International Biodeterioration & Biodegradation 63 (2009) 981–987

Contents lists available at ScienceDirect

International Biodeterioration & Biodegradation

journal homepage: www.elsevier.com/locate/ibiod

Review

Fate of linear alkylbenzene sulfonates in the environment: A review

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ARTICLE INFO

Article history:
Received 28 June 2008
Received in revised form
31 March 2009
Accepted 31 March 2009
Available online 22 September 2009

Keywords:
Linear alkylbenzene sulfonates (LAS)
Biodegradation
Removal of surfactants
Sludges
Environmental risk

ABSTRACT

The fate and risk of linear alkylbenzene sulfonates (LAS) in various compartments of the environment have been reviewed. Under aerobic conditions LAS degrade rapidly and concentration of LAS has been found very low in effluents from aerobic sewage treatment plants (STP). On the contrary, in anaerobic STPs effluents, LAS concentrations have been found very high. Anaerobic effluents containing high LAS concentrations have been found to pose risk to aquatic environment. Similarly, LAS concentrations have been found high in anaerobically treated sewage sludge's. LAS enter the soil as a result of sludge application to the land. Risk to aquatic and terrestrial ecosystems is increased when wastewaters and sludge's are treated anaerobically.

Contents

1. Introduction

Surfactants are a diverse group of chemicals that are designed to have cleaning or solubilization properties. They generally consist of a polar head group (either charged or uncharged), which is well solvated in water, and a nonpolar hydrocarbon tail, which is not easily dissolved in water. Hence, surfactants combine hydrophobic and hydrophilic properties in one molecule. Synthetic surfactants are economically important chemicals. They are widely used in household cleaning detergents, personal care products and industries like textiles, paints, polymers, pesticide formulations, pharmaceuticals, mining, oil recovery and pulp and paper (Ying, 2006).

Surfactants are mainly of four types: anionic, nonionic cationic and amphoteric. Linear alkylbenzene sulfonates (LAS), alkyl ethoxy sulphates, alkylphenol ethoxylates, and quaternary ammonium compounds are the commonly used commercial surfactants. LAS are the most widely used synthetic anionic surfactants. They have been extensively used for over 40 years with an estimated global consumption of 18.2 million tonnes in 2003 compared to 9, 4.5, 1.7, 0.5, 0.1, and 2.4 million tonnes of soap, anionic, nonionic, cationic, amphoteric, and other surfactants respectively (Hauthal, 2004).

However, surfactants may have a negative impact on the environment during their life-cycle. Typically, production, formulation,

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doi:10.1016/j.ibiod.2009.03.012

the use phase, and the discharge phase are counted among the life-cycle. In the first two phases, i.e. production and formulation, the amount of chemical released or emitted into the environment is usually low since these processes take place in chemical plants equipped with appropriate installations for a safe handling. In the use phase and the discharge phase the differences can be considerable. The exact percentage of the chemical released into the environment depends on the physico-chemical and biological properties of the chemical and the way it is used and disposed off.

Anionic surfactants (AS) especially LAS deserve special interest since they (i) impact on aquatic/terrestrial ecosystems and (ii) are used in large quantities in consumer products, which are discharged into the wastewaters after use. Venhuis and Mehrvar (2004) have reported that 0.02–1.0 mg l^{-1} LAS in aquatic environment can damage fish gills, cause excess mucus secretion, decrease respiration in the common goby, and damage swimming patterns in blue mussel larva and 40–60 mg LAS kg^{-1} dry wt. of sludge interfere with the reproduction and growth of soil invertebrates and earthworms. Venhuis and Mehrvar (2004) reported acute effects of LAS on freshwater plankton and organisms (including bacteria to crustaceans) under field conditions. It was observed that LAS have a negative impact on the survival of heterotrophic nanoflagellates and ciliates at very low concentrations. Fairchild et al. (1993) have also found that LAS concentration of 0.36 mg l^{-1} had no effect on biological population. van de Plassche et al. (1999) reported no-effect concentration of 0.25 mg l^{-1} for LAS to aquatic organisms. Belanger (1994) reviewed available model ecosystem tests on LAS in a case study prepared for the European Workshop on Freshwater Field Tests. No-observed-effect levels varied from 1.1 to 27 mg l^{-1} using a vast array of model types including simplified food chains, lentic exposures, and more physically complex stream mesocosms. Belanger et al. (2002) estimated a no-observed-effect concentration of 0.29 mg l^{-1} which was based on a broad array of organisms that responded in similar time frames and concentrations. The key for protecting the environment from the negative impact of down-the-drain-chemicals (surfactants) is the biological treatment of the sewage wastewater treatment plant (WWTP).

LAS are manufactured in large quantities, used by many people, and disposed from production and after household use into the environment. The vast majority of this waste stream is treated via domestic WWTPs which reduce significantly the load of chemical substances to the receiving surface waters. Sewage sludges after treatment are incorporation into soil as soil fertilizers (Fig. 1).

There is a need to know the concentration of LAS at each step of its life-cycle and the effect of the type of treatment process on the efficiency of LAS removal. This information is needed to construct fate models, to compare different STP processes, and to establish the assessment of the risk to the aquatic and terrestrial ecosystems. This paper reviews the fate of LAS in the environment i.e. (a) LAS concentrations in the sewage (b) removal of LAS at various WWTPs (c) LAS concentrations in different types of sewage sludges (d) their concentrations in sludge amended soils and in rivers with possible associated environmental risk to the aquatic and terrestrial organisms.

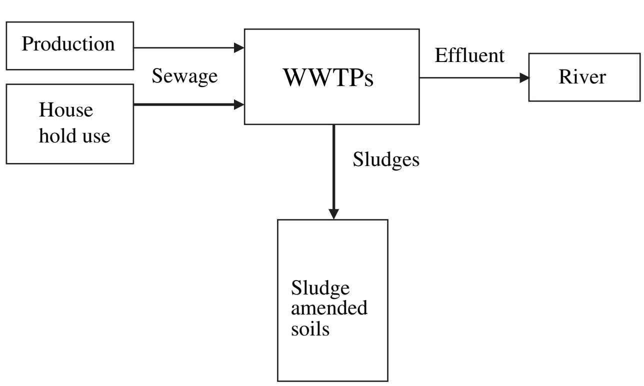

Fig. 1. Route of LAS in the environment.

2. Biodegradation of linear alkylbenzene sulfonates

Biodegradation is an important process to treat LAS in sewers, in sewage treatment plants, and it also enhances the removal of these surfactants in the natural environments, thus reducing their impact on biota. Microorganisms can either utilize surfactants as substrates for energy and nutrients or they can co-metabolize them through the initial reactions involved in catabolic pathways. There are many chemical and environmental factors that affect biodegradation of LAS in the environment. The most important influencing factors are chemical structure, and aerobic and anaerobic environments.

2.1. Aerobic degradation

Under aerobic condition, LAS co-metabolism generates shorter-chain homologues. LAS can also be mineralized to CO_2 and H_2O, but this normally requires the contribution of several species of bacteria.

Biological degradation of LAS under aerobic condition has been demonstrated by a vast number of investigations (Swisher, 1987; Cavalli et al., 1993; Leal et al., 1994; Karsa and Porter, 1995; Prats et al., 1997; Scott and Jones, 2000; Haggensen et al., 2002). Biodegradation of LAS is initiated with a ω-oxidation of the alkyl chain followed by successive cleavage of C_2 fragments (β-oxidation) (Fig. 2). The reaction occurring during ω- and β-oxidations generate sulpho phenyl carboxylates (SPCs) resulting in the loss of interfacial activity and toxicity (Kimerle and Swisher, 1977; Kimerle, 1989). SPCs aromatic ring cleavage then follows to achieve LAS mineralization (Karsa and Porter, 1995).

2.2. Anaerobic biodegradation

Degradation processes in anaerobic systems depends on alternative acceptors such as sulphate, nitrate or carbonate yielding, ultimately, hydrogen sulfide (H_2S), molecular nitrogen (N_2), methane (CH_4) and/or ammonia (NH_3). LAS mineralization under anoxic conditions has not been documented and the known enzymatic steps involved in aerobic mineralization, i.e. the ω- and β-oxidation and cleavage of the benzene ring, require molecular oxygen (Federle and Schwab, 1992; Garcia et al., 2005; Gejlsbjerg et al., 2004).

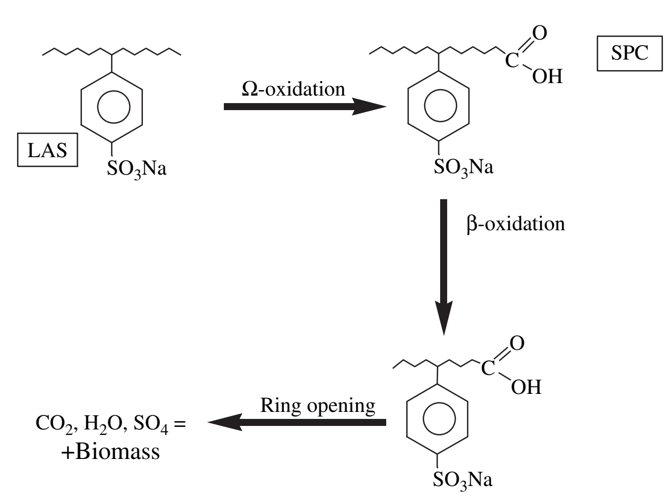

Fig. 2. Biodegradation pathway of LAS (Swisher, 1987).

LAS removal from synthetic wastewaters, under anaerobic conditions has been reported in few recent publications for bench-scale or pilot-scale experiments (Denger and Cook, 1999; Angelidaki et al., 2000; Almendariz et al., 2001; Haggensen et al., 2002; Mogensen and Ahring, 2002; Sanz et al., 2003; Angelidaki et al., 2004; Lobner et al., 2005). The limited information indicates that (a) LAS can be used as a source of sulfur by anaerobic bacteria under sulfur limited conditions (Denger and Cook, 1999), (b) benzenesulfonic acid and benzaldehyde may be produced as metabolites under thermophilic conditions (Mogensen and Ahring, 2002), (c) LAS can be degraded using NO_3^- as the electron acceptor in the acidogenic step of a two-stage UASB reactor (Almendariz et al., 2001), (d) degradation may occur if inoculum from aerobic environments is used (Angelidaki et al., 2000), (e) increased removal of LAS is possible if bioavailable fraction (soluble) of LAS is increased (Haggensen et al., 2002; Angelidaki et al., 2004), and (f) surfactant can be partially used as a carbon and energy source by anaerobic bacteria in the presence and absence of additional source of carbon (Sanz et al., 2003). An investigation by Sanz et al. (2003) indicated that LAS degraded more in a UASB reactor without co-substrate than with a co-substrate. This implies that some microorganisms utilize AS when they have no other option. Lobner et al. (2005) observed that anaerobic reactors operated under the same condition sometimes exhibit very different degradation capabilities. They obtained 40–80% removal of LAS in a bench-scale UASB reactors under mesophilic and thermophilic conditions and they concluded that it depends on the process stability.

Sewers contain microbial populations capable of initiating LAS biodegradation (Matthijs et al., 1995). LAS removal in sewers due to biodegradation can reach values of the order of 50% of the total LAS load when the sewer system is properly aerated (Moreno et al., 1990). However, mechanisms others than biodegradation are also involved in LAS removal, these include adsorption onto, and settling of, suspended solids and precipitation of calcium salts. Although it is expected that anaerobic conditions prevail constantly in certain part of the sewer, between 40 and 60% of the LAS load has been found to be removed/biodegraded in the sewers/or in the in-line storm tanks (Feijtel and Van de Plassche, 1995). According to HERA (2007), up to 68% of LAS was found to be removed in sewers through a combination of biodegradation, adsorption, and precipitation. Laboratory studies have demonstrated that the concentration of LAS in STP influents depends on the length of the sewer, travel time and the degree of microbial activity present in the sewers (Matthijs et al., 1999).

3. Fate of LAS in wastewater treatment plants

LAS concentrations in raw wastewater have been reported to range from 3 to 21 mg l^{-1} (Brunner et al., 1988; DeHanau et al., 1989; Ruiz Bevia et al., 1989; Holt et al., 1995; Adak et al., 2005). According to McAvoy et al. (1993), in USA, monitoring at 50 wastewater treatment facilities in eleven states showed average LAS levels in raw sewage ranging from 4.0 to 5.7 mg l^{-1}. LAS levels in raw sewage from five European countries ranged from 4.0 to 15.1 mg l^{-1} (DiCorcia et al., 1994; Waters and Feijtel (1995)). Although LAS have been found to be readily biodegradable by aerobic processes, much of the load into a sewage treatment facility (reportedly 20–50%) is associated with suspended solids (McAvoy et al., 1998) and thus escapes aerobic treatment processes (McEvoy and Giger, 1986; Swisher, 1987). In general, at STPs, aerobic, anaerobic, and facultative or combination of these are applied to treat the sewage. Conventional activated sludge process (ASP), trickling filters (TF), oxidation ponds (OP), upflow anaerobic sludge blanket (UASB) reactor, and sequencing batch reactors (SBR) are in general, the technologies which are used for the sewage treatment.

In conventional ASP based STPs, LAS removal has been found to be mostly in 95–99.9% range (Berna et al., 1989; Painter and Zabel, 1989; Waters et al., 1989; Cavalli et al., 1993; McAvoy et al., 1998; Matthijs et al., 1999; Jensen et al., 2007). The LAS removal in ASP measured in five European countries, averaged 99.2% (Waters and Feijtel, 1995) and 99.4% (range was 98.9–99.9%) (Holt et al., 2003). Results of a mass balance study at an activated sludge treatment plant indicated that removal is primarily due to biodegradation with only about 10–20% of the influent LAS retained in the sludge and about 1% released to surface waters (Berna et al., 1989; Painter and Zabel, 1989; Cavalli et al., 1993; DiCorcia et al., 1994).

In trickling filter based STPs, total removals were found to be lower and more variable than ASP. It was found in the range of 89.1–99.1% in Europe with an average removal of 95.9% (Holt et al., 2003), higher than that reported in USA, average removals of 83% (Trehy et al., 1996) and 77% (McAvoy et al., 1993). McAvoy et al. (1998) also concluded that compared to ASP (~99%) average removal was only around 82% in trickling filter based sewage treatment plants. According to OECD (2005), LAS were found to be removed by 98% for lagoons/oxidation ditches, and 96% for rotating biological contactors.

Removal of LAS calculated as methylene blue active substance (MBAS), was studied in five full scale UASB based STPs (27, 34, 38, 56, and 70 ML d^{-1} capacity) with polishing ponds (1–1.6 days retention) as a final step by Mungray and Kumar (2008). In UASB at four out of five STPs, average removals of anionic surfactants ranged from 2 to 18%. At fifth STP, AS were found to increase by around 8%. UASBR effluents contain substantial concentrations of AS (4.25–5.91 mg MBAS l^{-1}). Particulate fraction was reduced by around 45% while soluble fraction increases. In a combined UASB-PP, AS were reduced by only 30% and particulate AS were reduced by >80%. Gasi et al. (1991) also found that UASB effluent (120 m^3 cylindrical UASBR, Brazil) was rich in AS (concentrations between 4.63 and 5.30 mg l^{-1} of AS as MBAS). High levels of AS compared to ASPs were discharged from the conventional anaerobic STPs like UASB. This supports the hypothesis that LAS are not easily degraded under anaerobic conditions, although other investigations suggested the contrary (Swisher, 1987; Cavalli et al., 1993; Leal et al., 1994; Karsa and Porter, 1995; Prats et al., 1997; Haggensen et al., 2002; Scott and Jones, 2000). Concentrations of LAS in different type of treatment systems (ASP, TF, Lagoon, UASB and OP) are summarized in Table 1.

Environmental risk of LAS when it is discharged from different STPs after treatment has been estimated using the risk quotient (RQ) method (Mungray and Kumar, 2008). The risk assessment to aquatic and terrestrial environments due to presence of LAS in treated sewage and sludges was evaluated according to the procedure laid down in European Union Technical Guidance Document (EU-TGD, 2003). RQ is the ratio of the predicted environmental concentration (PEC) i.e. the measured average concentrations of LAS in final effluents from different STPs after using a dilution factor of 10 (TGD default dilution coefficient) to the predicted-no-effect concentration (PNEC) of chemicals i.e. the concentration below which unacceptable effects on organisms are not likely to occur. Therefore RQ = PEC/PNEC. If this ratio is less than 1, it tells that the environmental impact due to this chemical is less than the target value of the endpoint. As this value increases, the concern increases. PNEC can be based on the no-observed-effect concentration (NOEC) of 0.27 mg l^{-1} (Petersen et al., 2003; HERA, 2007) which has been experimentally established following long-term laboratory screening tests on broad array of freshwater plants/organisms at different trophic levels. An assessment factor of 10 has been suggested (EU-TGD, 2003) for the estimation of PNEC values for LAS. This yields lowest value of PNECs in receiving water of 0.027 mg l^{-1}.

Mungray and Kumar (2008) found that for liquid effluents from aerobic STPs, RQ values were below 1 whereas values above 1 were

Table 1
LAS in ASP, TF, Lagoon, UASB, and OP Based STPs.

Plant (no of plants studied)	Country	Influent (mg l^{-1})	Effluent (mg l^{-1})	Removal (%)	Reference
ASP	Switzerland	–	–	>99%	Brunner et al., 1988
ASP	Spain	–	0.2	>98	Berna et al., 1989
ASP	UK	–	0.05	95–99	Painter and Zabel, 1989
ASP	UK	–	–	99	Moreno et al., 1990
ASP (15)	USA	4.2–5.7	–	99.3	McAvoy et al., 1993
ASP	USA	4.6	0.068	98.5	DiCorcia et al., 1994
ASP	Netherlands	3.1–7.2	0.008	99.9	Feijtel et al., 1995
ASP	England	11.8–18.2	–	99.9	Holt et al., 1995
ASP (5)	Europe	–	0.009–0.140	98.5–99.9	Waters and Feijtel, 1995
ASP (4)	Europe	–	–	99.5	Trehy et al., 1996
ASP	Germany	–	–	99.7	Schroder, 1997
ASP (4)	UAS	–	–	>99	McAvoy et al., 1998
ASP (7)	Netherlands	3.4–8.9	0.019–0.07	98–99.6	Matthijs et al., 1999
ASP	Europe	3–21	0.008–0.27	99	HERA, 2007
ASP (6)	Dutch	1–15	0.039	99.1	Venhuis and Mehrvar, 2004
TF (12)	USA	–	–	77.4	McAvoy et al., 1993
TF (5)	USA	–	–	82.9	Trehy et al., 1996
TF(6)	USA	–	–	82	McAvoy et al., 1998
TF (4)	UK	–	0.04–0.43	92.9	Holt et al., 2000
TF	USA	–	–	89	HERA, 2007
Lagoon/oxidation ditch	USA	–	–	98–98.5	McAvoy et al., 1993
Lagoon	Spain	5.71 & 1.25	0.39 & 0.8	>97	Moreno et al., 1994
Lagoon	Italy	–	–	90	Marcomini et al., 1999
UASB	Brazil	–	4.63–5.30	–	Gasi et al., 1991
UASB (5)	India	5.16–6.01	4.25–5.91	2–18	Mungray and Kumar, 2008
UASB + PP (5)	India	5.16–6.01	3.60–4.91	8–30	Mungray and Kumar, 2008
OP (2)	India	5.57 & 6.22	0.67 & 3.31	88 & 47	Mungray and Kumar, 2008

obtained for discharges from anaerobic STPs. Therefore, for lowering down the RQ values of anaerobic STP effluents below 1, aerobic post treatment steps seem to be necessary.

Literature also support that not only dissolved fraction of LAS is biodegraded but the adsorbed fraction is also degraded after it is desorbed from the biomass. Cowan et al. (1993) developed a model assuming biodegradation of sorbed fraction (after desorption) with the dissolved fraction. Hand and Williams (1987) found that desorption of LAS from river sediment was rapid (equilibrium reached within 3–8 h) and nearly 100% reversible. Furthermore, Shimp and Young (1988) found that the availability of sorbed chemical for biodegradation is related to the mechanism of sorption with hydrophobic sorption being more reversible than the ion exchange. Due to the high organic carbon content of the mixed liquor suspended solids of 30–60% (Metcalf and Eddy, 2003) hydrophobic interactions are most probably the dominant sorption mechanism in activated sludge reactors.

4. Concentrations of LAS in waste sludges of STPs

Sewage sludge consists of 90–99% water and an accumulation of settleable solids, mainly organics that are removed during primary, secondary or advanced wastewater treatment processes but does not include grit and screenings (EPB, 2004). The amount of LAS present in the final sludge is highly dependent on the type of sludge produced and the biological processes running at the WWTP. Berna et al. (1991) have reported that primary sludge usually contains 10–20% of the LAS found in the raw sewage. Sludge when ready to be applied to the soil has usually undergone one or two anaerobic or aerobic digestion processes. This may be followed by one or several of the following processes: drying, centrifugation and composting, after which the sludge is designated as stabilized. Thermal treatment by raising the temperature to 70 °C is practiced in some sewage treatment plants to eliminate the pathogens. Effects of the temperature treatment on anionic surfactants have not been reported in the literature. The most important parameter in controlling the LAS content of final sludge is the aerobic conditions during digesting. LAS biodegradation under anaerobic methanogenic conditions has not yet been demonstrated (Schoberl, 1989; Birch et al., 1992; Painter and Mosey, 1992), whereas LAS are in general readily biodegradable under aerobic conditions, e.g. 95% primary degradation in the OECD screening test. Typically, LAS levels in aerobically digested sludges are found in the range of 100–500 mg kg^{-1} dry wt., whereas anaerobically digested sludges contain on an average 1000–30,000 mg LAS kg^{-1} dry wt. (McEvoy and Giger, 1986; DeHenau et al., 1986; Giger et al., 1987; Matthijs and DeHenau, 1987; Brunner et al., 1988; Jensen et al., 2007; Krogh et al., 2007; Schowanek et al., 2007). Mungray and Kumar (2008) analysed anaerobic sludges from five UASB based STPs. AS concentrations varied from 4480 to 9233 mg kg^{-1} dry wt. at all the five STPs.

There are several reasons for the elevated levels of LAS in anaerobically digested sludges (Schowanek et al., 2007): high usage volumes, sorption to primary sludge, precipitation as insoluble Mg/Ca-salts in the primary settler (K_d sludge = 1000–4000 l kg^{-1}) depending on chain length, absence of anaerobic degradation, and the solids concentration effect caused by the digestion process. Biodegradation resumes however after a few days if the anaerobic sludge is stored aerobically, composted, or applied to land. Concentrations of LAS in different types of sludges from a number of countries are summarized in Table 2.

Drying/stabilization on conventional sand drying beds (SDBs) are the most widely used method of sludge dewatering in developing countries like India. It ranges from few weeks to few months during which most of the matter contained in the sludge is eliminated and the product attains the right dryness for further use. The mechanisms/processes occurring on an SDB include thickening, dewatering, storage, stabilization, evaporation, aerobic/anaerobic biodegradation, thermal treatment, milling/tilling and aeration etc., all happening almost simultaneously. Advantages of SDBs are high solid content in the dried product, and low initial and operating costs. Disadvantages are dependence of climate changes on drying characteristics, labor-intensive sludge removal procedures, and insect and potential odor problems. Variations of the drying beds are: (a) conventional sand

Table 2
Concentrations of LAS in different types of sludges.

Sludge description	Country	No. of WWTP	LAS concentration (mg kg^{-1} dry wt.)	Reference
Aerobically digested	Germany	10	182–432	Matthijs and DeHenau, 1987
	Spain	2	100 & 500	Berna et al., 1989
	US	8	152	McAvoy et al., 1993
	–	–	205	Feijtel et al., 1995
	–	–	150	DeWolf and Feijitel, 1998
	–	–	100–500	Jensen et al., 2007
	Europe	–	<1000	Schowanek et al., 2007
Anaerobically digested	Switzerland	10	2900–11 900	McEvoy and Giger, 1986
	Germany	8	1600–11 800	DeHenau et al., 1986
	Germany	–	1330–9930	Matthijs and DeHenau, 1987
	–	–	5500	Matthijs and DeHenau, 1987
	Spain	5	7000–30 200	Berna et al., 1989
	US	5	3120–6200	Rapaport and Eckhoff, 1990
	UK	5	9300–18 800	Holt and Bernstein, 1992
	Italy	1	11 500–14 000	Cavalli et al., 1993
	US	–	10 460	McAvoy et al., 1993
	Spain	3	12 100–17 800	Prats et al., 1997
	–	–	6000	DiCorcia et al., 1994
	–	–	1000–16 100	Rasmussen, 1999
	US	28	5292–15 632	Jensen, 1999
	Europe	–	5600	Carlsen et al., 2002
	Europe	–	1000–30 000	Schowanek et al., 2007
	–	–	3000–30 000	Jensen et al., 2007
Aeration/settling	–	2	400–700	Berna et al., 1989
Anaerobic digestion	–	–	150	Painter and Zabel, 1989
+ Sand drying beds (SDBs)	Spain	5	4800	Berna et al., 1989
	Spain	1	5200	Prats et al., 1997
UASB wet sludges	India	5	4480–9233	Mungray and Kumar, 2008
UASB wet sludges + SDBs	India	5	336–5880	Mungray and Kumar, 2008

drying beds (b) paved (c) wire-wedge, and (d) vacuum assisted beds (Metcalf and Eddy, 2003). Other dewatering methods using filter press, vacuum filter and centrifuge are considered as mechanical and the solid content of mechanically dewatered sludge is higher than that of sludge drying beds (EPB, 2004). Fate of LAS on SDBs has been studied only by a few researchers. Berna et al. (1989) reported reduction of LAS concentration to 4.8 g kg^{-1} from 30.2 g kg^{-1} by drying and milling operation. Painter and Zabel (1989) also reported a considerable reduction of LAS in anaerobic sludge on open beds from 6.64 g LAS kg^{-1} to 0.15 g LAS kg^{-1}. Mungray and Kumar (2008) measured dried sludges for AS from five UASB based STPs. Drying was carried out on sand drying beds for a period of almost two to three months. The AS concentrations in dried-stabilized sludges were found to range widely from 336 to 5880 mg kg^{-1} dry wt. with an overall average of 1452 mg kg^{-1} dry wt. Drying on SDB on an average resulted in an overall AS reduction of around 80%.

The use of sewage sludge as organic fertilizer on arable land is the most important source of LAS in terrestrial environments. After sludge amendment, aerobic conditions normally prevail and biodegradation of LAS occurs. Degradation of LAS is a result of microbial activity and hence is influenced by temperature, water content, availability of oxygen and the amount of LAS supplied as well as the chain length and position of the benzene ring (Jensen, 1999). Sorption of LAS to solids and organic matter are other important parameters controlling the degradation of LAS in the soil compartment. It can be found in elevated concentrations in soil immediately after sludge amendment, but a half-life of approximately 1–3 weeks prevents accumulation in soil and biota (Jensen, 1999).

Risks to terrestrial environment are associated to elevated concentration of LAS in sludge. Use of sewage sludge as a soil conditioner can be harmful to soil quality because of the adverse effect of LAS to soil invertebrates and microorganisms (Venhuis and Mehrvar, 2004; Schowanek et al., 2007). Risks to terrestrial environment that are associated to sludge depend upon (i) type of treatment given to sewage sludge before application to fields (i.e. aerobic or anaerobic digestion) (ii) post treatment to sludges in addition to aerobic or anaerobic digestion and (iii) number of application of sludges to agricultural fields per year. RQ values were calculated for sludge samples from various treatment facilities that were applied to different types of agricultural soils (Mungray and Kumar, 2008). The RQ values were based on a PNEC value of 4.6 mg kg^{-1} dry wt. that has been suggested by Gejlsbierg et al. (2001), Jensen et al. (2001), and HERA (2007). RQ values are found greater then 1 in only one case for a sludge sample that was digested anaerobically. There was no risk for any type of soil for sludge samples obtained from aerobic digestion or samples prepared by anaerobic digestion followed by an aerobic post treatment step like drying on sand drying beds. However, Jensen et al. (2001) have observed that RQ values may increase to more than one for soil receiving several successive applications of dried sewage sludges.

5. Linear alkylbenzene sulfonates in rivers

Effluents after the treatment are discharged into the rivers. Many studies (McAvoy et al., 1993; DiCorcia et al., 1994; Feijtel et al., 1995; Trehy et al., 1996; Schroder, 1997; Gandolfie et al., 2000; Mungray and Kumar, 2008) were conducted for finding the LAS concentration in river water. The final concentrations of LAS in river water depend upon the percent removal at the STPs, sorption, solids settling and in stream degradation as well (both in sediment and overlying water) (Takada et al., 1994; Lee et al., 1995) which are summarized in Table 3. All the aerobic STPs like ASP remove >99% of LAS and finally very low concentration of LAS is discharged in rivers.

Table 3
Concentration of LAS in rivers for different type of STPs.

STPs (% Removal)	Concentration of LAS in rivers (μg l^{-1})	Reference
ASP (99.3)	50	McAvoy et al., 1993
ASP (99.5)	2–81	Trehy et al., 1996
ASP	9.7	DiCorcia et al., 1994
ASP (99.2)	<2.1–47	Waters and Feijtel, 1995
ASP	14.2	Feijtel et al., 1995
ASP	28	Gandolfie et al., 2000
ASP	<4 - 81	Nishiyama et al., 2003
ASP	3	Mungray and Kumar, 2008
ASP (99)	<6	Schroder, 1997
TF	50	McAvoy et al., 1993
TF (82.9)	4–94	Trehy et al., 1996
Lagoon/oxidation ditch (98.5)	42	McAvoy et al., 1993
RBC (96.2)	46	McAvoy et al., 1993
UASB-PP (8–30)	360–490	Mungray and Kumar, 2008
OP (47 & 88)	310 & 60	Mungray and Kumar, 2008

McAvoy et al. (1993) monitored LAS concentration in ASP, Lagoon/oxidation ditch, rotating biological contactor and trickling filters. Concentrations of LAS below the mixing zone of WWTPs were generally below 50 μg l^{-1}, even though the samples were collected under low flow (i.e. low dilutions) conditions. The mean surface water concentrations ranged from <10 to 330 μg l^{-1}, with the mean values of 42–46 μg l^{-1}. The highest concentration was observed in a low (less than 3 fold) dilution irrigation canal below a trickling filter plant. All other values were <180 μg l^{-1}, with more than 80% of the sites below 50 μg l^{-1}. Schroder (1997) determined LAS concentrations under realistic worst-case conditions (i.e. low dilution rates) in section of Rur River affected by the discharge of treated wastewater. The results indicate that elimination of LAS in the Duren and Monschau treatment works exceeds 99%, resulting in a rather low effluent value of <6 μg l^{-1}.

Mungray and Kumar (2008) measured LAS concentrations (as MBAS) in five upflow anaerobic sludge blanket-polishing ponds (UASB-PP) and two oxidation ponds (OP) based STPs in India. Effluents from five STPs i.e. 27, 34, 38 56 and 70 ML d^{-1} capacity are discharged to river Hindon, a tributary of river Yamuna. AS concentrations were found to be in the range from 3.6 to 4.9 mg l^{-1} in five UASB-PP based STPs and 3.31 mg l^{-1} and 0.61 mg l^{-1} in two oxidation ponds based STPs. AS concentrations were predicted in river Hindon and it was found from 0.36 mg l^{-1} to 0.49 mg l^{-1} from five UASB-PP based STPs and 0.31 mg l^{-1} and 0.06 mg l^{-1} from OP based STPs which is very high compared to the concentration of AS in Rivers after discharging the effluents from ASP based STPs (Table 3). Based on these values, a risk was calculated to aquatic organism in river and it is concluded that the effluents which are discharged from UASB-PP based STPs can create a substantial risk.

6. Conclusion

Surfactants are important chemicals whose consumption is upraising because of the increasing living standard. After cleaning operations almost all the surfactants are discharged to the sewers. Surfactants enter the environment through the discharge of sewage effluents in to surface waters and through the application of the sewage sludge on land as soil conditioner. Surfactants are found in sewage, STPs, rivers, sludges, and sludge amended soils. LAS concentrations depend upon the type of treatment i.e. aerobic or anaerobic involved at STPs. Almost negligible concentrations of LAS are found in all the compartments of the environment when aerobic processes is used but when anaerobic processes like UASBs are used, substantial concentration of LAS is found, which can increase the risk to aquatic and terrestrial environments. Many advantages favor the use of anaerobic based STPs like UASBs over aerobic based STPs. Therefore, efforts need to be invested in order to improve anaerobic STPs post treatments for LAS removal.

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