared to some earlier estimates based on landfill fermentation studies, e.g. [10,13,14], which range from 60-75 % for the remaining carbon fraction.

In Sweden, an average sized landfill bioreactor, before the ban on landfilling of organic matter, annually could be assumed to treat about 100 000 ton residual municipal waste. This amount of waste corresponds to approximately 25 000 ton organic carbon, or a fraction of approximately 15 000 x 10^6 g C each year which will be included in a long-lived fraction. This corresponds to about 45 000 tons of carbon dioxide. The losses of organic carbon through the leachates only amount to a minor fraction. The long-lived carbon fraction accumulated in the fermentation residue thus corresponds to the total amount of carbon emitted per year from 12 000 – 15 000 cars per year, running approximately 15 000 km per year and emitting approximately 212 g CO_2 per km. In addition to the accumulation of organic carbon resistant to mineralization, landfilling or treatment in reactor cells, also means an increasing amount of organic matter, which in spite of microbial degradation will be accumulated during a few years. Thus the positive effects on the carbon balance are not only related to the most long-lived fractions with fossil origin, which finally will be stored for a very long period of time.

With another comparison, calculated that the deposition of organic matter in a medium sized controlled landfill (100 000 tons per year) equals the total amount of carbon in approximately 65 hectares of grown-up spruce forest, or approximately 45 hectares of deciduous forest. It is of great importance to establish carbon accumulating functions in the urban society, as the natural CO_2 balancing processes are insufficient to compensate for the increasing emissions of carbon dioxide [15,16].

3 LANDFILL GAS AS A MOTOR FUEL

Landfill gas extraction in Sweden 2008 was approximately 370 GWh. Mainly because of lack of available technologies for landfill gas upgrading and high assessed upgrading costs, landfill gas has so far only been used for heating and cogenerations plants (CHP). In recent years, interest has been brought to upgrading landfill gas and this facilitates the possibility of using landfill gas as fuel for vehicles [18]. In Stockholm region in Sweden (called biogas east region) there was approximately 32 million Nm^3 of landfill gas produced in 2008. This means that over 7000 cars and 250 buses can run on CNG Biogas from landfills in this region if the gas was upgraded [17].
Cost-effective upgrading of landfill gas is important, but it normally requires relatively large landfill gas flows [18]. An important factor is the treatment of smaller residual gas flows that can be used for energy production, possibly after addition of fuel. While comparing landfill gas upgrading, heat production and combined heat and power production (CHP), the value of vehicle fuel, heat and electricity are very important for the overall economy of each system. An important parameter is also how much of the generated heat that can be sold over the year. In comparison between landfill gas upgrading and upgrading of biogas from reactor digesters, the investment cost is twice as high for landfill gas upgrading as a result of higher operational and capital costs, and a lower initial methane gas concentration. When the raw landfill gas is valued low, there is room for higher upgrading costs since the digester gas has a higher production cost. In an example for a landfill gas flow of 750 Nm^3/h with a methane content of 46% (~ 30 GWh/year), the landfill gas upgrading cost is estimated to 0.026 Euro/kWh [18]. Liquid Biogas has about three times as high energy density as compressed biogas [19]. In order to maintain the biogas in a liquid form it must be frozen to -160 degrees C, which is not possible to use for tanking of cars. This means that biogas should be converted into ordinary compressed gas at gas service stations, before re-fuelling of cars can be done. One of the advantages of liquid biogas is that transportation to the gas stations will be more efficient.
The largest upgrading units for landfill gas today are found at Claye Souilly outside Paris in France. However one of the largest units in Europe is soon to be built in Helsingborg, South Sweden. This upgrading facility to produce liquid biogas, to be built in Helsingborg, will have a production of biogas corresponding to 15 million litres of liquid biogas per year (equivalent to about 60 GWh) for at least 20 years with mainly the already landfilled waste as raw material. It would present a fossil carbon dioxide reduction by 40 000 metric tones of CO_2 per year compared to conventional diesel [7].
The future of liquid biogas is positive, especially in the regions where there under the current situation is no existing natural gas grid available. For the transport of liquid biogas conventional LNG trucks (Liquid Natural Gas) can be used.

4 LEACHATE IRRIGATION AS A TECHNIQUE TO INCREASE THE CARBON STORAGE IN SOIL AND VEGETATION

Some organic fractions accumulated in a landfill are resistant to anaerobic decay, e.g. substances derived from the lignin in paper and wood material. This organic matter helps to retain water in the fermentation residue and maintain high moisture content. This will provide for a reliable long-term storage in the fermentation residue of heavy metals bound up as sulphides, which are insoluble under anoxic conditions. Thus, in the leachate mainly metals with a low atomic weight, as nutrients like sodium, potassium, magnesium and calcium, will occur together with nitrogen fractions. Thus, long-lived organic matter from landfilling of e.g. paper and wood products will act as a stabilizer for the landfill and immobilize toxic elements in a rather stable fermentation residue, which is left in the landfill. This chemical separation will open possibilities for nutrient separation and recovery through the utilization of the leachates as fertilizer in e.g. energy forests situated within the controlled landfill area.

Nutrients from the leachates contribute with macro and micro nutrients to the trees, while dissolved organic matter in the leachates is decomposed by soil organisms [20]. An optimal growth is necessary to maximise the nutrient uptake. After a heavy irrigation with leachates the soil becomes waterlogged. When a soil is flooded, water occupies the soil pores, causing almost immediate deficiency of soil O_2 [21]. Most plant species develop injury symptoms, such as wilting, in a few days of water-logging, since the root respiration is disturbed. A drop in redox-potential as a result of water-logging can thus be responsible for tree injuries [23,23], and water-logging can lead to reduced growth [21] and decreased uptake of nutrients [24], which counteracts the purpose of the vegetation filter.

After irrigation of a forest stand with leachate ain increased biomass production has been found, which means that more organic carbon is accumulated here than in reference areas. Thus, forest irrigation means that a carbon sink is created, which e.g. can compensate for anthropogenic CO_2 emissions, e.g. from traffic. The technique thus gives an additional environmental effect than just to clean the leachate and prevent the release of eutrophic substances into water systems. This increased amount of organic carbon bound up in a full scale forest plantation, irrigated with leachates, can be comparable with the annual CO_2 emissions from up to 3 000 - 5 000 cars.

If the wood biomass is used as a fuel, the ashes, containing most of the nutrients, can be used as a fertilizer or vitalizing agent in normal forestry. This in turn opens the possibilities to use a larger proportion of the logging residues from forests as a renewable fuel, decreasing the emissions of fossil CO_2 even more. This circulation of nutrients will not be possible after waste incineration, as in this case the nutrients cannot easily be separated from toxic heavy metals, and the total amount of mixed ashes have to be long-term stored in landfills. Ashes from waste incineration mostly are classified as toxic waste.

Planting of an energy forest within a landfill area also immobilizes CO_2 in standing biomass. In such a forest plantation, supplied by nutrients extracted from the waste when using the leachates for irrigation, organic carbon corresponding to approximately 10 metric tonnes of organic carbon can be stored in plant biomass per hectare each time unit, in spite of a short turnover time for the produced biomass. The content of total organic carbon (C) also increases in soil, mainly in the humus layer, after irrigation, which is an effect of a larger litter production from a gradually increasing biomass. Even a slight increase can be measured of the carbon content in the uppermost mineral soil layer, which depends on leaching of resistant humic substances. This means that leachate fertilization leads to an increased amount of organic carbon that can be accumulated in the soil, which is positive from a carbon balance perspective [25].

Also other techniques for biological waste treatment, producing compost or bio-residues, where a fraction of the organic matter is rather resistant to degradation, are beneficial for the carbon dioxide balance. The use of compost or bio-residues in sandy soils, or in soils with high clay content, will increase the humus concentration and thus the contents of organic carbon present in the soil.

5 CONCLUSION

- With modern landfill techniques, in improved landfill reactor cells, over 90 % of the produced biogas can be collected and used to substitute fossil fuels.
- Reactor cells constructed in landfills act as a carbon sink for long-term sequestration. Provided produced methane is effectively collected (more than approximately 60% of the produced gas), well-controlled landfills can counteract global warming.
- Organic matter in the landfill also stabilizes biochemical processes and is important to minimize leaching of heavy metals.
- Leachate irrigation to a short rotation forest can increase the capacity of the vegetation to act as a carbon sink. This is both related to increased standing biomass and increased concentrations of organic matter in the soil. This increases the effect of landfills as carbon sinks, and compensates for emissions of fossil carbon dioxide (e.g. from traffic).
- A MSW landfill, receiving approximately 100 000 tons of waste per year, and with biogas extraction and leachate irrigation in a forest plantation, can compensate for the annual CO_2 emissions from around 15-20 000 cars.
- A carbon trading system, including the waste sector, is a way to stimulate an effective control of diffuse methane emissions. Limits should be based on improved actual, technical measurements. On the other hand a ban on landfilling of organic matter has negative effects in the long run on reduced green-house gas emissions, as it retards technical development and the willingness to invest in improvements for an effective landfill gas extraction from already existing landfills.

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ECONOMIC MODELLING IN WASTE MANAGEMENT

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ABSTRACT

Money is an efficient tool for the successful steering of waste management. If there's just an economic incentive to have something made, it will take place. The versatile EUROPE model based on the *equality principle* invented by Jan Stenis is a novel economic instrument that facilitates necessary economic incentives for most kind of industrial and public actors to at the same time improve their profitability, their technology and the environment by employing so called *shadow costs* that are allocated to unwanted residual fractions of any kind in order to obtain economic incentives to reduce and/or optimize them. Also, the EUROPE model is useful as a tool for monitoring the performance of the activity in question and for evaluation, meaning that management obtains a device for in quantitative terms measure the improvement of the resource-efficiency of companies. The *equality principle* has successfully been applied on, for example, mechanical work-shops, construction industry, bulk industry, baling plants and ore mining. Now, the authors work on applying the *equality principle* to optimize the usage of energy resources, monetary flows and the global distribution and equity of natural resources in general through promoting the redistribution of natural resources and commodities most for those actors that need it most. The EUROPE model is hoped to be useful for optimizing the industrial economy in these contexts including optimization of the flow of currencies in the global perspective. In doing so, this novel model promotes the long run sustainability of industrial and other human activities as regards optimization of the flows of material goods as well as energy-flows. Thereby, the *equality principle* also promotes the harmony of society by inducing an improved equity.

KEYWORDS

The *equality principle*, The EUROPE model, Resource economy, Waste management.

1 INTRODUCTION

The common use of the *equality principle* [6] is believed to promote societal matters and the material welfare through imposing economic incentives for people, companies and all kind of actors to perform well from mainly a technological, an environmental and a financial point of view. Also, the common application of the *equality principle,* whenever it is possible, is also believed to contribute to the long-run survival of our own species, the general and ultimate goal of Homo sapiens being an unusually adaptive organism among many others in an ordinary, cosmic eco-system that reaches to the end of the whole, known universe.

If mankind just has the adequate resources, humans namely stand a fair chance to survive, in one form of life or the other. Resources could be financial, material and/or immaterial, such as energy-supply. The *equality principle* is believed to be applicable on all kinds of resources. Therefore, in this paper is outlined a review of both already successfully tested applications of this model as well as novel applications for the future.
The approach of the paper is descriptive. An analysis is presented of the many promising possibilities to apply the *equality principle* in different areas of interest to a broad audience. Thus, no results or conclusions are presented.
Instead, emphasis lays here on showing the past and future possibilities for application of the *equality principle*. Thereby, material, energy and monetary flows are examined.

2 THE EUROPE MODEL BASED ON THE EQUALITY PRINCIPLE

2.1 Background

Disposal of industrial waste often creates serious environmental problems. In the long run, these problems need to be eliminated, or markedly reduced, so that nature can be kept as clean as possible.
This calls for the clarification and establishment of links between company profits, as expressed in consolidated profit and loss accounts, and both the avoidance and proper utilisation of waste. A new way of looking at waste, or, to a certain extent, a shift in views, is needed. Otherwise, the process of achieving environmental cleanliness in industry can be too slow for gaining public acceptance.
A comprehension argued for here involves equating industrial waste with normal products as regards revenues and costs, an approach that is termed **the equality principle**. A framework upon which such a principle, oriented to the maintenance of a sustainable development, can be based is described in the paper.
In line with this, this study suggests that the waste fractions studied are to be regarded as a kind of company output. This is mathematically considered by adding the sum of the actual quantities of a certain kind of waste belonging to a certain waste fraction set up scenario, to the quantity of normal production output in the denominator in the following expression (1) which is to be used for the allocation of revenues and costs to a certain waste fraction through multiplication by the costs or revenues in question that are to be allocated by splitting them up in their proper proportions [11].

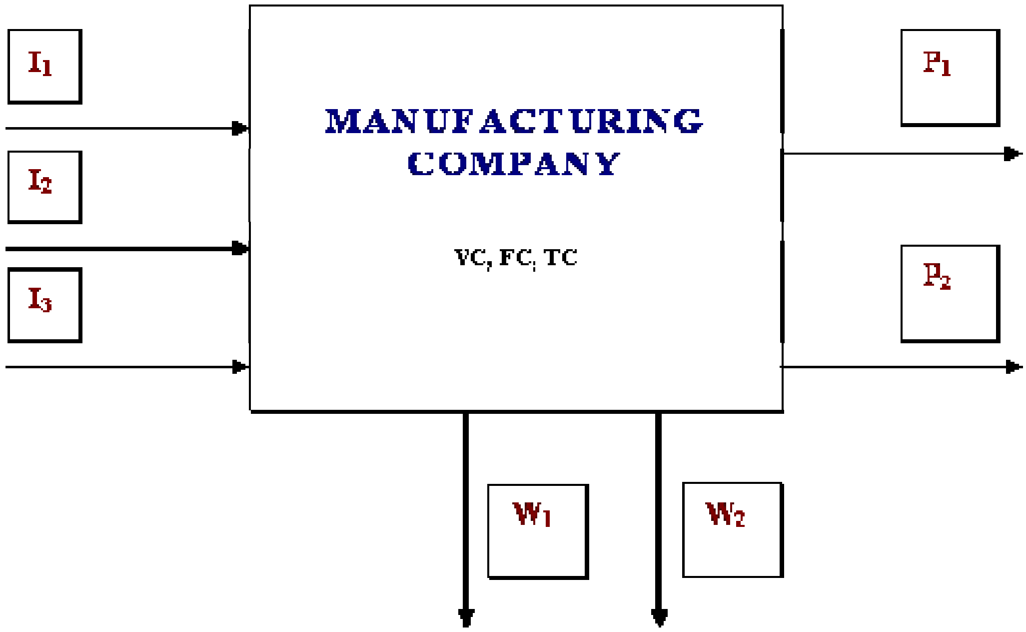

Figure 1. Exemplification of the application of the theory on the black box-unit [12]

The current waste fraction thus represents a new, substantially larger, cost, a so called *shadow cost* or *shadow price* which, if fully taken into account, imposes very strong financial incentives to drastically reduce the waste in question. Expression (1) is the mathematical model representation of the *equality principle* mainly expressing its financial implications and is termed *the model for Efficient Use of Resources for Optimal Production Economy* (EUROPE).

$$A / (B + C) \qquad (1)$$

where

A = quantity of the waste fraction in question produced
B = quantity of normal product output
C = sum of the quantities of the different waste fractions considered

One must define a suitable production or administrative unit to apply expression (1) on. This suitable unit can be everything from e.g. the entire company via divisions or profit centre workshops to an individual machine or any other level of the production system, depending on the circumstances.

2.2 Exemplification of the use of the scientific approach in practice

Figure 1 exemplifies the application of the theory on the black box-unit

where

I_1 = Input good No. 1. Unit: tonnes, litres, value etcetera.
I_2 = Input good No. 2
I_3 = Input good No. 3

P_1 = Product output good No. 1. Unit: tonnes, litres, value etcetera.
P_2 = Product output good No. 2

W_1 = Waste bad No. 1. Unit: tonnes, litres, value etcetera
W_2 = Waste bad No. 2

VC = Variable cost. Unit: $, €, SEK etc.
FC = Fixed cost
TC = Total cost = VC + FC

Definitions:

TP = Total product output goods = $\Sigma P_x = P_1 + P_2 + ,,, + P_n$ (2)
TW = Total waste output bads = $\Sigma W_x = W_1 + W_2 + ,,, + W_n$ (3)
TR_x = Total revenue from the waste fraction W_x

The EUROPE model gives PF = The Proportionality Factor

$PF_x = W_x / [TP + TW]$ (Compare expression (1) above.) (4)

The Environmental Adjustment Cost = EAC = (5)
= all costs connected with making the production processes
of a company environmentally friendly =
= net present value (NPV) of profit (loss) from the investment =
= present value of revenue – investment cost [12]

2.3 Schematic principles for allocation of shadow prices to waste fractions using different economic methods

2.3.1 Cost-Benefit Analysis

The method of overhead rates based on normal capacity

TR_x
- FC * PF_x
- VC * PF_x
= Amount to be allocated to the waste fraction W_x

TC/item = [Estimated VC / Calculated quantity of items] (6)
+ [Estimated FC / Normal quantity of items]

The average cost estimation method

TR_x
- FC * PF_x
- VC * PF_x
= Amount to be allocated to the waste fraction W_x

TC/item = [FC + VC] / TW (7)

It is to be noted that the possibilities for applying the equality principle on construction waste management are biggest for the average cost estimation method, [8], [9] and [11].

Example of a construction waste management application:

PF_x = [cost of waste fraction W_x /
/ (production cost + total waste management cost)] (8)

(Compare expression (4) above.)

Cost to be allocated to waste fraction W_x = total production cost * PF_x (9)

The equivalent method of cost estimation

TR_x
- FC * PF_x
- VC * PF_x
= Amount to be allocated to the waste fraction W_x

where

$PF_x = W_x / [(\Sigma (P_x * ER_x)) + TW]$ (10)

ER_x = the equivalent rate for a particular product P_x = (11)
= [Normal cost per unit for a given product] /
/ [Normal cost per unit for the product with the lowest cost per unit]

The absorption costing method

TR
- TC estimated using the absorption costing method * PF_x
= Amount to be allocated to the waste fraction W_x

The Activity-Based Costing (ABC) method

TR_x
- TC estimated using the ABC method * PF_x
= Amount to be allocated to the waste fraction W_x [7]

2.3.2 Contribution Margin Analysis

Income from sale of the fraction sold
- Variable cost of the fraction sold = VC * PF_x
= Contribution margin covering the fixed cost

Specific fixed cost of the fraction in question = FC * PF_x

= Contribution margin after deduction of costs traceable to the fraction =
= Operating income (or contribution margin)

Operating income per unit of waste fraction = Operating income / W_x = (12)
Amount to be allocated to each unit of the waste fraction (W_x)

In the case of n waste fractions, the total contribution margin can be calculated as follows:

$$CM_{tot} = \sum (CM_j\, x_j) \qquad (13)$$

where

CM_{tot} = total contribution margin of the n waste fractions
CM_j = Contribution margin per unit of waste fraction j
calculated using expression (1) involving shadow prices
x_j = The amount of tonnes, litres etc. of waste fraction j
$x_j \geq 0$, j = 1,2,3, …, n [10]

2.3.3 The Polluter-Pays Principle

Profit (loss) from the investment per unit of waste = [EAC * PF_x] / W_x = (14)
= Amount to be allocated to each unit of the waste fraction (W_x) [13]

2.3.4 Joint Production Theory

Joint production theory, which among other things concerns the optimal output proportions to aim at obtaining when desirable products and wastes are jointly produced in the same process, makes frequent use of the linear programming technique. Application of this method involves considering there to be different possible scenarios. A particular waste fraction is studied within a given production scenario, one which involves in part a set of different waste fractions with which various revenues and costs are associated.

The outcome of the profitability analysis for a given fraction guides the decision of whether the fraction in question is to be the object of separation. Such assessments are performed again for any further fraction within the set of fractions to be examined in terms of profitability. In any given scenario, therefore, a new assessment of profits and losses is required for each further fraction considered.

Note that since it is a question of making a choice between different products that can be produced, each product and the wastes related to it needs to be regarded as a unit. It is also assumed that the quantities of various wastes related to a particular product are known and are constant per unit of time. This allows the problem to be expressed in the following way, which is adapted to the production of waste:

Find the values of $x_1, x_2, \ldots, x_n$ that will

$$\text{maximize } z = (p_1' + p_1'')\, x_1 + (p_2' + p_2'')\, x_2 + \ldots + (p_j' + p_j'')\, x_j \qquad (15)$$

where

p_j' = profit (or loss, which produces a negative value) per unit of the product j or of the input
p_j'' = profit (or loss, which produces a negative value) from all the waste related to one unit of the product j or of the input j,

subject to the following constraints:

$$a_{11}x_1 + a_{12}x_2 + \ldots + a_{1n}x_n < b_1 \qquad (16)$$

$a_{21}x_1 + a_{22}x_2 + \ldots + a_{2n}x_n < b_2$

$a_{m1}x_1 + a_{m2}x_2 + \ldots + a_{mn}x_n < b_m$

where

$x_j \geq 0, j = 1, 2, \ldots, n.$ (17)
a_{mn} are constant coefficients of production

This allows the most profitable product mix of the n products and the related wastes to be calculated, and also the total profit to be estimated, by multiplying the calculated profit-maximizing amounts of the product by the marginal contribution of each of the n products and the related wastes. Mathematically, this can be expressed as follows:

Find the values of $x_1, x_2, \ldots, x_n$ that will

maximize $CM_{Tot} = (CM_1' + CM_1'')\, x_1 + (CM_2' + CM_2'')\, x_2 + \ldots + (CM_j' + CM_j'')\, xj$ (18)

where

CM_{Tot} = total marginal contribution of the product and waste mix
CM_j' = marginal contribution per unit of the product j or of the input j
CM_j'' = marginal contribution from all the waste related to one unit of the product j or of the input j,

subject to the following constraints:

$a_{11}\, x_1 + a_{12}\, x_2 + \ldots + a_{1n}\, x_n < b_1$ (19)
$a_{21}\, x_1 + a_{22}\, x_2 + \ldots + a_{2n}\, x_n < b_2$

$a_{m1}x_1 + a_{m2}x_2 + \ldots + a_{mn}x_n < b_m$

where

$x_j \geq 0, j = 1, 2, 3, \ldots, n.$ (20)
a_{mn} are constant coefficients of production

What is new in this approach is the maximization stipulated in (15) and (18), which reflects the assumption of a product and the related wastes representing a unit, [5] and [6].

3 MODEL APPLICATIONS

The theory described in chapter 2 has successfully been applied to, or will be applied to, for example, the following areas: (i) mechanical work-shops [5], [7], [10] and [13]; (ii) bulk industry [1], [5], [6], [7], [10] and [13]; (iii) joint production [5]; (iv) construction [8], [9] and [11]; (v) ore mining [14]; (vi) landfill mining [18]; (vii) recycling and recirculation [16]; (viii) energy spillage [15], (ix) landfilling [2], [3] and [4], and; (x) baling plants [2], [3], [4], [20] and [21].

4 PROSPECTS FOR THE FUTURE

Since the Stone Age, people have engaged in mining activities and exploited non-renewable resources. As a result, the lives of people have become nicer, easier and more secure.
During the past century, the consumption pace has been higher than during all earlier centuries together. Also, the pace constantly accelerates. Today, the global production and consumption of most metal and mineral commodities is higher than ever. The development is explosive [14].
Therefore, the objective is to provide practically useful methods that managers on the UN and OECD levels etcetera can use to obtain guidelines for how to increase the cost efficiency of different activities related to natural resources in general by the employment of the EUROPE model that is based on the *equality principle*. In doing so, the equity of the distribution of natural resources and commodities is intended to be promoted based on the optimization of the production of residuals when such facilities are exploited. Below, the theory is briefly outlined for how to apply the *equality principle* in such novel ways.

4.1 Optimization of the usage of energy resources

During the last two decades, recycling of energy has been the object of substantial attention. Also, most citizens contribute today to make the recirculation of the energy they use work properly. In spite of these efforts, still the anthropogenic cycles leak energy. These leakages may also affect the environment in a negative way, except from the economic losses caused by the leaking cycles.
Therefore, the objective is to provide practically useful basic methods that managers on multiple levels can use to obtain guidelines for how to increase the cost efficiency of energy-utilisation activities and similar schemes by the employment of the EUROPE model based on the *equality principle*. In doing so, the equity of the distribution of energy resources and commodities related to the usage of energy is intended to be promoted based on optimization of the occurrence of spillages when such facilities are exploited.
The practical application of the proposed theory in the corporate, municipal, national and international context is emphasized. The methodology is useful for increasing the cost-effectiveness of mainly the occurring spillage in the usage of energy resources and commodities in general. Also, the equity of the distribution of such facilities is improved. The developed methods will be suitable information support tools for decision-making in the management of energy resources, with emphasis on the economy of spillage [15].

4.2 Optimization of monetary flows

Voluntarily but ”punishing” shadow costs are imposed upon the unwanted fees on monetary transactions by the employment of the EUROPE model based on the *equality principle* to increase the economic incentive to optimize the financial transaction in question. Wherever plausible, additional monetary units (MU) are hence added to the relevant profit and loss accounts and balance sheets plus the current budgets and forecasts etcetera. Thereby, the CEO gets a strong economic incentive to start chasing the surplus transaction fees that now has increased due to the fictive and purely internal *shadow costs* being allocated to them. Thus, the transaction fees in total will decrease and the business in question will increase its profit and efficiency and the company’s customers obtain more value for their money.
The theory will be applied not only on single corporations, but also on transaction costs within nations, continents and all over the globe in order to improve, monitor and evaluate monetary flows. Possible users are, for example, the World Bank, the International Monetary Fund (IMF) and central banks [19].

4.3 Optimization of the global distribution and equity of natural resources

A cost structure is proposed for improving the resource economy of metal and mineral commodities based on the optimization of residuals of resource exploitation on a global scale. The methodology proposed involves business administration and economics theory and employs the *equality principle* and the EUROPE model. The practical application of the proposed theory will be studied in the Swedish context. The methodology is useful for increasing the cost-effectiveness of mainly the occurring residuals in the exploitation of natural resources and commodities in general. Also, the equity of the distribution of such facilities is improved on a global scale. The developed methods will be suitable information support tools for decision-making in the management of natural resources, with emphasis on the economy of residuals at primarily the global level with a longer time-perspective.
The study will provide practically useful basic methods that managers on the UN and OECD levels etcetera can use to obtain guidelines for how to increase the cost efficiency of mining activities and other similar schemes by the employment of the EUROPE model based on the *equality principle*. In doing so, the equity of the distribution of natural resources and commodities is intended to be promoted most for those actors that need it most based on the economic optimization of the production of residuals when such facilities are exploited [17].

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