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Chapter 8

Resource recovery from sewage sludge

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X.1 INTRODUCTION

More than 10 million tons of sewage sludge was produced in the European Union (EU) in 2010 (Eurostat, 2014). For the disposal of sewage sludge (solid, semisolid, or liquid residue generated during the treatment of domestic sewage), chemical, thermal or biological treatment, which may include composting, aerobic and anaerobic digestion, solar drying, thermal drying (heating under pressure up to 260◦C for 30 min), or lime stabilisation (addition of $Ca(OH)_2$ or CaO such that pH is ≥ 12 for at least 2 h), produces a stabilised organic material.

The Waste Framework Directive (2008/98/EC; EC 2008) lays down measures to protect the environment and human health by preventing or reducing adverse impacts resulting from the generation and management of waste. Under the directive, a hierarchy of waste is applied: prevention, preparing for re-use, recycling, other recovery and disposal. The objective of the Directive is to maximise the resource value and minimise the need for disposal (EC 2008). This has prompted efforts within sewage sludge management to utilise sewage sludge as a commodity. The terminology 'biosolids' reflects the effort to consider these materials as potential resources (Isaac and Boothroyd 1996). Biosolids may be used in the production of energy, bio-plastics, polymers, construction materials and other potentially useful compounds. However, as the disposal of sewage sludge is commonly achieved by recycling treated sludge to land, nutrient recovery, particularly in the context of pressure on natural resources, and potential barriers to its reuse on land (environmental, legislative), deserves particular attention.

The aim of this chapter is to examine the recovery of nutrients and other compounds, such as volatile fatty acids (VFA), polymers and proteins, from sewage sludge. Due to the increasing awareness regarding risks to the environment and human health, the application of sewage sludge, following treatment, to land as a fertilizer in agricultural systems has come under increased scrutiny. Therefore, any potential benefits accruing from the reuse of sewage sludge are considered against possible adverse impacts associated with its use. Finally, the potential costs and benefits arising from its re-use are examined.

X.2 DEFINING TRENDS FOR MUNICIPAL SLUDGE TREATMENT

The amount of sewage sludge produced in Europe has generally increased (EC 2011), which is mainly attributable to implementation of the Urban Waste Water Treatment Directive 91/271/EC (EC, 1991) and other legislative measures.

The treatment and disposal of sewage sludge presents a major challenge in wastewater management. As seen over the last decade, the upgrading and development of effective treatment plants has facilitated efforts to improve the quality of the effluent (i.e. removal of microorganisms, viruses, pollutants). Subsequently, legislation regarding sewage sludge in the EU (Sewage Sludge Directive 86/278/EEC; EEC, 1986) and the USA (40 CFR Part 503; USEPA 1994) has focused on effluent quality and potential contamination. Within the EU, treated sewage sludge is defined as having undergone biological, chemical or heat treatment, long-term storage, or any other appropriate process so as to significantly reduce fermentability and any health hazards resulting from its use (EC 2012). Physical-chemical treatment of wastewater has been widely practiced, introducing biodegradation and chemical advanced oxidation for biological treatment (Mouri et al. 2013). In the treatment of wastewater, biological treatments, such as aerobic and anaerobic digestion, appear to be the more favoured option. Aerobic treatment has a high degree of treatment efficiency, whilst anaerobic biotechnology has significantly progressed, offering resource recovery and utilization while still achieving the objective of waste control (Chan et al. 2009). A variety of sewage sludge treatment technologies can be employed and are implemented according to regulations. As can be seen from Table X.1, significant differences in sewage sludge treatments can be observed between the EU, USA and Canada. With regards to sludge stabilization, aerobic and anaerobic treatments are the most widely used methods of sewage sludge treatment. Within the EU, anaerobic and aerobic wastewater treatments appear to be the most common methods, with 24 countries out of 27 applying this method (Kelessidis and Stasinakis 2012).

Anaerobic digestion (AD) is most commonly used in Spain, Italy, United Kingdom and Czech Republic (Table X.1). Within the USA and Canada, biosolids are classed according to their pathogenic levels. Class A biosolids contain minute levels of pathogens and must undergo heating, composting, digestion, or increased pH. Thus, these methods are more commonly employed (Table X.1). Class B biosolids have less stringent parameters for treatment and contain small, but compliant, amounts of bacteria (USEPA 2011). In order to achieve Class A biosolids, the sewage sludge must undergo stringent treatment. Stabilization methods such as aerobic, anaerobic, liming and composting, are the recommended options in both the USA and Canada.

X.3 SEWAGE SLUDGE AS A RESOURCE

The two components in sewage sludge that are technically and economically feasible to recycle are nutrients (primarily nitrogen (N) and phosphorus (N)) and energy (carbon) (Tyagi and Lo 2013). As sewage sludge contains organic matter, energy can be recovered whilst treating it. There are a considerable amount of nutrients within sewage sludge, especially P and N. However, P is fast becoming the most significant nutrient due to depleting sources. Emerging technologies have been developed to extract this valuable resource including KREPO, Aqua-Reci, Kemicond, BioCon, SEPHOS and SUSAN, and are based on physical-chemical and thermal treatment to dissolve the P, with final recovery by precipitation (Cordell et al. 2011; Tyagi and Lo 2013). Other resources include the reuse of sludge for construction materials, heavy metals, polyhydroxyalkanoates (PHA), proteins, enzymes and VFA. Table X.2 gives an overview of resource recovery products from sewage sludge, their typical values and uses. Apart from the recovery products mentioned in Table X.2, advances in technology have revealed innovative emerging products from treated sewage sludge (biosolids) and include VFA, polymers, and proteins in the form of worms, larvae and fungi. A short review regarding production, processes and further use is provided on each emerging product.

Table X.1. Global municipal sewage sludge treatment processes

	Denmark[a]	France[a]	Germany[a]	Greece[a,b]	Ireland[a]	Italy[a]	Spain[a]	Sweden[a]	UK[a]	Czech Rep.[a]	Poland[a]	USA[c]	Portugal[d]
Stabilisation													
Aerobic	✓	✓		✓✓	✓	✓	✓	✓	✓	✓	✓✓	✓	✓
Anaerobic	✓	✓	✓	✓	✓	✓✓	✓✓	✓	✓✓	✓✓	✓	✓	✓
Lime	✓	✓		✓	✓	✓	✓	✓	✓		✓	✓	✓
Composting	✓	✓		✓	✓	✓	✓	✓	✓	✓✓	✓	✓	✓
Conditioning													
Lime	✓					✓							
Inorganics						✓		✓					
Polymers				✓									
Thermal			✓			✓		✓			✓		
Drying belts				✓									
								✓					
Dewatering													
Filter press		✓		✓	✓	✓			✓		✓		
Centrifuges		✓						✓			✓		✓
Belt filter press	✓			✓✓	✓	✓		✓	✓		✓		✓
Others													
Thermal		✓	✓✓	✓	✓	✓	✓	✓	✓			✓	
Solar drying	✓	✓				✓	✓						
Pasteurisation												✓	
Long-term storage				✓	✓	✓	✓	✓	✓		✓		
Cold fermentation											✓		
bag filling													

✓ Common use ✓✓ most common use

[a] Kelessidis and Stasinakis (2012); [b] Tsagarakis et al. (1999) [c] Lu et al. (2012) [d] Martins and Béraud, pers. comm.

Table X.2. Resource recovery products from sewage sludge

Products	Typical values and uses	Reference
Nitrogen	2.4 – 5% total solids	Tchobanoglous et al., 2003
Phosphorus	0.5 – 0.7% total solids	Tchobanoglous et al., 2003
Heavy metals	Typical recovery values: Ni 98.8%; Zn 100.2%; Cu 93.3%	Pérez-Cid et al., 1999
Construction materials	Dried sludge or incinerator ash. Biosolid ash is used to make bricks	Tay and Show, 1997
Bio-plastic	Microorganisms in activated sludge can accumulate PHAs ranging from 0.3 to 22.7 mg polymer / g sludge	Yan et al., 2008
Enzymes	Protease, dehydrogenese, catalase, peroxidase, α-amylase, α-glucosidase	Tyagi and Surampalli, 2009

X.3.1 Nutrient recovery from sewage sludge

Treated sewage sludge may be used as an agricultural fertiliser, as they contain organic matter and inorganic elements (Girovich 1996). The recycling of treated sewage sludge to agriculture as a source of the fundamental nutrients and metals required for plant growth is going to be essential for future sustainable development, as it is estimated that there are only reserves of 50-100 years of P depending on future demand (Cordell et al. 2009). When spread on arable or grassland, and provided that it is treated to the approved standards, treated sewage sludge may offer an excellent source of nutrients and metals required for plant and crop growth (Jeng et al. 2006). Treated sewage sludge can also contribute to improving soil physical and chemical characteristics (Mondini et al. 2008). It increases water absorbency and tilth, and may reduce the possibility of soil erosion (Meyer et al. 2001).

Land application of treated sewage sludge to agricultural land can be relatively inexpensive in countries in which it is considered to be a waste material. An alternative, but costly, option in such countries is to pay tipping fees for its disposal (Sonon and Gaskin 2009). However, in some countries sewage sludge is seen not as a waste but instead as a product containing valuable nutrients (e.g. the U.K) with an associated fertiliser replacement value (FRV) and cost for its usage.

As the world population increases, pressure on natural resources, especially food, oil and water, will increase. Inorganic fertilizer prices are tied to crude oil prices globally and demand (Bremer 2009): when prices of oil are high, inorganic fertilizer prices also climb. For instance, in Ireland, the cost of inorganic fertilisers has continually increased, with the cost of a mean kg of N, P and potassium (K) rising from €0.41, 1.06 and 0.23 in 1980 to €103, 203, 105 in 2011 (Figure X.1). Similar price increases of 13% were seen in the U.K. in 2010 (Tasker 2010). Recent fertiliser increases since 2008 can be attributed to increases in both energy costs and global demand for fertilisers. Increased prices and volatility are important considerations, as they lead to volatility in farm input costs and profit margins, and make farm planning more difficult and risky (Lalor et al. 2012).

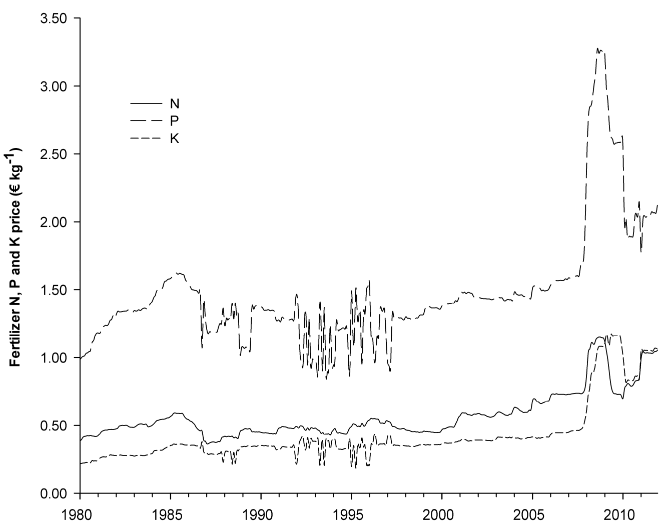

Figure X.1. Trends in unit cost of nitrogen (N), phosphorus (P) and potassium (K) in chemical fertilisers in Ireland from 1980 to 2011 (Lalor et al. 2012).

Nutrient price equivalents of sewage sludge will depend on the nutrient availability and the FRV of the nutrients in the sludge. The FRV of nutrients in cattle slurry over time was calculated in Lalor et al. (2012)

assuming a total N, P and K content in slurry of 3.6, 0.6 and 4.3 kg m^{-3}, respectively, and an assumption of respective FRV of 25%, 100% and 100% (Coulter 2004). Of course in treated sewage sludge as in other nutrient streams, micronutrients used by the plant give added value to the product. In addition, factors such as transport and land application costs would also need to be considered in an overall assessment. It is therefore essential that such data are known for treated sewage sludge.

There is a good body of literature that has examined its fertilisation potential (Smith and Durham 2002; Epstein 2003; Singh and Agrawal 2008). Siddique and Robinson (2004) mixed AD-treated sewage sludge, poultry litter, cattle slurry and an inorganic P fertiliser with five soil types at rates equivalent to 100 mg P kg^{-1} soil and, following incubation at 25^{o}C for 100 d, found that AD-treated sewage sludge and poultry litter had a slower rate of P release compared with cattle slurry and inorganic P fertiliser. This may indicate that it may have good long-term fertilisation potential.

One of the main concerns associated with the use of treated sewage sludge as an organic fertiliser on grassland are the loss of nutrients, metals and pathogens along a transfer continuum (Wall et al. 2011) to a waterbody *via* direct discharges, surface and near surface pathways and/or groundwater discharge. More recently, so-called ‘emerging contaminants’, which may include antibiotics, pharmaceuticals and other xenobiotics, have been considered, as they have health risks associated with them. Therefore, nutrient recovery from treated sewage sludge must be considered against possible adverse impacts associated with its use.

X.3.2 Volatile fatty acids

Volatile fatty acids are short-chained fatty acids consisting of six or fewer carbon atoms which can be distilled at atmospheric pressure (Lee et al. 2014). Proteins and carbohydrates in sewage sludge can be converted into VFA to enhance methane, hydrogen and poly-hydroxyalkanoate production (Yang et al. 2012). The production of VFA from biosolids is an anaerobic process involving hydrolysis and acidogenesis (or dark fermentation) (Su et al. 2009). In hydrolysis, complex polymers in waste are broken down into similar organic monomers by the enzymes excreted from the hydrolytic microorganisms. Subsequently, acidogenesis ferment these monomers into mainly VFA such as acetic, propionic and butyric acids. Both processes involve a conglomerate of obligate and facultative anaerobes such as Bacteriocides, Clostridia, Bifidobacteria, Streptococci and Enterobacteriaceace (Lee et al. 2014).

X.3.3 Polymers

Extracellular polymeric substances (EPS) are the major constituents of organic matter in sewage sludge floc, which comprises polysaccharides, proteins, nucleic acids, lipids and humic acids (Jiang et al. 2011). They occur in the intercellular space of microbial aggregates, more specifically at or outside the cell surface (Neyens et al. 2004), and can be extracted by physical (centrifugation, ultrasonication and heating, for example) or chemical methods (using ethylenediamine tetraacetic acid, for example), although formaldehyde plus NaOH has proven to be effective in extracting EPA from most types of sludge (Liu and Fang, 2002). Extracellular polymeric substances perform an important role in defining the physical properties of microbial aggregates (Seviour et al. 2009). There are many biotechnical uses of EPS, including the production of food, paints and oil drilling ‘muds’; their hydrating properties are also used in cosmetics and pharmaceuticals. Furthermore, EPS may have potential uses as biosurfactants e.g. in tertiary oil production, and as biological glue. Extracellular polymeric substances are an interesting component of all biofilm systems and still hold large biotechnological potential (Flemming and Wingender 2001). A relatively new method for treatment of sewage sludge is aerobic granular sludge technology (AGS; Morgenroth et al. 1997). A special characteristic of AGS is the high concentration of alginate-like exopolysaccharides (ALE) with different properties compared to converted activated sludge. Aerobic granular sludge technology produces a compound with similar characteristics as alginate, which is a polymer normally harvested from brown seaweed. Alginate-like exopolysaccharides can be harvested and used as a gelling agent in textile printing, food preparation and the paper industry (Hogendoorn 2013). Lin et al. (2010) demonstrated that the potential yield of extractable alginate-like exopolysaccharides reached 160 ± 4 mg/g (VSS ratio). It was also found that they were one of the dominant exopolysaccharides in aerobic granular sludge.

X.3.4 Proteins

Vermicomposting (sludge reduction by earthworms) is a relatively common technology, especially in developing countries with small scale settings. The main product of this process is vermicompost, which consists of earthworm faeces that can be used as a fertilizer due to its high N content, high microbial activity and lower heavy metal content (Ndegwa and Thompson 2001). Vermicomposting results in bioconversion of the waste streams into two useful products: the earthworm biomass and the vermicompost. In a study by Elissen et al. (2010), aquatic worms grown on treated municipal sewage sludge, produced high protein values with a range of amino acids. These proteins can be used as animal feed for non-food animals, such as aquarium fish or other ornamental aquatic fish. Other outlets for the protein could be technical applications such as coatings, glues and emulsifiers. The study also revealed that the dead worm biomass can be utilized as an energy source in anaerobic digestion. Experiments have shown that biogas production of worms is three times that of sewage sludge. Other applications include fats and fatty acid extraction. Treatment of sewage sludge using earthworms has been well documented; however, research studies on protein extraction of earthworms grown on sewage sludge are very limited.

Bioconversion of biosolids using fly larvae has been studied for years. Organic waste has a high nutritional and energy potential and can be used as a feed substrate for larvae. Apart from significantly reducing organic waste, grown larvae make an excellent protein source in animal feed. The insect protein could be used in animal feed to replace fishmeal (Lalander et al. 2013). One of the most studied species is the larvae of the Black Soldier fly (*Hermetia illucens* L.). The larvae of this non-pest fly feed on, and thereby degrade, organic material of different origin (Diener et al. 2011a). The 6th instar, the prepupa, migrates from the sludge to pupate and can therefore easily be harvested. Since prepupae contain on average 44% crude protein and 33% fat, it is an appropriate alternative to fishmeal in animal feed (St-Hilaire et al. 2007). Proposals for other uses for the pupae other than animal feed have been put forward. The other components of the pupae (protein, fat, and chitin) could be fractioned and sold separately. The extracted fat can be converted to biodiesel; chitin is of commercial interest due to its high percentage of N (6.9%) compared to synthetically substituted cellulose (1.25%) (Diener et al. 2011b). There has been ample research on the *H. illucens* and its contribution to significantly reducing organic wastes; however, there are several knowledge gaps on the potential utilization of the pupae in terms of protein, fat and chitin.

Filamentous fungi are often cultivated in food industries as a source of valuable products such as protein and a variety of biochemicals, using relatively expensive substrates such as starch or molasses (More et al. 2010). The biomass produced during fungal wastewater treatment has potentially a much higher value in the form of valuable fungal by-products such as amylase, chitin, chitosan, glucosamine, antimicrobials and lactic acids, than that from bacterial activated sludge process (van Leeuwen et al. 2012). The use of fungi for the production of value added products has been presented by several researchers (Molla et al. 2012).

X.3 LEGISLATION COVERING DISPOSAL OF BIODEGRADABLE WASTE ON LAND

Recent estimates of the disposal methods of sewage sludge in EU Member States indicate that although the amount of sewage sludge being applied to land in the EU has dramatically increased, landfill and incineration are still common (EC 2010), particularly in countries where land application is banned. Less common disposal routes are silviculture, land reclamination, pyrolysis, and reuse as building materials. The drive to reuse sewage sludge has been accelerated by, amongst other legislation, the Landfill Directive, 1999/31/EC (EC, 1999), the Urban Wastewater Treatment Directive 91/271/EEC (EC 1991), the Waste Framework Directive (2008/98/EC; EC 2008), and the Renewable Energy Directive (2009/28/EC; EC 2009), which places an increased emphasis on the production of biomass-derived energy.

The application of treated sewage sludge to agricultural land is governed in Europe by EU Directive 86/278/EEC (EEC 1986), which requires that sewage sludge undergoes biological, chemical or heat treatment, long-term storage, or any other process to reduce the potential for health hazards associated with its use. In the EU, land application of treated sewage sludge is typically based on its nutrient and metal content, although individual member states often have more stringent limits than the Directive (EC 2010; Milieu et al. 2013a,b,c). Generally, when applying treated sewage sludge based on these guidelines and depending on the nutrient and metal content of the treated sewage sludge, P becomes the limiting factor for application. In the USA., the application of treated sewage sludge to land is governed by *The Standards for the Use or Disposal of Sewage Sludge* (USEPA 1993), and is applied to land based on the N requirement of the crop being grown and is not based on a soil test (McDonald and Wall 2011). Therefore, less land is required for the disposal of treated sludge than in countries where it is spread based on P content. Evanylo (2006) suggests that when soil P poses a threat to water quality in the USA, the application rate could be determined on the P needs of the crop.

X.4 EXISTING AND EMERGING ISSUES CONCERNING THE RE-USE OF BIODEGRADABLE WASTE ON LAND

X.4.1 Societal issues

One of the major stumbling blocks in the use of treated sewage sludge as a low-cost fertiliser is the issue of public perception (Apedaile 2001). Concerns have been raised over potential health, safety, quality of life and environmental impacts that the land spreading of sludge may have (Robinson et al. 2012). This perception could be, in part, due to the fact that treated sewage sludge is heavily regulated or that animal manure is more commonly seen and used. In many countries such as Ireland, for example, companies that produce products for the food and drinks industry will not allow the use of the raw materials produced from agricultural land which has been treated with treated sewage sludge (FSAI 2008). This limits their use as a fertiliser at the current time.

X.4.2 Nutrient and metal losses

Phosphorus and reactive N losses to a surface waterbody originate from either the soil (chronic) or in runoff where episodic rainfall events follow land application of fertiliser (incidental sources) (Brennan et al. 2012). Such losses to a surface waterbody occur *via* primary drainage systems (end of pipe discharges, open drain networks (Ibrahim et al. 2013), runoff and/or groundwater discharges. Application of treated sewage sludge to soils may also contribute to soil test phosphorus build-up in soils, thereby contributing to chronic losses of P, metal and pathogen losses in runoff (Gerba and Smith 2005). Dissolved reactive P losses may also be leached from an agricultural system to shallow groundwater (Galbally et al. 2013) and, where a connectivity exists, may affect surface water quality for long periods of time (Domagalski and Johnson 2011; Fenton et al. 2011).

The metal content of treated sludge and of the soil onto which it can be spread is also regulated by legislation in Europe (86/278/EEC; EEC, 1986). However, guidelines governing the application of treated sewage sludge to land (e.g. Fehily Timoney and Company 1999) mean that is frequently the case that application rates are determined by the nutrient content of the sludge and not its metal content (Lucid et al. 2013). Regardless, concerns have been raised about the potential for transfer of metals into water bodies, soil structures and, consequently, the food chain (Navas et al. 1999). In countries such as the USA, where treated sewage sludge is applied to land based on the N requirement of the crop being grown and not on a soil-based test (McDonald and Wall 2011), excessive metal losses may potentially occur.

X.4.3 Pathogens

During wastewater treatment, the sludge component of the waste becomes separated from the water component. As the survival of many microorganisms and viruses in wastewater is linked to the solid fraction of the waste, the numbers of pathogens present in sludge may be much higher than the water component (Straub et al. 1992). Although treatment of municipal sewage sludge using lime, AD, or temperature, may substantially reduce pathogens, complete sterilisation is difficult to achieve (Sidhu and Toze 2009) and some pathogens, particularly enteric viruses, may persist. Persistence may be related to factors such as temperature, pH, water content (of treated sludge), and sunlight (Sidhu and Toze 2009). Also, there is often resurgence in pathogen numbers post-treatment, known as the ‘regrowth’ phenomenon. This may be linked to contamination within the centrifuge, reactivation of viable, but non-culturable, organisms (Higgins et al. 2007), storage conditions post-centrifugation (Zaleski et al. 2005), and proliferation of a resistant sub-population due to newly available niche space associated with reduction in biomass and activity (McKinley and Vestal 1985).

The risk associated with sludge-derived pathogens is largely determined by their ability to survive and maintain viability in the soil environment after landspreading. Survival is determined by both soil and sludge characteristics. The major physico-chemical factors that influence the survival of microorganisms in soil are currently considered to be soil texture and structure, pH, moisture, temperature, UV radiation, nutrient and oxygen availability, and land management regimes (reviewed in van Elsas et al. (2011)), whereas survival in sludge is primarily related to temperature, pH, water content (of treated sewage sludge), and sunlight (Sidhu and Toze 2009). Pertinent biotic interactions include antagonism from indigenous microorganisms, competition for resources, predation and occupation of niche space (van Elsas et al. 2002). Pathogen-specific biotic factors that influence survival include physiological status and initial inoculum concentration (van Veen et al. 1997).

Following landspreading, there are two main scenarios which can lead to human infection. First, pathogens may be transported *via* overland or sub-surface flow to surface and ground waters, and infection may arise via ingestion of contaminated water or accidental ingestion of contaminated recreational water (Jaimeson et al. 2002; Tyrrel and Quinton 2003). Alternatively, it is possible that viable pathogens could be present on the crop surface following biosolid application, or may become internalised within the crop tissue where they are protected from conventional sanitization (Itoh et al. 1998; Solomon et al. 2002). In this case, a person may become infected if they consume the contaminated produce. Therefore, it is critical to accurately determine the pathogen risk associated with land application of sewage sludge to fully understand the potential for environmental loss and consequently, human transmission.

However, survival patterns of sludge-derived pathogens in the environment are complex, and a lack of a standardised approach to pathogen measurement makes it difficult to quantify their impact. For example, Avery et al. (2005) spiked treated and untreated sludge samples with a known concentration of *E. coli* to quantify the time taken to achieve a decimal reduction. The pathogen response was variable and ranged from 3 to 22 days, depending on sludge properties. Lang and Smith (2007) investigated indigenous *E. coli* survival in dewatered, mesosphilic anaerobically digested (DMAD) sludge, and in different soil types post DMAD sludge application. Again, decimal reduction times proved variable, ranging from 100 days when applied to air-dried sandy loam, to 200 days in air-dried, silty clay. This time decreased to 20 days for both soil types when field moist soil was used, demonstrating the importance of water content in regulating survival behaviour. Therefore, in order to quantify pathogen risk in a relevant, site-specific manner, it is necessary to incorporate both soil and treated sewage sludge characteristics in risk assessment modelling. This has been done previously by conducting soil, sludge and animal slurry incubation studies, where pathogens are often spiked to generate a survival response (Vinten et al. 2004; Lang and Smith 2007; Moynihan et al. 2013). Pathogen decay rate is then calculated based on decimal reduction times, or a first-order exponential decay model previously described by Vinten et al. (2004), and has been shown to be highly contingent on soil type and sludge or slurry combinations. Currently, the Safe Sludge Matrix provides a legal framework for grazing animals and harvesting crops following landspreading of treated sewage sludge, and stipulates that a time interval of three weeks and 10 months should be enforced to ensure safe practice, respectively (ADAS 2001). However, further work is required to determine if these regulations are overly stringent, particularly in light of the comparatively higher pathogen concentrations reported for animal manures and slurries. For example, *E. coli* concentrations ranged from $3x10^{2}$ to $6x10^{4}$ CFU g^{-1} in sludge (Payment et al. 2001), compared to $2.6x10^{8}$ to $7.5x10^{4}$ CFU g^{-1} in fresh and stored cattle slurry, respectively (Hutchison et al. 2004). Therefore, environmental losses associated with treated sewage sludge application may not be as extensive as previously thought, and further comparisons on pathogen risk should form the basis of future research.

X.4.4 Pharmaceuticals

Pharmaceuticals comprise a diverse collection of thousands of chemical substances, including prescription and over-the-counter therapeutic drugs and veterinary drugs (USEPA 2012). Pharmaceuticals are specifically designed to alter both biochemical and physiological functions of biological systems in humans and animals (Walters et al. 2010). Pharmaceuticals are referred to as ‘pseudo-persistent’ contaminants (i.e. high transformation/removal rates are compensated by their continuous introduction into the environment) (Barceló and Petrovic 2007). Pharmaceuticals are likely to be found in any body of water influenced by raw or treated waste water, including river, lakes, streams and groundwater, many of which are used as a drinking water source (Yang et al. 2011). Between 30 and 90% of an administered dose of many pharmaceuticals ingested by humans is excreted in the urine as the active substance (Cooper et al. 2008). In a survey conducted by the US Environmental Agency (see McClellan and Halden 2010), the mean concentration of 72 pharmaceuticals and personal care products were determined in 110 treated sewage sludge samples. Composite samples of archived treated sewage sludge, collected at 94 U.S. wastewater treatment plants from 32 states and the District of Columbia were analysed by liquid chromatography tandem mass spectrometry using EPA Method 1694. The two most abundant contaminants found in the survey were the disinfectants triclocarban and triclosan. The second most abundant class of pharmaceuticals found were antibiotics, particularly Ciprofloxain, Ofloxacin, 4-epitetra-cycline, tetracycline, minocycline, doxycycline and azithromycin (McClellan and Halden 2010). It was concluded that the recycling of treated sewage sludge was a mechanism for the release of pharmaceuticals in the environment.

Pharmaceuticals have received increasing attention by the scientific community in recent years, due to the frequent occurrence in the environment and associated health risks (Chen et al. 2013). In 2007, the European Medicines Agency (EMEA) issued a guidance document (ERApharm) on environmental risk assessment of human medicinal products. It relies on the risk quotient approach used in the EU and is also used for industrial chemicals and biocides where the predicted environmental concentration is compared to the predicted no-effect concentration. The overall objective of ERApharm is to improve and complement existing knowledge and procedures for environmental risk of human and veterinary pharmaceuticals. The project covers fate and exposure assessment, effects assessment and environmental risk assessment (Lienert et al. 2007). A considerable amount of work focused on three case studies. Two of the case studies focused on human pharmaceuticals, β-blocker atenolol and the anti-depressant fluoxetine, and the third on a veterinary parasiticide ivermectin. Atenolol did not reveal any unacceptable risk to the environment but cannot be representative for other β-blockers, some of which show significantly different physiochemical characteristics and varying toxicological profiles in mammalian studies (Knacker and Metcalfe 2010). Although found in trace levels (several nanograms per litre), some therapeutic compounds such as synthetic sex hormones and antibiotics, have been found to cause adverse effects on aquatic organisms (Chen et al. 2013). Therefore, understanding their environmental behaviour and impact has recently become a topic of interest for many researchers.

X.5 QUANTIFICATION OF COSTS AND BENEFITS FROM RE-USE OF SEWAGE SLUDGE

The main pathways for the disposal of sewage sludge in Europe is re-use in agriculture, landfill and incineration. The implementation of the Landfill Directive means that in the coming years, re-use in agriculture or incineration will become common pathways. In countries that preclude the re-use of treated sewage sludge in agriculture, incineration or alternative disposal methods, such as pyrolysis (used in the creation of biochar), the creation of engineering products (e.g. building materials; Hytiris et al., 2004), or reuse in power stations, may be alternative options. Landspreading is estimated to be the most cost-effective means of disposal of treated sewage sludge (Table X.3); however, this does not take into account factors such as legislative requirements, potential savings to the farmer through the use of a low-cost fertiliser, or environmental benefits (or drawbacks) accruing from its use.

Depending on the type of treatment applied, costs associated with the re-use of sewage sludge may include, amongst other issues, drying, lime amendment, thermal drying costs, along with costs of installation of storage facilities in which to carry out these treatments; labour, energy and transport costs; and where the treated sewage sludge is re-used on land, soil and sewage sludge analysis costs and other professional service costs (Table X.3). Potential benefits accruing from the land application of treated sewage sludge may be enhanced nutrient availability to crops and enhanced crop yield, and in countries where sewage sludge, treated or untreated, is considered a waste material (e.g. Ireland), there is a substantial saving for the farmer.

X.5.1 Impact of nutrient recovery, energy/product generation on energy and cost savings in a sewage treatment plant

It is well known that the potential energy available in the raw wastewater influent significantly exceeds the electricity requirements of the treatment processes. Energy captured in organics entering the plant can be related to the chemical oxygen demand load of the influent flow. Based on calorific measurements, a capita-specific energy input of 1760 KJ per population equivalent (PE) in terms of 120 g chemical oxygen demand of organic matter can be calculated (Wett et al. 2007). This specific organic load is subjected to aerobic and anaerobic degradation processes, partly releasing the captured energy. Traditional wastewater treatment plants (WWTP) have unusually high energy demands and create problems associated with the disposal of sewage sludge and chemical residues. It is estimated that wastewater treatment accounts for about 3-5% of the electrical energy load in many developed and developing countries (Chen and Chen 2013). Kapshe et al. (2013) demonstrated how energy generation in four WWTPs in India can utilize the methane recovery through anaerobic digestion to produce 1.5 to 2.5 million kWh electricity for captive use every year. An additional benefit is the reduction of 80,000 tonnes of CO_2 emission per year.

Dewatered sludge (15-35% D.S.) has a very low Lower Heating Value (LHV), so its use in energy recovery or incineration is not currently feasible. Dried sludge (about 70-75% D.S.), however, may be a valuable energy source, if mixed with fuels (e.g. natural gas) and/or other waste with a high calorific value (e.g. Residue Derived Fuels, RDF), as its LHV may reach up to 16 MJ / kg, allowing its use as a secondary fuel in, for example, the cement industry. The reader is referred to Tsagarakis and Papadogiannis (2006) for further information on energy recovery from sewage sludge in a treatment plant in Greece.

Within Germany, 344 WWTPs in North Rhine Westphalia (NRW) have undergone energy analysis (Wett et al. 2007) comprising two stages: a first stage, where operational data are collected and energy consumption rates and biogas yields are targeted; and a second stage, where optimization measures are adopted. By application of this protocol, energy costs can be reduced. Through the re-use of energy produced during wastewater treatment, the long-term sustainability of the WWTPs is enhanced, while also contributing to offset installation and on-going operational costs.

In Southern European countries, including the Mediterranean area, cultural, social and economic reasons means that the management of the sewage sludge is not necessarily the same as in other EU countries. Here, recycling to agriculture is the main route for final disposal. For example, in Portugal and Spain about 50% of the sewage sludge is recycled in agriculture (Milieu et al., 2013a, 2013b, 2013c). Therefore, sewage sludge management in these countries should be governed by the following objectives (Martins and Béraud, pers. comm.): (1) provision of solutions that are technically and economically adapted to the economic realities of these countries (lower investment and operating costs); (2) full legal compliance, including the ability to adapt to future restrictions, which may be placed on the disposal of treated sludge in agriculture; (3) diversification of the final disposal of sludge with new sludge treatment systems; (4) reduction in the quantity of sewage sludge to be disposed of; (5) optimization of the utilisation of weather conditions for sludge treatment, which makes solar drying an appealing solution.

Table X.3 Some treatment and disposal routes for sewage sludge, capital and operating costs, and benefits and drawbacks (adapted from RPA, Milieu Ltd. And WRc., 2008; Fytili and Zabaniotou, 2008; Astals et al., 2012; Cao and Pawłowski, 2012).

Treatment/disposal route for sludge	Costs			Benefits	Drawbacks
	Capital	Operating	Overall costs € per ton DM		
On-site treatment					
Thermal drying			90-160		
Anaerobic digestion			90-160	Biogas produced has a high calorific value (15.9-27.8 MJ m^{-3})	Elevated heating requirements to heat digester, odour potential.
Lime stabilisation			90-160		
Composting			90-160		
Solar drying	Land acquisition and construction	Labour	30 – 70	Low investment and operation costs. Final product is useful for industrial valorisation. Sewage volume is reduced.	It depends on sunlight/air temperature. Large areas are required for the greenhouses. Odour emissions.
Landfill	Land acquisition and construction	Labour Vehicle fuel Electricity Landfill tax and gate fees	309	Energy production from gas capture	Leachate production GHG emission (may be reduced in capture) Noise, odour, dust generation
Re-use in agriculture		Labour Regulatory	126 – 280[1]	Potential yield improvement Less reliance on chemical	Potential application of emerging contaminants to soil.

(landspreading)		testing of soil		fertiliser	Potential for leaching, runoff and volatilisation. Potential for introduction of contaminants into food chain.
Thermal (incl. incineration, wet oxidation, gasification and pyrolysis)	Land acquisition and construction	Labour Transport to site Quality control	332 – 411[2]	Energy production (but less than is used within the process) Large reduction in sludge volume. Thermal destruction of toxic compounds. Pyrolysis can be used to maximize production of chars.	Emissions to air, soil, water. Noise, dust generation. Visual intrusion. Possible impact on human health. Incomplete disposal – 30% of solids remains as ash. In pyrolysis, majority of energy consumption is used to reduce sludge moisture content.
Forestry and silviculture		Labour Regulatory testing of soil	210 – 250[3]	Increased tree growth Nutrient input to soil	Leaching of nutrients to groundwater. Impact on ecosystems.

[1] About €40 ton^{-1} DM in Portugal (Martins and Béraud, pers. comm.) [2] Cost for incineration (RPA, Milieu Ltd. And WRc., 2008) [3] From Anderson and SEDE (2002)

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