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Bauhaus-Institut für zukunftsweisende Infrastruktursysteme (b.is)

Das Bauhaus-Institut für zukunftsweisende Infrastruktursysteme (b.is) verfolgt das Ziel, die Kooperation der derzeit beteiligten Professuren Siedlungswasserwirtschaft, Biotechnologie in der Ressourcenwirtschaft und Urban Energy Systems zu intensivieren sowie die Honorarprofessur Urbanes Infrastrukturmanagement, um Lehr-, Forschungs- und Beratungssaufgaben auszubauen. So werden beispielsweise die Weiterentwicklung von Studiengängen, gemeinsame Doktorandenkolloquien oder gemeinsame Forschungs- und Entwicklungsaufgaben durchgeführt.

Das b.is will sich deutlich sichtbar im Bereich der Infrastrukturforschung aufstellen. Die Forschung und Lehre in diesem Bereich orientiert sich am medienübergreifenden Modell der nachhaltigen Gestaltung von Stoff- und Energieflüssen sowie ressourcenökonomisch ausgerichteten Systemen, die verbindendes Konzept der Kernprofessuren des Instituts sind. Die Professur Betriebswirtschaftslehre im Bauwesen ist mit dem b.is assoziiert.

Bauhaus-Institute for Infrastructure Solutions (b.is)

The Bauhaus-Institute for Infrastructure Solutions (b.is) aims to strengthen the cooperation of the university´s research teams in Urban Water Management and Sanitation, Biotechnology in Resources Management and Urban Energy Systems in the areas of teaching, research and consultancy work. This encompasses the further development of degree programmes, joint doctorate colloquia and joint research and development activities.

Currently the chair of urban water management and sanitation, the chair of biotechnology in resources management and the chair of urban energy systems as well as the honorary professorship for urban infrastructure management are members of the institute. The chair of construction economics is associated with the institute.

The b.is will increase its visibility in infrastructure research. Education and research are geared to the comprehensive model of sustainable material and energy flows and resource economy oriented systems, which are the linkage of the institute's chairs.

Development of Soil-Willow-System for wastewater treatment and wood production under the extreme climate conditions of Mongolia

Dissertation zur Erlangung des akademischen Grades

Doktor-Ingenieur (Dr.-Ing.)

An der Fakultät Bauingenieurwesen der Bauhaus-Universität Weimar

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List of abbreviations

BOD_5	Five Day Biochemical Oxygen Demand
CFU	Colony Forming Unit
COD	Chemical Oxygen Demand
DBH	Diameter at Breast Height
DM	Dry matter
DNRA	Dissimilatory Nitrate Reduction to Ammonium
DO	Dissolved Oxygen
EC	Electric Conductivity
ECe	Electric conductivity of soil paste
EDTA	Ethylenediaminetetraacetic acid
ET	Evapotranspiration
FC	*Fecal Coliforms*
FC	Field Capacity
HLR	Hydraulic Loading Rate
LF	Leaching Fraction
LRB	Langenreichenbach
MD/D	Multiple Dose per Day
MNS	Mongolian National Standard
MoMo	Model region Mongolia
MPN	Most Probable Number
MUST	Mongolian University of Science and Technology
NH_3	Ammonia
NH_4-N	Ammonium Nitrogen
NO_2-N	Nitrite Nitrogen
NO_3-N	Nitrate Nitrogen
OF	Overland Flow System
Org.-N	Organic Nitrogen
ORP	Oxidation/ Reduction Potential
O&M	Operation and Maintenance
PE	Population Equivalent
PLC	Programmable Logic Controller
PO_4-P	Phosphate Phosphorus
PSD	Particle Size Distribution
PVC	Polyvinylchloride
PWP	Permanent Wilting Point
RI	Rapid Infiltration Land Treatment
RQ	Research Question
SAR	Sodium Adsorption Ratio
SAT	Soil-Aquifer Treatment
SOM	Soil Organic Matter
SD/D	Single Dose per Day
SD/W	Single Dose per Week
SR	Slow Rate Land Treatment
TC	Total Coliforms

TDS	Total Dissolved Solids
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total phosphorus
TSS	Total Suspended Solids
UFZ	Helmholtz Environmental Research Centre
USAG	Water Supply and Sewerage Company Darkhan
UV	Ultra Violet
VDS	Volatile Dissolved Solids
WUE	Water Use Efficiency
WWTP	Wastewater Treatment Plant
XRFA	X Ray Fluorescence Analysis

1. Introduction

1.1. Statement of problem

The existing and already altering wastewater treatment plants in Mongolia are facing a number of challenges due to a combination of environmental, technical, and financial factors.
The high continental climate of Mongolia is characterized by long cold and dry winters followed by warm but short summers (MoMo-I, 2009). The annual mean temperature in Drakhan city in Northern Mongolia is - 0.6°C, with minimum and maximum temperatures reaching -40°C (January) and +30°C (July), respectively (Climate, 2016). Six months per year (from October to March) are showing temperature under 0°C, making it a challenge to operate and maintain water and wastewater infrastructures (Figure 1.1). These cold climatic conditions impact heavily the water and wastewater infrastructure. For instance, the sewer network has to be buried at a depth of 3.5 m to 4.5 m to avoid sewage freezing that could break the pipes. Additionally, wastewater treatment plants including a biological treatment stage have to be housed or insulated to keep the temperature high enough for the biological processes to occur. Therefore, high investment costs are often required for implementing new infrastructures and renovating the existing ones.

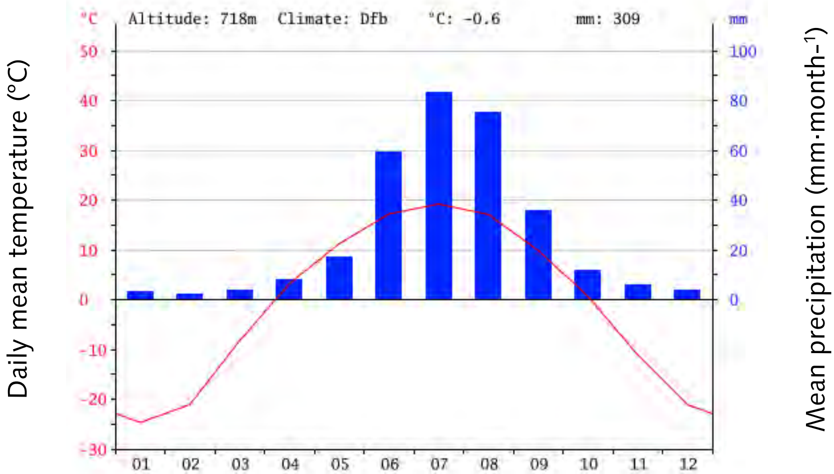

Figure 1.1: Climatic condition of City of Darkhan (Source: Climate (2016))

About 90% of the wastewater treatment plants WWTPs were built between 1970 and 1990 and have not been upgraded since, due to a lack of investment (Figure 1.2). According to Dolgorsuren *et al.* (2012)

out of a total of 115 wastewater treatment plants countrywide, 41 are out of service, 38 are working in poor condition, 36 are working normally in summer and all of them are under notable pressure (poor treatment) during winter. Furthermore, the WWTPs were designed to treat only organic pollutions but not nutrients and pathogens (Dolgorsuren *et al.*, 2012).

a) A WWTP out of service in Orkhon *b) A WWTP in critical condition in Sharyn Gol*

Figure 1.2: Current situation of WWTPs in Orkhan and Sharyn Gol (Photos: G. Kurelbaatar, 2013)

Countrywide, a total of 64 treatment plants were designed to discharge treated effluent into infiltration basins. Another 51 treatment plants have been designed to discharge treated effluent into rivers (Dolgorsuren *et al.*, 2012). Both these options are posing the risk of groundwater contamination and eutrophication of water bodies (Figure 1.3). In addition, valuable resources in the form of nutrients and water are being disposed of instead of being reused.

This combination of environmental and technical problems demands a robust and affordable wastewater treatment technology, which is compatible with the climate conditions.

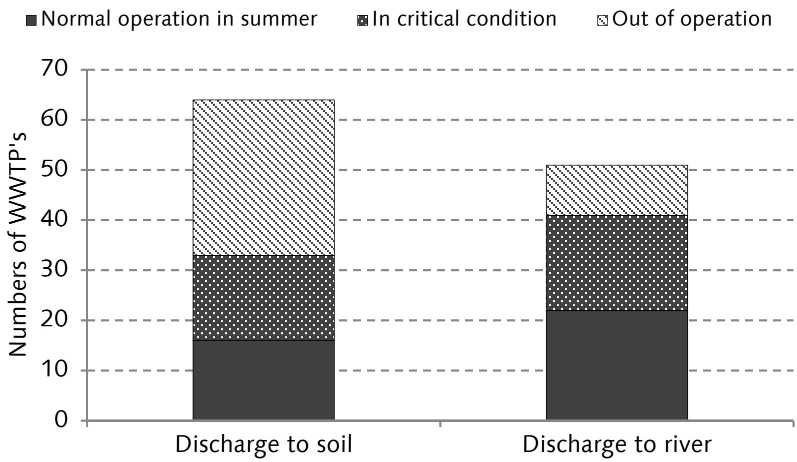

Figure 1.3: Discharge situation of WWTPs in Mongolia (according to Dolgorsuren et al., 2012)

1.2. Objectives

In frame of Integrated Water Resource Management Project in Central Asia (MoMo-II) the Helmholtz Centre for Environmental Research-UFZ proposed an approach "Wastewater treatment plant with Integrated Wood production" as contribution for improvement of poor sanitary problems in Mongolia. This technology involves application of wastewater after primary treatment onto fields planted with fast growing trees. Due to its simple system components, this technology is known for its robust and reliable treatment, under proper management (Paranychianakis *et al.,* 2006). Therefore, this might be an attractive solution to address the sanitary problems in Mongolia. In addition, fast growing trees such as willow and poplar will provide wood production, which will make a valuable firewood resource in the region. This additional resource would ease the ongoing deforestation, especially along the riparian zones (Tsogtbaatar, 2004). Furthermore, by using the wastewater as irrigation for the trees, this technology will contribute reducing the excessive use of groundwater in agriculture (UN-Water, 2013).

While a number of studies have been conducted focusing on land application and short rotation willow coppice for wastewater treatment in North America, Scandinavia, and China (US-EPA, 1987; Börjesson and Berndes, 2006; Ou *et al.,* 1997), no relevant studies have been conducted in Mongolia. Especially, studies dealing with the use of primary treated wastewater are low in number when compared to studies that used secondary or tertiary treated effluent as irrigation for willow and poplar trees.

The main objective of this work is to assess the suitability of a treatment system based on land application of primary treated wastewater combined with short rotation coppice (Soil-Willow-System) under extreme climatic conditions. For this purpose, five research questions have been asked to gain the understanding of the treatment process involved in the technology:

> **RQ-1**: What influences do the trees have on the water and nutrient mass balance?
>
> **RQ-2**: What influences do different hydraulic loading rates (HLR) have on water and nutrient mass balance?
>
> **RQ-3**: What influences do the variations in seasonal loading regime have on water and nutrient mass balance?
>
> **RQ-4**: What influences do the variations in weekly loading regime have on water and nutrients mass balance?
>
> **RQ-5**: What influences do the variations in daily loading regime have on water and nutrient mass balance?

For answering the questions experiments on water, soil, and plants were conducted at two research sites:

1. A pilot plant was established at the Mongolian University of Science and Technology (MUST) in Darkhan. Focusing on the first three research questions (RQ-1, RQ-2, and RQ-3), four different wastewater treatment options were tested over two years.
2. A second pilot plant was established at Langenreichenbach (LRB), Germany. Three different loading patterns were tested on two treatment beds for two consecutive years and compared for answering the RQ-4 and RQ-5.

Due to the specific characteristic of the Soil-Willow-System, mass balance and mass removal approaches are used in order to evaluate the different systems.

The mass balance data from the individual four beds at the pilot plant in Darkhan are compared on an **annual** basis in order to answer the RQ-1, RQ-2, and RQ-3. The mass balance data from the two beds at LRB over two consecutive years are compared on a **seasonal** basis in order to answer the RQ-4 and RQ-5.

The key findings of the present study are outlined as the main outcome of the work. Based on the results obtained, a design recommendation for Soil-Willow-System with three different options is proposed for the town of Khongor as an example for the regional conditions.

2. Literature review

2.1. Definition and categories

2.1.1. Land treatment

According to Crites and Tchobanoglous (1998), land treatment is the controlled application of wastewater onto the land surface to achieve a designed degree of treatment through natural, physical, chemical, and biological processes within the plant soil water matrix. Although trees are often not the preferred vegetation cover used for land treatment systems, there are a variety of examples of their successful applications (Pedrero and Alarcon, 2009; Tzanakakis *et al.,* 2009). According to Crites *et al.* (2006), there are three types of land treatment systems; Slow Rate (SR), Soil aquifer treatment (SAT) or Rapid Infiltration (RI), and Overland Flow (OF) systems. While OF systems are relied on the utilization of soil surface and the plants on it, SR and SAT systems share similarities by relying on the natural physical, chemical, and biological mechanisms within the soil-plant-water matrix along the soil profile (Crites *et al.,* 2014). The main difference between the two systems is the hydraulic loading rate. According to Crites and Tchobanoglous (1998) the annual hydraulic loading rate for SR systems range between 610 mm and 5490 mm, while it ranges between 5500 mm and 100000 mm for SAT systems.

Depending on the objective, SR systems can fall into two main types (Paranychianakis *et al.,* 2006):

- SR-Type 1-Systems have the main objective of wastewater reuse by relying on the crop water use efficiency (WUE). Hydraulic loads applied to those systems usually do not exceed the crop WUE in order to maximize the reuse efficiency.
- SR-Type 2-Systems are mainly used for disposal purposes rather than for reuse purposes such as for constant loading of wastewater onto the land despite having high percolation during non-growing season. Although plant uptake of nutrients contributes to the reduction of pollutants, the main removal mechanisms rely on the soil matrix.

2.1.2. Short rotation coppice and short rotation forestry for wastewater treatment

According to Hardcastle *et al.* (2006) the main differences between short rotation forestry and short rotation coppice are the rotation period and the desired end production. The goal of short rotation forestry with a rotation period of 8-20 years is to harvest timber or stem wood. The short rotation forestry uses willow, poplar, birch, and maple trees but the short rotation coppice only prefers willow and poplar trees, due to their fast growing ability after being coppiced. The coppice rotation period is 3-5 years and the goal is to produce biomass for fuel by harvesting everything above ground (coppicing).

Short rotation forestry / coppice for wastewater treatment belongs to slow rate (SR) systems. Rosenquist *et al.* (1997) suggested using short rotation willow coppice for wastewater treatment in Sweden during summer, to avoid the percolation of excess pollutants during winter, when the plants are inactive. The systems operated only during summer belong to SR system Type 1, while there are a number of examples of Type 2 – systems where it was operated all year around (Mirck *et al.,* 2005).

2.2. Historical background

Since the beginning of civilization, land disposal and land application have been the primary ways of human solid and liquid waste management (Ikehata and Liu, 2011). Evidences have been found confirming the existence of sewerage systems, public sanitation, and indicate that wastewater was applied in the irrigation of agricultural land during the ancient Greek civilizations approximately 4000 years ago (Tzanakakis *et al.,* 2014). Since then, there is insufficient data to prove the existence of wastewater application for agricultural irrigation until 14th century (US-EPA, 1979). The earliest documented use of wastewater effluent for crops irrigation was in Bunzlau, Germany in 1531 (Gerhard, 1909). This was followed by the "Crargentinny Meadows" project in Edinburgh, Scotland around 1650, which stimulated the interest of many observers as it clearly demonstrated the beneficial effects of wastewater application on crop yield (Stanbridge, 1977). The development and implementation of land treatment took place in the US during the mid-19th century and the concept was adapted and implemented under different terminology "Sewage farming" (US-EPA, 1979; Reed *et al.,* 1995). The implementation of land treatment declined during the 20th century due to development of mechanical wastewater treatment plants (Tzanakakis *et al.,* 2006).

A renewed interest in the implementation of land treatment system was observed in the United States (U.S.) after the commencement of Clean Water Act (CWA) in 1972 (Reed *et al.,* 1995). The requirement proposed by the CWA could be met by using land treatment systems. Braatz and Kandiah (2002) pointed out that even though irrigated agriculture had received far more attention than irrigated forestry, health hazards and cultural and aesthetic values, however, limited the agricultural use of wastewater. Additionally, under specific conditions irrigated forestry might be economically competitive with irrigated agriculture. Proving this argument in the 1970´s in Europe, the Short Rotation Willow Coppice, which shares the same concept with land treatment was receiving increased attention due to the energy crisis (Vande Walle *et al.,* 2007) even though the research for willow production actually started in the late 1960's due to a predicted shortage of raw material for the pulp and paper industry (Mirck *et al.,* 2005). Experiments with wastewater irrigation of willow had been undertaken during the 1960's in Poland (Perttu and Kowalik, 1997).

In the 1980´s, development programs were launched for commercial short rotation forestry systems in the U.S. (Perlack *et al.,* 1986), the United Kingdom (U.K.) (Mitchell *et al.,* 1999), Western Europe (Vande Walle *et al.,* 2007), and Scandinavian countries (Dimitrou and Aronsson, 2011). As the result commercial short rotation forestry systems were created, implying that the systems were mainly for biomass production and partly received irrigation, fertilization, and weeding control instead of wastewater.

In the last three decades, the main focus of short rotation forestry for biomass production has moved towards environmental applications such as utilization of short rotation forestry for treatment of municipal wastewater and landfill leachate (Mirck *et al.,* 2005). Sweden as one of the leading countries, implemented short rotation willow coppice as a polishing stage for wastewater treatment plants in five municipalities for additional nutrient removal (tertiary treatment) and biomass production (Börjesson and Berndes, 2006). Around 80% - 90% of the total short rotation forestry area in Sweden receives either secondary effluent or sludge (Börjesson and Berndes, 2006). In this instance, the residues such as municipal wastewater and sludge are considered more as resources than wastes. The harvested biomass is utilized by combined heating and power generation plants in the form of undried chips (Dimitrou and Aronsson, 2011).

2.3. Removal processes

When applied to land at a slow rate, the wastewater is purified through physical, chemical, and biological processes that occur in the soil-water-plant matrix (Paranychianakis *et al.,* 2006). The removal mechanism involves processes such as filtration, transformation, degradation, predation, natural die-off, soil adsorption, chemical precipitation, volatilization, denitrification, and plant uptake (Paranychianakis *et al.,* 2006; US-EPA, 2006).

2.3.1. Organic matter removal

Organic matter is present in wastewater in form of particulate matter, colloidal and dissolved carbon (Londong *et al.,* 2006). Physicochemical processes such as straining, sedimentation, interception, and adsorption are responsible for initial removal of organic matter right after the application. The biological degradation and oxidation mainly contribute to the removal of organic matter thereafter (Paranychianakis *et al.,* 2006). Generally, soil microbial biomass and the activity of enzymes involved in carbon and nutrient cycling increase as result of wastewater application due to the enrichment of the soil with energy substrates and nutrients (Paranychianakis *et al.,* 2006). The majority of the organic matter is mineralized in the top layer of the soil. Tzanakakis (2003) reported over 90% removal of COD in the first 15 cm of the soil, followed by no further removal in deeper soil layers. Similar observations were reported by Zaman *et al.* (2002), who found the main increase of microbial biomass that is responsible for organic carbon and nutrient mineralization occurring in the top 10 cm of soil layer.

Generally, the biodegradable organics are removed at great quantities when wastewater is applied onto soil. Application rates of up to 500 kg $BOD_5 \cdot ha^{-1} \cdot d^{-1}$ have been reported which did not limit the high removal efficiency of these types of systems (Paranychianakis *et al.,* 2006; Reed *et al.,* 1995).

2.3.2. Nutrient removal

2.3.2.1. Nitrogen removal

Total Nitrogen (TN) in primary treated wastewater mainly consists of ammonium nitrogen (NH_4-N) and organic nitrogen (Org.-N). The nitrogen removal process is very complex and requires both aerobic and anoxic (anaerobic) conditions. The removal mechanism involves processes such as ammonification, nitrification, denitrification, plant uptake, and physicochemical routes such as sedimentation, ammonia volatilization, and ion exchange (Kadlec, 1999). Org.-N is mineralized into ammonium under both aerobic and anaerobic conditions and the process is very fast especially under aerobic conditions due to the involvement of wide range of heterotrophic organisms including various fungi and bacteria (Powlson and Addiscott, 2005). The rate of mineralization in soil depends on a wide range of factors. C:N ratio is often used to predict the mineralization and immobilization rate of nitrogen in soil (Bengtsson *et al.,* 2003). While mineralization occurs when C:N ratio in the soil is below 30, immobilization tends to occur at higher C:N ratio (Horn *et al.,* 2010).

Volatilization is a minor process that partly contributes to nitrogen removal. This tends to happen when the soil, receiving wastewater application, has high (over 7.8) pH value and low absorption of ammonium (low cation exchange capacity) (Gerba, 2005). Additionally, the volatilization rate depends on parameters such as wastewater application methods (i.e sprinkler, drip, or furrow irrigation methods) (Al-Kaisi and Waskom, 2002), wind speed, and temperature (Sharpe and Harper, 2002).

Plant uptake is one of the major processes responsible for nitrogen removal in Soil-Willow-System. Plants utilize nitrogen in forms of both ammonium and nitrate; however, nitrate is more preferred form for plants (Larcher, 1994; Powlson and Addiscott, 2005). The nitrogen is then removed from the system via harvest of above ground biomass. The removal rate can vary depending on the climate, operations of the system, and the type of soil. Although the data reported for N removal through biomass harvest vary depending on the biomass yields, the N accumulated per t of dry biomass is reported to lie between 6-8 kg per t of DM for willow and poplar trees (Christersson, 1986; Labrecque and Teodorescu, 2003; Von Fircks *et al.,* 2001). In some cases, much lower N content (4-5 kg per t of DM) was recorded (Adegbidi *et al.,* 2001). In some short rotation coppice systems (mostly conventional and/or type-2 land treatment sites), the annual N input is about 150-180 kg $TN \cdot ha^{-1} \cdot a^{-1}$ in order avoid N leaching to the groundwater (US-EPA, 2006). This number, based on a proxy estimate, indicates that the biomass yield in conventional short rotation coppice system would be around 20 DM $t \cdot ha^{-1} \cdot a^{-1}$. However, there are many short rotation coppice systems which receive comparably higher N load (Dimitrou and Aronsson, 2011; US-EPA, 2006).

Nitrification and denitrification is the main TN removal process within the soil-water-plant matrix. Under aerobic conditions, the oxidation of ammonium (NH_4^+) to nitrite (NO_2^-) takes place mediated by autotrophic bacteria including widely studied *Nitrosomanas* and less-studied *Nitrosolobus, Nitorsospira, and Nitrosococcus*. Nitrite usually does not accumulate in soil, as it is rapidly oxidized in aerobic soil to nitrate by *Nitrobacter* species (Powlson and Addiscott, 2005). Under anaerobic conditions, the nitrate is converted by denitrifying bacteria. When oxygen is in short supply, nitrate ion can be used instead of O_2 during respiration. As the result of this process, a mixture of nitrous oxide (N_2O) and nitrogen gas (N_2) is produced. The mixture proportion of N_2O and N_2 depends on the soil temperature and pH. At a temperature of 25° C or more and a pH greater than 6, most of the N_2O is reduced to N_2. However, when the soil temperature is low and the pH is below 5, the

$N_2O:N_2$ ratio can be 1 (Powlson and Addiscott, 2005). Aditionally, denitrification is regulated by carbon source, since the heterotroph denitrifiers require reduced C as the electron donor (Robertson and Groffman, 2007).

In general, denitrification and nitrification are complex processes that depend on many factors such as temperature, pH, soil moisture content and C:N ratio (Robertson and Groffman, 2007). In case of Soil-Willow-System, the soil moisture content is the most important parameter regulating the nitrogen transformations. When moisture and temperatures are favorable, organic matter inputs via wastewater application lead to high rates of mineralization. Nevertheless, in water logged or cold soils, moisture, and temperature can limit microbial activity and inhibit the mineralization, nitrification, and denitrification rates (Robertson and Groffman, 2007). The relationship between the relative amount of microbial activity and the soil moisture content described by Linn and Doran (1984) is shown in Figure 2.1.

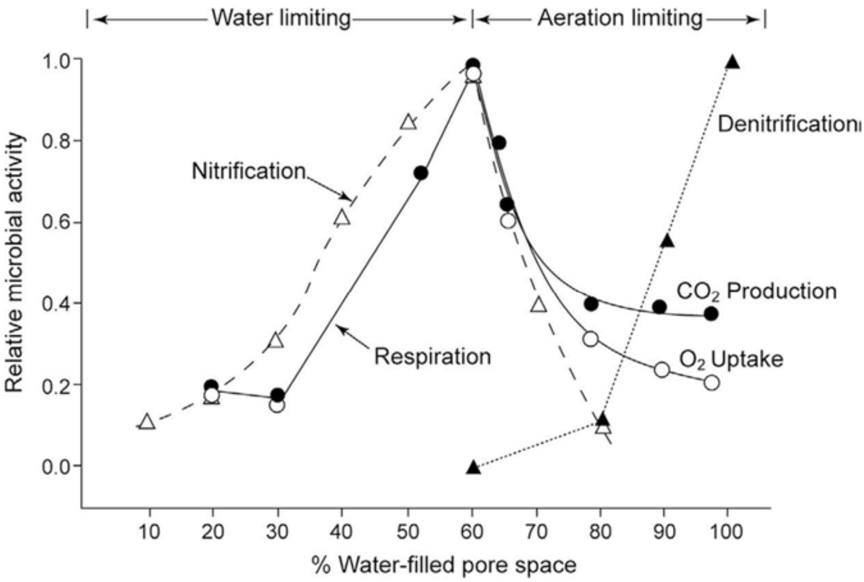

Figure 2.1: The relationship between relative microbial activity and soil moisture content (displayed as water-filled pore space) described by Linn and Doran (1984) and modified by Robertson and Groffman (2007)

Water filled pore space can be estimated once the soil moisture content, bulk density and absolute density is known (Robertson and Groffman, 2007).

There are several other transformation mechnisms of nitrogen in soil, which include *Dissimilatory nitrate reduction to ammonium* (DNRA), *non-respiratory denitrification*, *chemodenitrification,* and anammox processes. However, none of these microbial processes are thought to be as quantitatively important as the other processes described above (Robertson and Groffman, 2007).

In case of Soil-Willow-System, the combination of all of these processes including the N removal via biomass harvest and the TN retention soil is considered as the TN removal. The TN mass removal rates vary depending on a range of factors such as type of wastewater, loading rate, climate, vegetation, and the soil type. A removal rate of 875 kg $TN \cdot ha^{-1} \cdot a^{-1}$ was reported by Rastas *et al.* (2012) for willow planted filter bed system in cold region of Sweden, while considerably higher removal rate was reported by Tzanakakis *et al.* (2009), who found that slow rate land treatment with poplar was able to remove 1100 kg $TN \cdot ha^{-1} \cdot a^{-1}$. The later result was also confirmed by Aronsson *et al.* (2010), who noted a removal rate of 1220 kg $TN \cdot ha^{-1} \cdot a^{-1}$ for willow coppice system in central Sweden.

2.3.2.2. Phosphorus removal

Total phosphorus in wastewater is found in dissolved forms of inorganic ortho- (o-PO_4) and poly-phosphates (poly-PO_4), and organic and particulate phosphates (Londong *et al.,* 2006). Phosphorus is mainly retained in the soil. This retention mechanism is referred to as removal for land treatment systems. The phosphorus removal can occur through plant uptake, biological, chemical, and physical processes. Unlike nitrogen removal, the phosphorus removal in soil heavily depends on chemical reactions, which are slowly renewable. Therefore, the retention capacity will be reduced over time, but not exhausted (US-EPA, 2006). A rough estimation by US-EPA (1981) suggested that 30 cm deep soil is P saturated within 10 years. This suggestion is supported by the estimation of Ryden and Pratt (1980), who used the theoretical model of soil P saturation developed by Shah *et al.* (1975). According to this estimation, given that the TP load is 150 $kg \cdot ha^{-1} \cdot a^{-1}$, a 2 m deep soil layer will be saturated within 40 years.

Plant uptake contributes to phosphorus removal at a lower degree, resulting in soil being relied upon as the major removal media (US-EPA, 1981). According to US-EPA (2006) typical phosphorus concentration is 0.2% - 0.4% in the harvested dry biomass. However, the phosphorus concentration of poplar and willow trees reported by Paranychianakis *et al.* (2006) were relatively low ranging between 0.07% and 0.1%.

Biological assimilation of phosphorus occurs at lower extent (Kaila, 1949). Nevertheless, this process is considered as a temporal immobilization rather than a retention in soil, since the phosphorus assimilated in microbial biomass will be mineralized again as the result of degradation over time (Oehl *et al.,* 2001).

Sorption is one of the main removal mechanisms of phosphorus during wastewater application onto land (Paranychianakis *et al.,* 2006) The removal process is dependent on various soil properties. Soils with higher clay content have greater sorption capacity compared to sandy soils, due to their higher reactive surface area (Agbenin and Tiessen, 1995). High organic matter content also improves the sorption capacity as it provides additional contact surfaces (McGechan and Lewis, 2002).

The sorption process usually occurs rapidly and is followed by chemical precipitation process (Reed *et al.,* 1995) and phosphorus diffusion on poorly accessible sites (Paranychianakis *et al.,* 2006). Phosphorus compounds precipitate with Fe^{3+}, Al^{3+} and Ca^{2+} forming mainly complex phosphates of Fe and Al in neutral and acidic soils and calcium phosphate in calcareous soil (Ryden and Pratt, 1980). However, the kinetics of the precipitation reactions are relatively slow and should not be overestimated (Bouwer and Chaney, 1974). The phosphorus retention in soil as the result of these processes highly depends on the contact time (Ryden and Pratt, 1980) and temperature (McGechan and Lewis, 2002). In addition, soil pH and redox potential has significant impact on phosphorus retention. Under reducing conditions, phosphorus is released and becomes soluble. As soil pH decreases, phosphorus will be replaced by sulfate posing a risk of leaching downwards (Savant and Ellis, 1964). On the other hand, depending on the soil temperature crystallization of phosphorus might occur regenerating the phosphorus adsorption capacity of the soil (Ellis (1973) sited by Bouwer and Chaney, (1974)). Overall, sorption appears to be the dominant removal mechanism for phosphorus when wastewater comes into contact with soil while precipitation is very likely to occur only in soils containing high amount of Ca (Ryden and Pratt, 1980).

Generally, when wastewater is applied onto land, the removal rate for phosphorus can be very high. A wide range of phosphorus removal rate for short rotation coppice for wastewater treatment has been recorded. Tzanakakis *et al.* (2009) reported 220 kg TP $\cdot ha^{-1} \cdot a^{-1}$ for poplar planted slow rate system in Greece, which was irrigated only in summer. In northern Sweden, a willow planted filter bed system showed TP removal rate of 72 kg TP $\cdot ha^{-1} \cdot a^{-1}$ (Rastas *et al.,* 2012). Similar results were reported by Jonsson *et al.* (2004), who found that the TP removal rate of a willow planted soil filter system in southern Sweden was 65 kg TP $\cdot ha^{-1} \cdot a^{-1}$.

2.3.3. Pathogen removal

The known pathogens of concern in Soil-Willow-Systems are parasites, bacteria, and viruses. The removal of pathogens in soil-water media is accomplished by filtration, adsorption, desiccation, predation, and decay due to exposure to sunlight (US-EPA, 2006). The dominant removal processes for bacteria is filtration and sedimentation while adsorption is the main removal mechanism for viruses (Gilbert *et al.,* 1976). These removal mechanisms highly depend on the soil properties (clay content), pH, hydraulic conditions (different loading patterns), the application method (surface or subsurface), and the climatic conditions (Paranychianakis *et al.,* 2006). As an indicator for presence of pathogenic organisms, fecal coliforms (FC) (US-EPA, 2006) and/or *E. coli* (Salgot *et al.,* 2006) are often used.

The pathogens are entrapped within soil matrix due to the attachment to the available sorption sites. Therefore, fine-textured soils (for example clayey soils) are known to be the most effective in pathogen removal whereas coarse-textured soils allow downward microbial movement (Huysman and Verstraete, 1993). Despite the higher efficiency in pathogen removal, clayey soils have higher risk of preferential flow due to surface cracks when compared to that of sandy or loamy soils (Paranychianakis *et al.,* 2006).

Goyal and Gerba (1979) suggested that the sorption process of bacteria within soil matrix depends on the soil pH implying greater pathogen entrapment with decreasing soil pH. Extreme high or low pH conditions may enhance the pathogen removal since bacteria favor neutral conditions (Frankenberger, 1985).

Soil organic matter and biofilm have positive effects on pathogen removal due to its provision of additional sorption sites, reducing the medium porosity allowing better entrapment (Medema *et al.,* 1998). However, Yates and Gerba (1998) noted that the soil organic matter extends the survival time for entrapped pathogens. Mubiru et al. (2000) reported similar findings. Soil organic matter might have an indirect effect on pathogen survival according to Paranychianakis *et al.* (2006), by increasing the water holding capacity of the soil and thus creating more favorable condition for the pathogens.

Fate of pathogens is also affected by the hydraulic condition of the soil (Gerba and Smith, 2005). Unsaturated flow has been proven to promote the pathogen removal due to increased contact time with soil particles and biofilm, favoring the sorption process and biological degradation of pathogens (Powelson and Mills, 2001). In contrast, Mubiru *et al.* (2000)

found that higher soil moisture content affected the survival rate of *E. coli* positively. However, Oron *et al.* (1995) noted that by maintaining high and consistent soil moisture content a higher virus removal could be achieved.

According to Paranychianakis *et al.* (2006), the soil microbial activity might have a strong influence on pathogens in land treatment systems. Especially, the fate of virus is affected by both aerobic (Hurst, 1988) and anaerobic (Nasser *et al.,* 2002) microorganisms. Soil aerobic bacteria have been found to contribute to virus degradation by excreting substances and utilizing viruses as growth substrate (Lipson and Stotzky, 1985), whereas the anaerobic bacteria possible release extracellular enzymes or promote the sorption process (Nasser *et al.,* 2002). Van Cuyk *et al.* (2001) identified the positive influence of microbial activity and formation of biofilm on pathogen removal within soil media.

Once the pathogenic microorganisms are trapped in soil through filtration and sorption, their survival time can be up to 70 days for bacteria and 100 days for viruses, strongly depending on the climate, soil, and local conditions (Jiménez, 2003). Benarde (1973) stated that the pathogenic microorganisms do not multiply in soil. Moreover, exposure to the sun (UV light) causes desiccation affecting their survival. However, this removal process of radiation might be not significant in coppice and forestry application due to the shading from tree canopies (Frankenberger, 1985; Paranychianakis *et al.,* 2006). In general, temperature has a positive effect on pathogen removal. Yeager and O`Brien (1979) stated that increased pathogen removal was noted with higher temperature. Another temperature related crucial effect for such systems under Mongolian condition is the potentially reduced survival rate of pathogens under freeze-thaw process. Sullivan (2013) noted 3 log reduction of *E. coli* after one freeze-thaw circle. Similar results were also reported by Souzu (1980).

Overall, land treatment systems are known to be effective in removing bacteria and viruses from wastewater (Crites and Tchobanoglous, 1998). Tzanakakis (2003) reported a reduction of 6 log units for Fecal Coliforms (FC) and Total Coliforms (TC) in slow rate land treatment. His findings were supported by the statement of US-EPA (2006) suggesting the removal of *E. coli* can be up to 5 log reduction in this type of system.

2.3.4. Metals and other wastewater constitutes

Land-based systems irrigated with primary or secondary treated municipal effluent show low risks concerning heavy metal pollution due to the low metal concentration in applied water. In addition, most of the metals end up in primary sludge (Bouwer and Chaney, 1974). However, when purposely used for metal extraction from phytoremediation sites, willow trees have shown very high extraction capacity for metals such as Cr, Ni, Cu, Zn, Cd, and Pb (Riddell-Black, 1994). Börjesson (1999) stated that extraction capacity of willow trees for Cadmium was 35-70 times higher than that of energy grass and straw.

Another set of pollutants raising concern is toxic organic pollutants. Reed *et al.* (1995) pointed out the land treatment systems as the most efficient approach in terms of the removal of toxic and persistent organic compounds among the natural wastewater treatment systems. The removal mechanisms for persistent organics consist of principal processes such as sorption on soil colloids and organic matter, volatilization, microbial degradation, plant uptake, and transformation. These processes are highly dependent on the physicochemical properties of the applied wastewater, soil and environmental conditions (Paranychianakis *et al.,* 2006). The migration and transport behavior of several pharmaceuticals in different soils have been studied by Kaub (2011), who identified the importance of soil organic matter for removal (retention) of the pharmaceuticals within the soil profile. Although land treatment systems are practiced for centuries as a means to manage wastewater, relatively little information is available about the fate of toxic organic substances (Paranychianakis *et al.,* 2006).

2.4. Effects on soil

Primary treated wastewater might have both positive and negative impacts on the physical, chemical, and biological properties of soil (Vogeler, 2009). Although many studies reveal the positive impacts resulting in improved soil structure due to the increased soil organic matter (SOM), and improved purification efficiency for wastewater due to the enhanced microbial activity, some researchers suggest that land application practices may impact the soil properties negatively, causing surface clogging, reduced purification performance, and altering of the original soil quality (Paranychianakis *et al.,* 2006). Therefore, it is important to monitor following parameters and their long term effects in order to provide stability for Soil-Willow-Systems.

2.4.1. Soil pH

Soil pH is often influenced by the application of wastewater. Some studies reported increase in soil pH. However, these studies dealt with different types of wastewater such as landfill leachate, log yard run-off, and dairy effluents rather than municipal wastewater (Sparling *et al.,* 2001; Paranychianakis *et al.,* 2006). Furthermore, Schipper *et al.* (1996) also noted increase in soil pH, when tertiary treated wastewater was applied to planted soil filters. Other studies mostly dealing with application of primary treated municipal wastewater onto land reported decrease in soil pH (Tzanakakis *et al.,* 2011). The change in soil pH depends on the type of the wastewater, organic load, soil properties, and operational factors. The soil pH decrease is the result of nitrification processes and plant uptake of ammonium nitrogen whereas the opposite occurs as the result of denitrification process, plant uptake of nitrate nitrogen, and application of organic load via carbon dioxide (Larcher, 1994; US-EPA, 2006). When applied onto land at slow rate, wastewater forms unsaturated flow within the soil matrix, which in turn supports the nitrification process due to its aerated condition (Powlson and Addiscott, 2005). In addition, de-nitrification tends to be minor within soil matrix, due to the lesser occurrence of saturated flow of wastewater within soil matrix. This leads to decrease of soil pH especially over a long period of operation. Decreased soil pH may have several impacts on the treatment process and the buffering capacity of the soil. US-EPA (2006) stated that low soil pH might result in metals becoming more soluble and potentially leaching into groundwater. The same applies to phosphorus in soil. As it is mentioned, phosphorus becomes more soluble under acidic condition (Savant and Ellis, 1964). However, soil itself has very good pH buffering capacity and under proper operation, the rate at which the pH decreases can be very low (US-EPA, 2006).

2.4.2. Soil salinity

Another concern of wastewater application onto land is the increase of soil salinity, which mostly reduces the biomass yield and impairs the microbial activity in soil. Although 40% of total dissolved solids (TDS) comprise volatile dissolved solids (VDS), which can be removed by biological degradation process in soil, many studies report increased soil salinity as a result of various types of wastewater application onto land (US-EPA, 2006). High salinity in soil directly inhibits the nitrogen and potassium uptake of plants, whereas phosphorus uptake is indirectly reduced due to a decline in biomass yield (Paranychianakis *et al.,* 2006). High concentration of salts in root zone has direct bearing on the osmotic potential of the soil solution, which regulates the ability of the plants to adsorb water (US-EPA, 2006). Type-1 slow rate systems, which rely on hydraulic loading rate being equal to the water use efficiency (WUE) of the crops, often face salinity problems (US-EPA, 2006), whereas for type-2 slow rate systems, soil salinity seems to be a less of concern. High salt accumulation is managed by flushing out the soil with adequate amount of water (Corwin *et al.,* 2007). The required additional irrigation water known as leaching requirement (LR) or leaching fraction (LF) is originally defined as the water fraction, which needs to pass through plant root zone in order to keep soil salinity at low level, by flushing the salts downwards (Corwin *et al.,* 2007). The usual salinity measured by electric conductivity (EC) in municipal wastewater is in range between 700 $\mu S \cdot cm^{-1}$ and 2500 $\mu S \cdot cm^{-1}$ (Metcalf, 1991). With increasing soil salinity, the number of crops affected will increase (US-DA, 1992). However, Hangs *et al.* (2011) found out that some willow species were tolerant in saline soils with EC_e value up to 8 $dS \cdot m^{-1}$. In general, Hoffman (1985) stated that the minimum leaching fraction (LF) can be as low as 5% when irrigation water is used with EC value of around 1000 $\mu S \cdot cm^{-1}$ for plants with salt tolerance of up to 4 $dS \cdot m^{-1}$ whereas this number was roughly estimated to be 10-15% in order to maintain save operation (US-EPA, 2006).

2.4.3. Soil sodicity (Sodium Adsorption Ratio SAR)

Increased sodium concentrations were found in soils receiving municipal wastewater and other wastewaters such as tannery effluents, gray water, effluents from skin processing plants, or dairy effluents (Al-Hamaiedeh, 201; Menneer *et al.,* 2001; Paranychianakis *et al.,* 2006). The added sodium via wastewater onto land causes swelling of clay particles and dispersion when soil salinity is low, resulting in an altered soil structure and declined water permeability (Duan *et al.,* 2010). This leads to a prevailing water logged condition of the soil, although the plants cannot use the water, due to osmotic effects caused by sodium (Larcher, 1994). Soils with higher clay content are considered to be mostly affected by high SAR (Pearson and Bauder, 2006). According to Duan *et al.* (2010) when soil SAR exceeds 12, serious physical problems start to happen. Soil SAR generally tends to increase, when concentration of Na^+ cation in wastewater is significantly higher than the concentration of Ca^+ and Mg^+ cations (US-EPA, 2006). Therefore, wastewaters such as grey water and effluents from tannery and skin processing industry might pose a risk of soil SAR increase, when applied onto land (Menneer *et al.,* 2001). In short term, municipal wastewater application has lower risk of increase in soil SAR (Duan *et al.,* 2010). However, monitoring of soil SAR is an essential parameter included in proper operation for a land treatment site (Qian and Mecham, 2005).

2.4.4. Soil organic matter (SOM), soil permeability, and clogging

Additional to the decrease in soil permeability potentially caused by sodium, the increase of soil organic matter (SOM) at greater extent might reduce the hydraulic conductivity of the soil (Paranychianakis *et al.,* 2006). When the applied organic matter load (as total suspended solids, TSS) exceeds the mineralization rate of the system, excess organic matter is accumulated at the surface of the soil. This process is referred as the physical clogging of the soil (Vigneswaran and Suazo, 1987). However, the risk of physical clogging might be insignificant due to the fact that land treatment sites are known to be very efficient in organic matter removal at the usual organic loading rates as it was discussed in chapter 2.3.1. Another clogging mechanism known is the biological clogging of the soil surface (Magesan *et al.,* 1999). This is a result of multiple processes including accumulation of microbial cells in the soil pores, secretion of extracellular polymers, and accumulation of insoluble precipitates due to microbial activity (Paranychianakis *et al.,* 2006). Similar to the physical clogging, the biological clogging also depends on the load of organic matter. Additionally, the C:N ratio of applied wastewater has significant influence on biological clogging according to Magesan *et al.* (2000), who stated that there was higher risk of clogging when C:N ratio was high.

On the contrary, the wastewater application might result in decreased SOM, when the wastewater has low concentration of organic matter. McDaniel and Munn (1985) stated that the original mineralization rate in steppe soils is very low due to the cold winter and low precipitation during summer. The wastewater containing low organic matter might stimulate the mineralization rate of such soils.

Once the permeability of soil is significantly reduced, management applications such as changing the loading pattern, which allows wetting and drying cycle, is suggested (Reed *et al.,* 1995). Balks *et al.* (1997) also found that the soil hydraulic conductivity was recovered after 23 and 50 days at 25° C and 13° C, respectively, after the irrigation was stopped. Long term study shows that under proper management the SOM matter increased to a certain level, that improved the microbial activity, and the water holding capacity of the soil, but did not pose risk of clogging (Friedel *et al.,* 2000).

2.4.5. Soil nutrients

Numbers of studies show that application of wastewater onto land results in accumulation of nutrients in soil (Loehr, 1984). Nitrogen applied to the soil can be readily transformed to nitrates, posing risk of potential leachate (Powlson and Addiscott, 2005). However, when the groundwater table is low, the nitrate will travel long distance along the soil profile under anoxic condition. There might be potential occurrence of denitrification process before the water reaches groundwater. However, the denitrification process needs organic matter as source.

In terms of phosphorus, the risk that it will leach into groundwater is not evident (Paranychianakis *et al.,* 2006). However, due to the fact that phosphorus is accumulated close to the soil surface, there might be a potential risk of surface water pollution through runoff (Paranychianakis *et al.,* 2006). As it is discussed in chapter 2.3.2.2 most soils have high phosphorus retention capacity. Ryden and Pratt (1980) estimated that the saturation of 2 m deep soil would occur in 40 years, when the annual application rate is around 150 kg $TP \cdot ha^{-1}$. When the groundwater table is low, a short rotation coppice system for wastewater treatment can be properly operated for extensive period of time without any adverse effects on soil and groundwater in terms of phosphorus.

2.5. Willow and Poplar wood properties and biomass yield

Willow and poplar trees are often used in short rotation coppice for wastewater treatment due to its high biomass yield (Ledin, 1996), which also reflects the high nutrient uptake capacity (Perttu and Kowalik, 1997), high tolerance of salt and temperature (Hangs *et al.,* 2011; Sakai, 1970), and the production of environmental friendly fuel (IPC, 1979). Another factor which makes both trees attractive for short rotation intensive culture is their ability to propagate from cuttings, which makes the planting process easy with the aid of machinery (IPC, 1979). In addition Perttu and Kowalik (1997) identified that the proportion of nutrients associated with wastewater (N:P:K=100:14:64) was very similar to that required for willow growth (N:P:K=100:14:72). This was later also confirmed by Hasselgren (1998). Other benefit of willow and poplar trees is their tolerance to a wide range of climatic and edaphic factors (Ledin, 1996). However, poplars are more suitable for short rotation forestry rather than short rotation coppice systems, due to its high quality timber production (Ferm *et al.,* 1989). It is roughly estimated that the calorific value of 1 t of DM of willow and poplar trees are equivalent to that of 0.7 t brown coal (ECN, 2016).

Biomass yield depends on climatic conditions, soil properties, hydraulic load, and irrigation managements such as irrigation method and loading pattern. Mirch *et al.* (2005) reported that various biomass yield were achieved in different parts of Sweden, indicating the influence of climatic conditions on biomass yield. According to Dimitrou and Aronsson (2001), wastewater application on willow stands increases the biomass yield compared to the application of sewage sludge. He also stated that higher clay content in soil had positive influence on the growth of willow and poplar trees and biomass yield. It is suggested to maintain plant density of 16000-2000·ha^{-1} in order to achieve the high biomass yield and further increase in biomass yield as result of higher density would be very small compared to the effort for planting (Wilkinson *et al.,* 2007). Aronsson *et al.* (2002) noted the influence of spatial condition and irrigation on tree growth, while Christersson (1986) found no significant influence of different irrigations, such as sprinkler, drip irrigation and sub-surface irrigation methods on the biomass production in a relatively large (3000-6000 m^2) scale short rotation coppice with willow and poplar trees.

In general, it is reported that wastewater application often had positive influence in terms of biomass yield when compared to rain fed and fertilized systems (Perttu and Kowalik, 1997). These findings were also supported by Börjesson (1999), who stated that the wastewater irrigation increased the biomass production on average by 50% in short rotation willow coppice.

Hardcastle (2006) reported an average biomass yield of 8 DM t·ha^{-1}·a^{-1} and 6 DM t·ha^{-1}·a^{-1} for willow and poplar in conventional short rotation forestry. However, wastewater irrigated coppice systems have shown higher biomass yield. A biomass yield of 15 DM t·ha^{-1}·a^{-1} was reported for willows received wastewater application (Dimitrou and Aronsson, 2011), while for poplars a biomass yield of 12 DM t·ha^{-1}·a^{-1} was reported by Perttu and Kowalik (1997). However, in a short rotation intensive culture for willow and poplar trees, biomass yields up to 35 DM t·ha^{-1}·a^{-1} were recorded (Perlack *et al.*, 1986). Ceulemans *et al.* (1996) summarized the biomass yields for willow and poplar trees. The average biomass yield for willow trees in Sweden is reported to be 10-12 DM t·ha^{-1}·a^{-1}, ranging between 8 and 36 DM t·ha^{-1}·a^{-1}.

2.6. Soil-Willow-System under cold climate (Influence of cold climate)

A concern with a (natural wastewater treatment plant) Soil-Willow-System might be its operation during winter. Low temperatures cause reduction and inhibition of microbial activity of soil and ice formation, which result in deterioration of both physical and biological performance of the system (US-EPA, 1987). There are few potential operational options for Soil-Willow-System during winter.

Option one is suggested by Perttu *et al.* (2003). They suggested using Soil-Willow-System only during summer by providing either winter storage or building the Soil-Willow-System as an additional stage for existing conventional wastewater treatment plants. In both cases, the wastewater is either stored or treated in conventional wastewater treatment plant during winter. There have been several studies conducted on willow systems concerning both wastewater treatment and biomass production under cold climate in Sweden, Estonia, Poland, and northern China (Perttu *et al.,* 2003). One commonly referred example is the city of Enköping in Sweden with a population of 20 000 people uses an 80 ha of tree plantation as part of its treatment approach for domestic wastewater. Timber generated from the site is used in a local district heating system (Börjesson and Berndes, 2006). A similar system has been applied for the treatment of domestic wastewater from a population of 50 000 people in Huolinguole city, China (Ou *et al.,* 1997). Ou et al. (1997) described the application of wastewater to irrigate a number of different tree species including *Larix, Pinus* and *Populus*. The described system occupies a field area of 880 ha for daily treatment flow of 10 000 m^3 of municipal wastewater (Ou *et al.,* 1997).

Another option would be to use the system continuously throughout the year as US-EPA (1987) suggested. It mentions several successful applications of slow rate systems in cold tempered areas in Northern U.S. and their continuous year around operation under proper maintenance. In this case, higher hydraulic load is applied to the system to maintain higher temperature within the system to avoid freezing of the soil layer. Ice cover and snow, thereafter, serve as additional insulation, while the soil temperature is around 0 °C. US-EPA (1981) noted that the soil infiltration can occur even when the soil temperature drops to minus 4°C. Especially, forest crops are most suitable for this option when compared to agricultural crops. A slow rate system, in Central Vermont showed three times higher concentration of BOD, TSS, and TN in drainage water during winter compared to that during summer time operation.

The third option would be to implement this so called "internal winter storage". This approach is relatively new, however, similar to the Zero Discharge Systems, which receive hydraulic load throughout the year and the wastewater is stored within the bed during the wintertime (Gregson and Brix, 2001).

The potential solutions for winter operation in Mongolia are shown in Figure 2.2 and Figure 2.3.

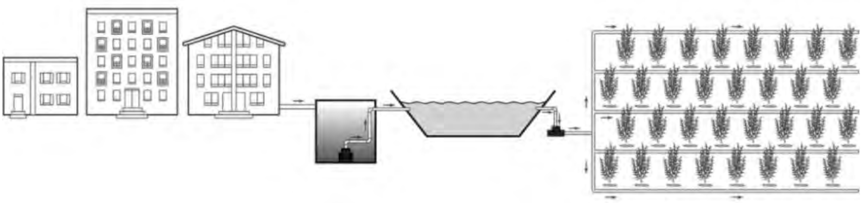

Figure 2.2: Winter storage and summer irrigation

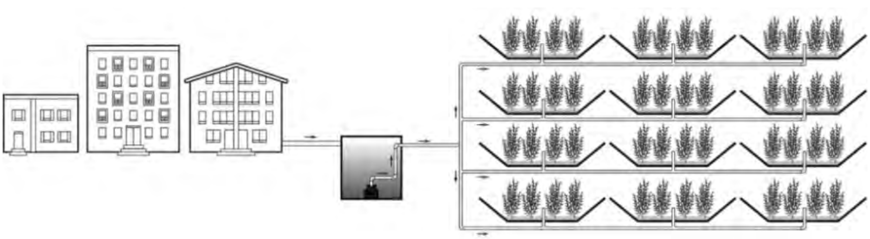

Figure 2.3: Continuous delivery of wastewater to integrated storage and treatment beds

2.7. Conclusion on basis of the literature review

There are available data related to the wastewater treatment performance and wood production of land treatment and short rotation coppice systems in many regions of the world. However, these numbers are not directly applicable to the Mongolian conditions.

It is not well studied how domestic trees will respond to the wastewater application under different loading rates and loading patterns in this climate. Additionally, it is not well studied how the presence of the trees will influence (potentially enhance) the treatment process when wastewater passes along the soil filter.

While the Mongolian winters are cold and long, the summers are relatively warm. This drastic change of temperature potentially influences both treatment performance and wood production. Similar systems have been investigated either in warm or cold regions. The behavior of both plants and the treatment performance of the system are not known under these extreme climatic conditions.

Several similar systems were tested during winter in northern parts of U.S. and in Scandinavian countries. However, these studies reported about continuous operation of the systems without having ice formation on the treatment area. The effects related to ice formation on the treatment area and the ice melt in spring are not well studied.

Operational variations, such as different weekly and daily loading patterns might have influence of the behavior of the system. These different loading patterns might reveal the beneficial effects such as increased wood production and/ or enhanced treatment performance. Unlike to technical variations, which require additional system component or equipment, these operational variations should be much easier to implement and adapt to the Soil-Willow-System. The numbers of studies and the results regarding this aspect are limited.

From these uncertainties in research, the following research demands are concluded:

1. As it is mentioned in chapter 1.1, there are 64 infiltration basins, which receive partly or non-treated wastewater in Mongolia. To estimate the approximate treatment performance of the existing infiltration basins, experiments on unplanted soil filter are necessary. Furthermore, an investigation on the influence of the trees on treatment performance is needed in order to demonstrate the beneficial effects of the trees.
2. It is also crucial to study the treatment performance under different loading rates in order to assess the removal capacity of the system. The loading rate is directly proportional to the surface area. Therefore, it is an important parameter of cost for Soil-Willow-Systems.
3. The investigation on different seasonal loading patterns will reveal the importance of the external winter storage and would offer an insight to the treatment performance of the system receiving irrigation during different seasons of the year. Both land treatment systems and short rotation willow coppice for wastewater treatment have been operated in temperate climate regions during winter times. These systems had often continuous load without any ice forming on the treatment area. There is a demand to study and investigate the so called "internal storage" (Figure 2.2) and its influence associated with the ice accumulation and ice melt on.
4. Furthermore, additional operational variations such as different weekly and daily loading patterns and their effects on treatment performance and the wood production should be investigated. The weekly loading pattern might reveal the advantages of Soil-Willow-System being irrigated once per week and thus the homogeneous distribution on the treatment area.
5. Moreover, investigation of different daily loading patterns might reveal the potential improvement of the system in terms of wood production and treatment performance.

6. A design recommendation for a town as an example is needed to demonstrate the beneficial effects of the Soil-Willow-System in Mongolia.

3. Materials and methods

3.1. Introduction

The experimental work was conducted at two pilot sites. One pilot plant was established at the Mongolian University of Science and Technology (MUST) in Darkhan and tested for two years from 2012 until 2014. The second pilot plant was established in Langenreichenbach (LRB), Germany and experiments were carried out for two years from 2013 to 2015.

3.2. Pilot plant in Darkhan, Mongolia

Within the frame of the MoMo-II project (Integrated Water Resource Management in Central Asia-Model Region Mongolia), a pilot plant was established at the Mongolian University of Science and Technology (MUST) in Darkhan City. The construction of the pilot plant was carried out in summer 2011 and the experiments started in May 2012. The pilot plant consisted of a primary settling tank, four identically built treatment beds, a control area, and a sampling manhole.

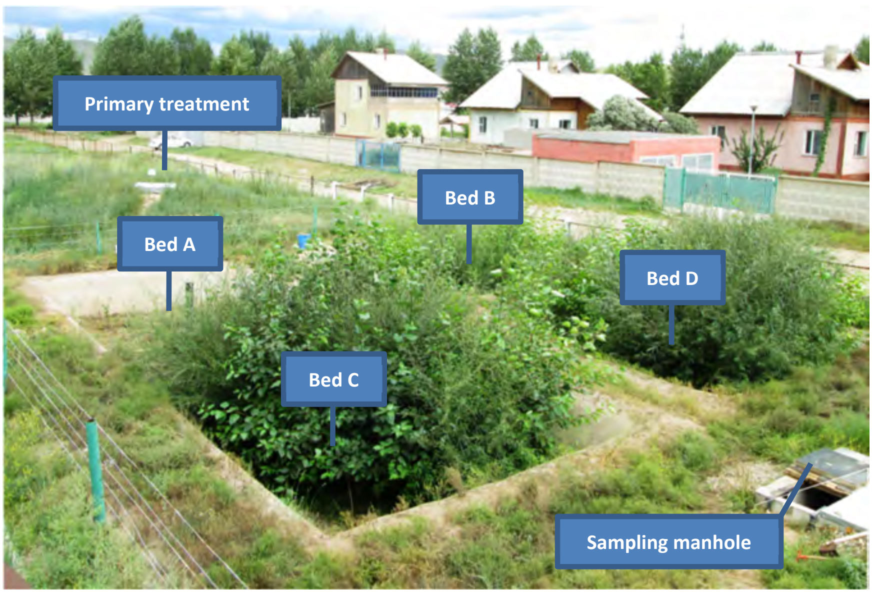

Figure 3.1: The Pilot plant in Darkhan (Photo by: Chris Sullivan)

3.2.1. The components of the pilot plant

3.2.1.1. Origin of wastewater

The wastewater for the pilot plant was obtained from an existing sewer line that crossed the campus of the MUST. The sewer line collects approximately 40-70 m^3 of wastewater per day, which originates from 50 one-family houses neighboring the university and its campus.

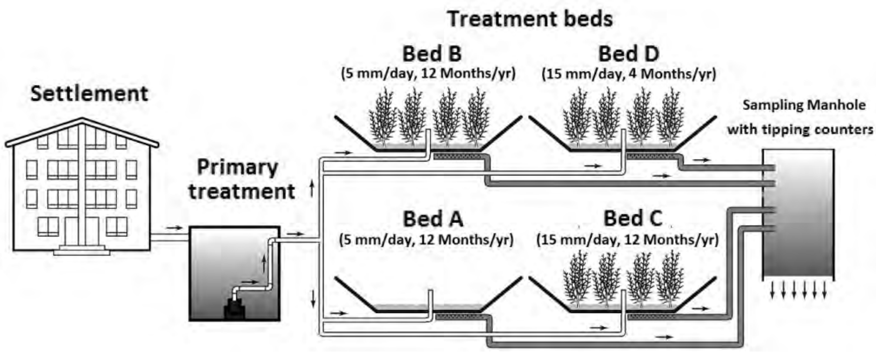

Figure 3.2: Schematic drawing and experimental conditions of the pilot plant at the MUST in Darkhan City, Mongolia.

3.2.1.2. Primary settling tank

An existing sewer manhole on the MUST-campus was intercepted at the depth of 3.5 m and the wastewater was diverted to the primary settling tank of the pilot plant.

Settling tank was constructed at 5.5 m below ground level as a concrete tank with 15 cm steel reinforced concrete walls and an access shaft of 4 m depth with diameter of 1.2 m. The settling tank was designed as a three chamber system with a total volume of approximately 5 m^3. The volume of the settling tank was calculated by considering the technical guideline for settling tank DIN 4261-1 (DIN, 2010). The volume of the first compartment was 2.5 m^3, while the remaining two compartments were 1.2 m^3 each. The third compartment contained three 20 $L \cdot min^{-1}$ submersible pumps (C290W, Homa-Sindersberger & Wohnwelt GmbH & Co. KG, Neunkirchen-Seelscheid, Germany).

The pumps fed the treatments beds A, B, C, and D with primary treated wastewater through a single 25 mm diameter irrigation pipes. Beds A and B were connected to one pump, while bed C and D were connected to separate pumps. The pumping time and duration was regulated by electronic timers (OBI Wochenzeitschaltuhr 98, OBI GmbH, Leipzig, Germany). In order to avoid freezing of the wastewater in the irrigation