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MINI-REVIEW

Degradation of triclosan by environmental microbial consortia and by axenic cultures of microorganisms with concerns to wastewater treatment

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Abstract

Triclosan is an antimicrobial agent, which is widely used in personal care products including toothpaste, soaps, deodorants, plastics, and cosmetics. Widespread use of triclosan has resulted in its release into wastewater, surface water, and soils and has received considerable attention in the recent years. It has been reported that triclosan is detected in various environmental compartments. Toxicity studies have suggested its potential environmental impacts, especially to aquatic ecosystems. To date, removal of triclosan has attracted rising attention and biodegradation of triclosan in different systems, such as axenic cultures of microorganisms, full-scale WWTPs, activated sludge, sludge treatment systems, sludge-amended soils, and sediments has been described. In this study, an extensive literature survey was undertaken, to present the current knowledge of the biodegradation behavior of triclosan and highlights the removal and transformation processes to help understand and predict the environmental fate of triclosan. Experiments at from lab-scale to full-scale field studies are shown and discussed.

Keywords Triclosan · Biodegradation · Transformation products · Wastewater treatment

Introduction

Triclosan (2,4,4′-trichloro-2’hydroxydiphenylether) (Table 1) is a synthetic, lipid-soluble, broad spectrum antimicrobial agent which was first introduced in the health care industry in 1972, and since 1985, it has been added to toothpaste in the USA and Europe (Jones et al. 2000). Currently, triclosan is used in 140 different types of consumer products including liquid hand soap, shower gels, hand lotions, toothpaste, mouthwashes, deodorants, as well as in textiles and polymeric products such as, mattresses, toothbrushes, kitchenware and plastic food containers, shoes, clothing, and children’s toys (Levy et al. 1999; Fiss et al. 2007). Triclosan has also been used in hospitals and medical products to control bacteria growth and the spread of disease (Bhargava and Leonard 1996). A registration as a biocide according the biocidal product directive was, however, not approved (EU 2012; EU 2016). Opposite to that, it still can (and is) used under the cosmetics regulation in the EU (EU 2009). In the European Union (EU), about 85% of the total amount of triclosan is used in personal care products, compared to 5% in textiles and 10% in plastics and food contact materials (SCCP/1192/08 2009).

According to its use, release of triclosan into the environment is mostly connected to wastewater which has also evoked a great concern for its environmental fate (Fig. 1). Triclosan has been widely detected not only in wastewater, surface water, and seawater (Sabaliunas et al. 2003; Sanchez-Prado et al. 2006; Aranami and Readman 2007; Jahangiri et al. 2017; van Wijnen et al. 2018), but also in sediments (Morrall et al. 2004; Dang et al. 2018), even in human milk (Adolfsson-Erici et al. 2002; Allmyr et al. 2006; Dayan 2007; Lu et al. 2018) and human tissue samples (Wang et al. 2015a, b; Lu et al. 2018). Triclosan inhibits the enoyl-acyl carrier reductase of sensitive bacteria, blocking the lipid synthesis and inhibiting microbial growth, which is answered by some microorganisms with formation of resistant strains

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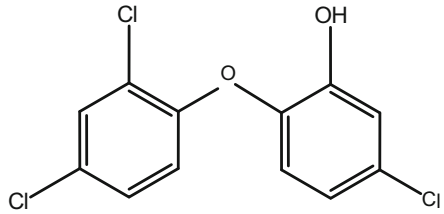
Table 1 Physico-chemical parameters of triclosan (MS Search v2.0; EPI Suit 4.0)

Molecular structure	
Chemical name	2,4,4′-trichloro, 2′-hydroxy-diphenylether
Chemical abstracts service registry (CAS) number	3380–34-5
Boiling point	363.62 °C
Melting point	136.79 °C
Synonyms	5-chloro-2-(2,4-dichlorophenoxy)phenol
Molecular formula	$C_{12}H_7Cl_3O_2$
Molecular weight	290 g/mol
Water solubility	4621 μg/L
log Kow	4.2–4.76
log Koc	4.265
pKa	7.9

(Dhillon et al. 2015; Ding et al. 2018). Wilson et al. (2003); Franz et al. (2008); Ding et al. (2018) found that triclosan may have long-term adverse effects on aquatic organisms in the environment even at low (i.e., ng/L) concentrations.

In the last few years, investigations have addressed different ways to remove triclosan from the environment. Triclosan can be removed, e.g., by biodegradation (Kim et al. 2011), photoelectrocatalytisis (Wong-Wah-Chung et al. 2007), ozonation (Chen et al. 2012) electro-fenton degradation (Munoz et al. 2012), and sonoelectrochemistry (Sanchez-Prado et al. 2008).

Removal of triclosan from wastewater is mostly due to biodegradation and sorption to sludge. However, degradation intermediates originated from biodegradation processes have been reported (Chen et al. 2015a). However, limited information related to the biodegradation intermediates is currently available.

This review summarizes biodegradation of triclosan in different systems, including axenic culture of microorganisms, lab-scale activated sludge simulations, full-scale wastewater treatment plants (WWTPs), sludge treatment system (reed beds, composting), sludge-amended soil, and sediments. The objective of this review is to provide a comprehensive overview on the respective processes relating to biodegradation, removal efficiencies, and possible transformation products in triclosan biodegradation.

Biological transformation of triclosan (metabolism and metabolites)

Biological transformation reactions of triclosan have been described and transformation products are formed by six different reactions (Fig. 2). Generally, adduct formation is observed as well as oxidative attacks and bond cleavage (catabolism).

Methylation is a reaction found for several phenols (including triclosan) in conventional activated sludge (CAS)-WWTPs (Bester 2005; Chen et al. 2011; Waria et al. 2011; Sadef et al. 2014a; Chen et al. 2015a). In classical wastewater treatment, triclosan-methyl was the most often reported transformation product. It has a log k_{ow} 5.0 and thus a higher potential for bioaccumulation than its parent triclosan (Bester 2003, 2005; Balmer et al. 2004; Waria et al. 2011). The amount of triclosan-methyl accounted for less than 5% of the incoming triclosan (Bester 2005; Chen et al. 2011; Huang et al. 2014).

Conjugation reactions It has been reported that the OH-group of triclosan can get conjugated by microorganisms by substituting one hydrogen atom with a sulfate group (Chen et al. 2015a). Conjugation reactions are well known for higher organisms such as vertebrates and including plants (Macherius et al. 2012). For plants especially glucuronation is described (Macherius et al. 2012), but less is known about this process in microorganisms. In addition, fungi have been reported to conjugate triclosan via glycosylation, and sulfatation products have been reported in activated sludge (Hundt et al. 2000).

Hydroxylation of aromatic rings during the biodegradation of triclosan is a typical oxidation reaction for aromatic systems, and biodegradation products such as monohydroxy-triclosan and dihydroxy-triclosan have been reported (Kim et al. 2011; Lee et al. 2012; Chen et al. 2015a; Kagle et al. 2015; Lee and Chu 2013) with the the structural formula of one isomer being elucidated by Chen et al. 2015a. Veetil et al.

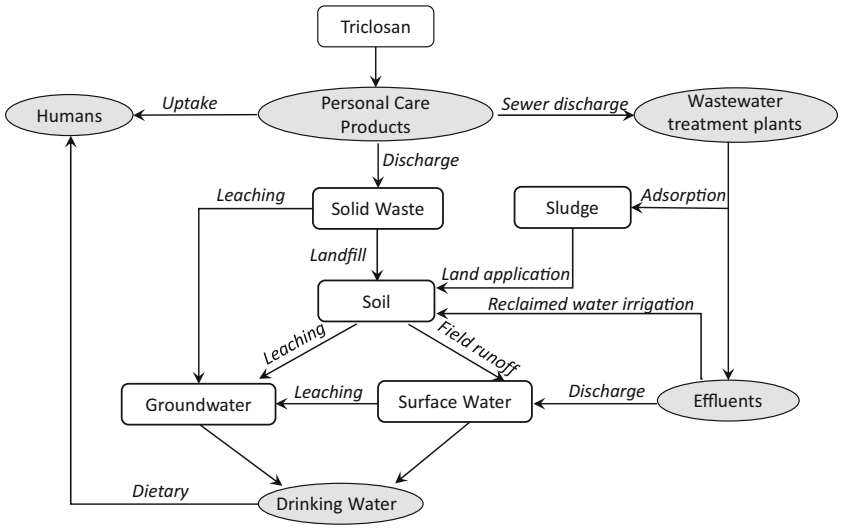


Fig. 1 Schematic diagram of the potential translocation pathways and behaviors of triclosan in the environment

(2012) isolated 77 bacterial strains tolerating triclosan, and found that the isolated pure strains and enrichment consortium were able to metabolize triclosan in the presence of monooxygenase inhibitor 1-pentyne, which indicated a probable involvement of a dioxygenase. Kagle et al. (2015) investigated that the initial triclosan degradation in RD1 was catalyzed by an inducible class IA aromatic dioxygenase in which *tcs*A and *tcs*B were suggested to be an inducible multicomponent triclosan oxygenase. Moreover, in the presence of an inhibitor of meta-cleavage enzyme (*3-fluorocatechol*), no triclosan degradation was observed, suggesting that triclosan degradation proceeds via dioxygenase oxidation and followed by ring opening (meta-cleavage) (Lee et al. 2012).

Lee et al. (2012) proposed an initial attack of regioselective dioxygenase at the 2,3-position of triclosan, which has resulted in the formation of hydroxylated triclosan by wastewater isolated *Sphingopyxis* strain KCY1. Whether the formation of monohydroxy triclosan is due to other enzymes or a back reaction of the respective dihydroxy triclosan isomers is not known at the moment. The formation of quinones occurs but rather seem to be typical for abiotic systems (Zhang and Huang 2003). Establishing full mass balances on this is

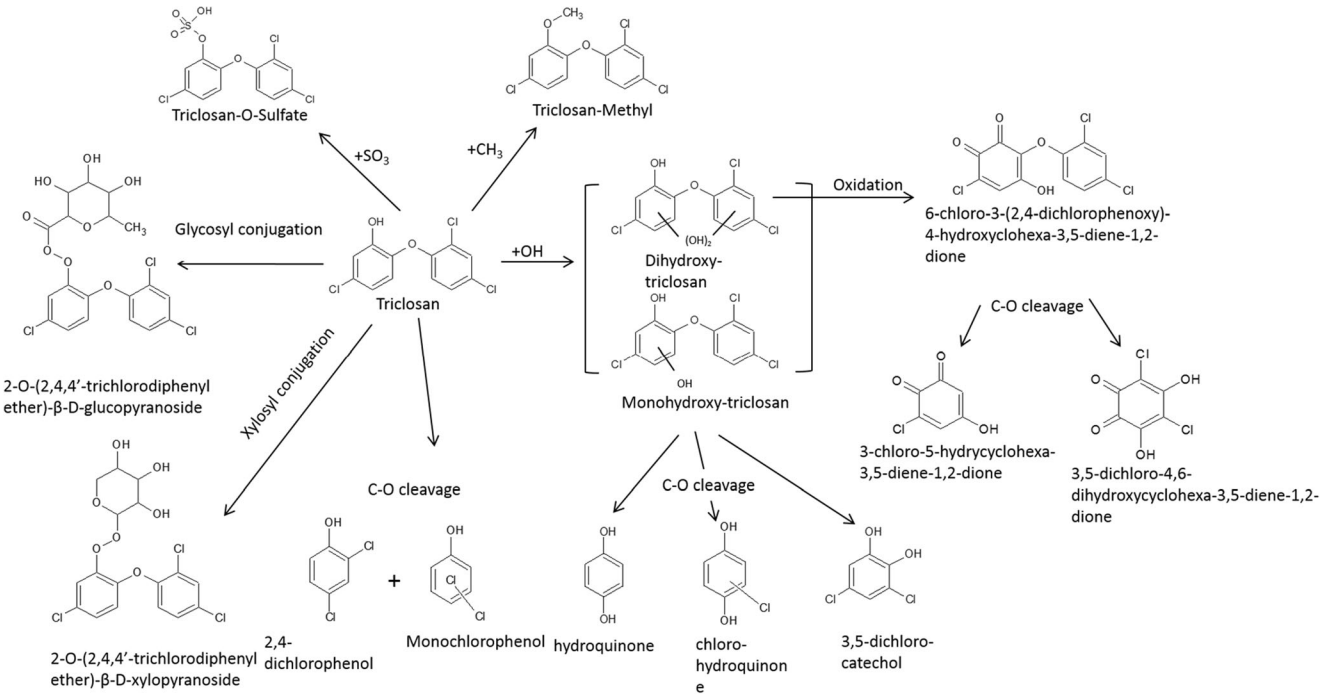


Fig. 2 Possible biodegradation pathway of triclosan (Hundt et al. 2000; Kim et al. 2011; Lee et al. 2012; Veetil et al. 2012; Lee and Chu 2013; Chen et al. 2015a; Kagle et al. 2015; Mulla et al. 2016; Orhon et al. 2017; Armstrong et al. 2018)

hindered as the respective true compounds are usually not available and several isomers are possible.

Ether-bond cleavage to form chlorinated phenols has been described by Lee et al. 2012; Veetil et al. 2012; Chen et al. 2015a; Mulla et al. 2016. Veetil et al. (2012) have found that triclosan can be biodegraded under aerobic, anaerobic, and anoxic conditions and phenol, catechol and 2,4-dichlorophenol were among the products. Ether bond cleavage of mono and dihydroxylated triclosans could also produce 2,4-dichlorophenol by *Sphingopyxis* strain KCY1 under aerobic conditions (Lee et al. 2012). 4-chloro- and 2,4-dichloro-phenol appeared when mono and dihydroxylated triclosans were detected during biodegradation, indicating that bond cleavage occurred on both sides of the ether bond of triclosan (Chen et al. 2015a). It is, however, unknown whether a selected ether bond cleavage exists, or whether the observed metabolites are result of a degradation/mineralization of the other ring.

Oligomerization was initially reported by Cabana et al. (2007). Under exposure to white rot fungus *Coriolopsis polyzona*, triclosan dimers, trimers, and tetramers were produced via the formation of C-C or C-O bonds. Although triclosan is transformed into high molecular weight metabolites, the triclosan structure remains in the polymer, which could make destruction of this compound more difficult. However, the ether bond of the oligomer can be cleaved to form chlorinated phenols (Lee et al. 2012; Veetil et al. 2012; Mulla et al. 2016).

Degradation of triclosan by axenic cultures of microorganisms

Observing degradation of triclosan by bacteria is limited as this compound is a strong antimicrobial agent. Dann and Hontela (2011) and Wilson et al. (2003) have demonstrated that triclosan is highly toxic to a wide range of bacteria. Thus, experiments to degrade triclosan with bacteria at high concentrations can only be conducted with microorganisms which are resistant to triclosan, which gives sometimes results that cannot easily transferred to real environmental conditions under which triclosan occurs at lower concentrations and thus the resistance towards triclosan is less important.

Triclosan degradation in axenic bacteria cultures

Usually, axenic cultures are used to identify which organisms have the ability to degrade triclosan, and under which conditions the microorganisms are able to do this. Typically, these experiments are conducted with triclosan as single carbon source and high (i.e., mg/L) concentrations are used (catabolic process).

To date, only a few bacteria have been isolated that are able to metabolize triclosan. Meade et al. (2001) isolated two triclosan-resistant bacteria from compost, water, and soil samples, and found that, *Pseudomonas putida TriRY* and *Alcaligenes xylosoxidans* subsp. *denitrificans TR1*, were able to utilize triclosan catabolically (i.e., as a sole carbon source). Triclosan (0.4 mg/L) removal in the liquid growth medium was 94% within 96 h by *P. putida TriRY* and 87% within 228 h by *A. xylosoxidans* TR1. Hay et al. (2001) isolated an auxotrophic *Sphingomonas*-like organism, strain RD1 from activated sludge, which was able to detoxify and mineralize triclosan. Approximately 35% of the ^{14}C triclosan was mineralized to $^{14}C\ CO_2$, when the strain RD1 was grown on 500 mg/L triclosan for 13 days. In addition, triclosan has also been found to be degraded catabolically by *Sphingomonas sp*. PH-07 (Kim et al. 2011), *Sphingopyxis sp*. KCY (Lee et al. 2012), *Mycobacterium vaccae* JOB5 and *Rhodococcus jostii* RHA1 (Lee and Chu 2013), *Nitrosomonas europaea* (Roh et al. 2009), *Methylobacillus* (Lolas et al. 2012) and *Sphingomonas sp*. YL-JM2C (Mulla et al. 2016). Lee and Chu (2013) identified active triclosan degraders in a triclosan-degrading enrichment culture using stable isotope probing (SIP) with universally ^{13}C-labeled triclosan, and demonstrated that *Defluvibacter* (α-Proteobacteria), *Alicycliphilus* (β-Proteobacteria), and *Stenotrophomonas* (γ-Proteobacteria) are capable of assimilating the ^{13}C of the triclosan. Thus, in spite of the inherent challenges in identifying bacteria that are able to degrade a bactericide, quite a bit of different organisms able to degrade triclosan catabolically in axenic cultures have identified.

Fungi

In contrast to bacteria, fungi are often used in biological treatment studies due to their high tolerance and ability to remove pollutants even at high concentrations. They were reported to be involved in the detoxification and degradation of polycyclic aromatic hydrocarbons (Sutherland et al. 1992).

Hundt et al. (1999) suggested that glucosylating and xylosylating enzymes may play a role in the detoxification of xenobiotics. They found that the ligninolytic *Trametes versicolor* was able to metabolize triclosan by converting it to methylated, glucosided, and otherwise carbohydrated conjugates, which suggested the involvement of a UDP-xylosyltransferase in the degradation process. Opposite to that, the fungus *P. cinnabarinus* was reported to be able to methylate the hydroxyl group of triclosan during cultivation (Hundt et al. 2000). Cajthaml et al. (2009) found that almost all the tested ligninolytic fungi (eight strains) were able to degrade triclosan efficiently (3 or 5 mg/L in 14 days), which indicated the involvement of ligninolytic enzymes in the degradation process, however no clear link between enzyme activities and degradation rates was reported. Fungus *Aspergillus versicolor* was able to tolerate the highest triclosan concentration (15.69 mg/L). Murugesan et al. (2009) have reported laccase-mediated triclosan transformation and detoxification through oligomerization and ether-bond cleavage in

the presence and absence of redox mediators. Inoue et al. (2010) investigated triclosan degradation in manganese peroxidase (MnP), laccase and the laccase-mediator system with 1-hydroxybenzotriazol (HBT), and reported that MnP is effective in detoxifying triclosan. In addition, laccase-mediated oxidation and detoxification of triclosan have recently been demonstrated using laccases isolated from *Trametes versicolor* and *Coriolopsis polyzona* (Kim and Nicell 2006; Cabana et al. 2007, 2011). Laccase from *Pleurotus ostreatus* was also reported to be effective in removing triclosan, the main product being oligomerized triclosan (Sun et al. 2017). The enzyme-catalyzed oxidation of triclosan was however significantly affected by pH and temperature (Kim and Nicell 2006; Cabana et al. 2007, 2011).

Algae

Uptake of triclosan by algae plays a significant role in the removal and detoxify triclosan in aquatic environments, as triclosan is bioaccumulating in freshwater algae (Roberts et al. 2014). Wang et al. (2013) investigated the removal of triclosan in water by using a ubiquitous green alga, *Chlorella pyrenoidosa*, which was able to remove more than 50% of triclosan (100 to 800 μg/L) within 1 h. Bai and Acharya (2016, 2017) investigated triclosan removal from Lake Mead water mediated by the green alga *Nannochloris sp.* In these experiments, nearly 100% removal after 7 days of incubation was detected. Wang et al. (2018b) reported that 99.7% of 400 g/L triclosan were removed within 1 day by three common freshwater microalgae: *Chlorella pyrenoidosa*, *Desmodesmus* sp., and *Scenedesmus obliquus*.

Understanding the biodegradation and biotransformation of triclosan by axenic cultures provides information on using microbial consortia to remove triclosan in the environment. However, extrapolation of these lab incubations to the environment or wastewater treatment should be performed with caution as these are considerable more complex systems.

Degradation of triclosan by microbial consortia

Degradation in activated sludge in lab-based simulations

To fill the gap in mass balance between triclosan removal by biodegradation and the total removal by WWTPs, laboratory treatments are conducted to gain insight into the behavior of triclosan. Chen et al. (2011) investigated the degradation of triclosan in laboratory activated sludge systems, and demonstrated that this compound can be degraded quickly under aerobic condition, while it is rather persistent under anaerobic and anoxic conditions.

Previous studies have indicated in activated sludge that about 5% of triclosan is biomethylated to triclosan-methyl (Bester 2003, 2005; Chen et al. 2011) and another 5% of triclosan is transformed to bound residues (Bester 2003), while approximately 10% was sorbed into sludge (Wick et al. 2011; Chen et al. 2011). Recently, Chen et al. (2015a) have reported that degradation products of triclosan, i.e., 2,4-dichlorophenol, monohydroxy triclosan isomers, and triclosan-*O*-sulfate represents 7, 10, and 7% of the initial triclosan, respectively, while the formation of dihydroxy triclosan is relatively less pronounced and slower. These lab-based simulation studies are essential to gain additional understanding of the triclosan removal process in activated sludge and are able to provide parameter guidelines for enhancement of triclosan removal in WWTPs.

It has been suggested that the removal efficiencies of triclosan in activated sludge simulation studies dependent on the initial concentrations. For example, a biodegradation study with an initial concentration of 0.1 mg/L yielded a removal of 61–91% (Stasinakis et al. 2010), whereas a study starting with 5 mg/L reported a removal of 99.8% (Chen et al. 2015a). In a static unfed laboratory-scale activated sludge reactors under aerobic conditions, the removal rates of triclosan were relatively low (75–86%) when the starting concentration was low (≤ 0.5 mg L^{-1}) and reached higher values (> 99%) when the starting concentration was high (≥ 1 mg L^{-1}) (Chen et al. 2011). However, the half-lives (54–86 h) of triclosan in these experiments were not dependent on the initial concentrations (Chen et al. 2011).

Stasinakis et al. (2010) demonstrated the importance of sludge retention time (SRT): operation with SRT = 20 d gave higher removal than operation with SRT = 3–20 d. Armstrong et al. (2018) reported that removal of triclosan is enhanced with increasing hydraulic retention time (HRT), SRT, and temperature in activated sludge system. Chen et al. (2011) have conducted lab-scale activated sludge incubation experiments under aerobic, anaerobic, and anoxic conditions and reported the fastest triclosan removal (49% within 80 h) and its highest transformation rate to triclosan-methyl (16%) for aerobic conditions, while no removal was observed under anaerobic or anoxic conditions. Taştan and Dönmez (2015) reported that triclosan biodegradation increased when pH was increased from five to seven.

The lab-scale studies on removal and biodegradation rate of triclosan in activated sludge incubations cannot always be fully validated with full-scale operation. The most striking reason is that lab experiments are often conducted with mg/L triclosan, while at full-scale usually 1–10 μg/L is experienced (Tables 2 and 3). However, also operational instabilities such as seasonal variations of temperature, rainfall could be reasons for the differences between laboratory and full-scale studies. Nevertheless, the lab-scale studies provided valuable opportunities to assess scaling up to full-scale measurements.

Table 2 Biodegradation of triclosan in axenic cultures and environment compartment and formation of by-products

Degradation process	Exposure conditions			Removal efficiencies	By-products	Ref.
	Triclosan concentration	Matrix	pH			
Bacteria	500 mg/L	In bacterial consortium enriched from activated sludge	7.3	35% of triclosan was mineralized within 3 days	–	Hay et al. 2001
	0.4 mg /L	Bacteria with high levels of triclosan resistance isolated from compost, water, and soil samples	–	94% removal within 96 h by P. putida TriRY and 87% within 228 h by A. xylosoxidans TR1.	–	Meade et al. 2001
	0.5–2 mg/L	In nitrifying activated sludge	8	2 mg/L of triclosan was degraded within 5 days	–	Roh et al. 2009
	0.001–10 mg/L	In complex and mineral salts media by Sphingomonas sp. PH-07	7.3	20% removal after 8 days	Monohydroxy-triclosan; dihydroxy-triclosan; dechlorinated dihydroxy-triclosan; 4-chlorophenol;2,4-dichlorophenol; 3,5-dichlorophenol; 2-methoxy-3,5-dichlorophenol	Kim et al. 2011
	5 mg/L	In a complex nutrient medium by Sphingopyxis strain KCY1	–	Half-velocity constant was 2.8 mg-triclosan/L	Monohydroxy-triclosan;6-chloro-3-(2,4-dichlorophenoxy)-4-hydroxycyclohexa-3,5-diene-1,2-dione; 3-chloro-4-(5,7-dichloro-3-oxo-2,3-dihydrobenzo[1,4] dioxin-2-yl)-2-oxobut-3-ena; 3,5-dichloro-4,6-dihydroxycyclohexa-3,5-diene-1,2-dione; 2,4-dichlorophenol.	Lee et al. 2012
	10 mg/L	In batch cultures with isolated pure strains under aerobic conditions	7.2	95% was degraded in 5 days	Phenol; catechol; 2, 4-dichlorophenol.	Veetil et al. 2012
		In batch cultures with isolated pure strains in methanogenic system with acetate as co-substrate		87% was degraded in 10 days		
	5 mg/L	In different growth substrates by Rhodococcus jostii RHA1 and Mycobacterium vaccae JOB5	–	Initial degradation rate was 0.018–0.021 mg-triclosan/mg-protein/d	2,4-Dichlorophenol; 2-chlorohydroquinone; monohydroxy-triclosan; dihydroxy-triclosan	Lee and Chu 2013
	10 mg/L	In minimal salts medium by Sphingomonas sp. RD1	–	–	2,4-Dichlorophenol;3,5-dichlorocatechol; 4-chlorocatechol;monohydroxytriclosan; acetylated hydroxytriclosan	Kagle et al. 2015
	5 mg/L	In batch cultures with isolated bacterial Sphingomonas sp. strain YL-JM2C from WWTP	7	Triclosan was completely removed within 72 h	Trimethylsilyl (TMS)-triclosan;2,4-dichlorophenol;2-chlorohydroquinone; hydroquinone	Mulla et al. 2016
Fungi	72 mg/L	In nitrogen rich medium	7.5	Most of triclosan was converted to by-products by days 3.	2-O-(2,4,4′-trichlorodiphenyl ether)-b-D-xylopyranoside; 2-O-(2,4,4′-trichlorodiphenyl ether)-b-D-glucopyranoside; 2,4-dichlorophenol; 2,4,4′-trichloro-2′-methoxydiphenyl ether; glucoside conjugate	Hundt et al. 2000
	2.5 mg/L	In static cultures of eight ligninolytic fungal strains	4.5	Triclosan was biodegrade within 14 days	–	Cajthaml et al. 2009
	5, 7.5, 12.5 and 15.5 mg/L	In simulated wastewater and semi-synthetic media by *A. versicolor*	5,6,7,8	Biodegradation yield was 37.47% and 71.91% in simulated wastewater and semisynthetic medium, respectively	2,4-dichlorophenol	Taştan and Dönmez 2015
	5.76 mg/L	Enzyme-catalyzed oxidation by laccase from Trametes versicolor	3–8	Conversion rate of 50–100% was achieved when the enzyme dose increased from 0.3–3 U/L	–	Kim and Nicell 2006

Table 2 (continued)

Degradation process	Exposure conditions			Removal efficiencies	By-products	Ref.
	Triclosan concentration	Matrix	pH			
		under a variety of reaction conditions				
	5 mg/L	By the enzyme preparation from the white rot fungus Coriolopsis polyzona	5	65% was removed after either a 4 or an 8-h treatment	Polymerization of Triclosan produced such as dimers, trimers and tetramers	Cabana et al. 2007
	57.6 mg/L	In an aqueous system using laccase from the fungus Ganoderma lucidum	4.0	56.5% removal was observed within 24 h	2,4-dichlorophenol; 2-chlorohydroquinone	Murugesan et al. 2010
	10 mg/L	By immobilized laccase on Vinyl-modified poly(acrylic acid)/SiO2 nanofibrous membranes	4.3	65% was removed within 2 h	–	Xu et al. 2014
		By free laccase		29% was removed within 2 h		
	5 mg/L	In a 10-mL brown glass tubes with 5 U/mL laccase and 0–20 mg/L humic acid.	7	With initial enzyme activity of 1, 2, 3, 4, 5 and 6 U/mL, the estimated half-life times of initial 5 mg/L Triclosan were 57, 33, 27, 21, 17 and 16 min, respectively.	2-Chlorohydroquinone; 2-chloro-5-(2,4-dichlodichlorophenoxy)-(1,4)benzoquinone; 2,4-dichlorophenol; three dimers; two trimmers and two tetramers	Dou et al. 2018
Algae	0.1–0.8 mg/L	In TAP medium by alga cells	7.4	More than 50% of Triclosan removed within 2 h	Dichlorohydroxydiphenyl ether	Wang et al. 2013
	0–0.8 mg/L	In diatom Navicula sp. cultures inoculated in 500 mL sterile D1 medium.	7.5, 8.3	–	4-chloro-2-hydroxyphenoxy; 4-chloro-6-(dihydroxymethoxy) benzene-1; 2-amino-1-(3-chlorophenoxy)-3-methylbut-1-en-1-ol; 5-chloro-2- methoxyphenoxy (pyrrolidin-2-ylidene) methanol; proline conjugates	Ding et al. 2018
	0–0.8 mg/L	In flasks with medium containing *Chlorella pyrenoidosa*, Desmodesmus sp., and *Scenedesmus obliquus*		High removal rate up to 99.7% was observed during the treatment of 400 g/L Triclosan by the three microalgae for 1 day.	Dichlorohyfroxydiphenyl ether; glucosyl conjugates; 2,4 DCP; pentosyl conjugates	Wang et al. 2018b
Activated sludge	0.5 and 2 mg/L	Lab-scale batch and continuous-flow activated sludge systems	7.8 ± 0.2	More than 90% of triclosan can be removed within 10 h	–	Stasinakis et al. 2007
	0.01–0.04 mg/L	In lab-scale continuous-flow activated sludge systems operated at SRT of 3–20 days	7.7 ± 0.2	94% of triclosan can be removed within 3–20 days	–	Stasinakis et al. 2010
	0.02–3 mg/L	In laboratory-scale activated sludge reactor	6.9 ± 0.5	Half-life of 54–86 h	Triclosan-methyl	Chen et al. 2011
	5 mg/L	In laboratory-scale activated sludge reactor	6.9 ± 0.5	99.8% of Triclosan was removed within 150 h	2,4-dichlorophenol; 4-chlorocatechol; monohydroxy-triclosan; dihydroxy-triclosan; methyl-triclosan; triclosan-O-sulfate	Chen et al. 2015a
	0.1 mg/L	In laboratory-scale activated sludge collected from four different WWTPs	7.0–7.6	61–91% of Triclosan was removed within 120 h and half-life varied from 40 to 100 h	–	Chen et al. 2015b
	0.001 mg/L		7–8.8		–	Inyang et al. 2016

Table 2 (continued)

Degradation process	Exposure conditions			Removal efficiencies	By-products	Ref.
	Triclosan concentration	Matrix	pH			
		In activated sludge batch bench-scale experiments during aerobic, anoxic and anaerobic treatments		59%, 69%, 99% of Triclosan was removed at 8 h (anoxic) and 12 h (anaerobic and anoxic)		
	0.0001–100 mg/L	In laboratory-scale pulse-fed sequencing batch reactors with acclimated and non-acclimated cultures.	7.3–7.5	Triclosan was biologically removed with a percentage of 97%, 80%, 63% at influent Triclosan concentration of 1, 20 and 100 mg/L, respectively.	2,4-dichlorophenol; 2,4-dichloroanisole	Orhon et al. 2017
	4 mg/L	In laboratory-scale activated sludge reactors under different HRT, SRT, and varying treatment temperatures (21 °C and 30 °C)	6.5–7.5	Triclosan was degraded in the solids fraction, with increased rates at 30 °C (−0.0224 ± 0.007/h) when compared to reactors run at 21 °C (− 0.0170 ± 0.003/h).	2,4-dichlorophenol; 4-chloroanaline	Armstrong et al. 2018
	0.2, 1, 5, and 10 mg/L	In glass bottle containing sterile mineral medium and Dyella sp. WW1 isolated from the acclimated activated sludge.	5,7,9	More than 90% of Triclosan can be removed at 15 °C and pH 7.	3,5-dichloro-4-hydrobenzoic	Wang et al. 2018a
Soil and sludge-amended soil	1 mg/kg	Aerobic soil	–	Half-life of 18 days	–	Ying et al. 2007
		Anaerobic soil		Half-life of 70 days		
	2 mg/kg	Sandy loam and silty clay soils	4–8	Estimated half-life of 20–58 days	–	Wu et al. 2009
	4.1–66.6 mg/kg dry wt.	Biosolid-treated soils	–	Estimated half-life of 107.4 days	–	Lozano et al. 2010
	40 mg/kg	In silty clay loam	4.5	Half-life of 78 d	Triclosan-methyl	Waria et al. 2011
		In fine sand.	6.6	Half-life of 42 d		
		In biosolids-amended soils	8	Half-life of 100 d		
	21.9 ± 5.35 mg/kg	Sludge-amended soil	8.2	59%–73% removal within 4 months	–	Butler et al. 2012
	16.53 mg/kg	In biosolids under semi-field conditions	7.3	Triclosan were removed more rapidly in subsurface (DT50 = 17.3 days) than in surface (DT50 = 80.0 days).	–	Al-Rajab et al. 2010
Sludge reed bed treatment plant	1.4–2.0 mg/kg	In full-scale sludge reed bed treatment plant	–	Half-life of 315–770 days	–	Chen et al. 2009a
	0.8 mg/kg	In lab-scale sludge reed bed treatment lysimeters	–	Half-life of 330–462 days	–	Chen et al. 2009b
Sludge compost	0.29 mg/kg	In a full-scale compost windrow made from digested sewage sludge, yard waste, and horse manure for 24 days		Half-life of 25 days	Triclosan-methyl	Poulsen and Bester 2010
	1.3 mg/kg	In full-scale compost piles		Half-life of 25 days	Triclosan-methyl	Sadef et al. 2014a, b
	8.1 mg/kg	In sludge composting bench-scale measurements conducted at 50 and 70 °C over a 52 day period		Half-life of 33 days	Triclosan-methyl	Sadef et al. 2015
Sediment	1 mg/L	In water	8.4	Half-life of 89–161 days	Triclosan-methyl	Huang et al. 2014
		In water-sediment system	7.6	Half-life of 32–62 days		
	1,10,100 mg/kg	In sediments	–	Half-life varied from 55 to 239 days	–	Huang et al. 2015
	30 mg/kg	In water-sediment system	6–9	Half-life varied from 16 to 93 days	–	Guo et al. 2016
Human and animals	18,576 mg/L	In alcoholic solution for vivo topical application	–	23%, 6.3% of the dose had penetrated completely through rat skin and human skin	Glucuronide and sulfate conjugates	Moss et al. 2000
	1.44 mg/L	In rat liver	7.4	Half-life was less than 45 min		Wu et al. 2010

Table 2 (continued)

Degradation process	Exposure conditions			Removal efficiencies	By-products	Ref.
	Triclosan concentration	Matrix	pH			
	0.00015 to 0.217 mg/L	Human urine, serum	5.0	–	Hydroxylated triclosan; hydroxylated and sulfated triclosan; sulfated triclosan; hydroxylated and glucuronidated triclosan; glucuronidated and acetylcysteine; cysteine conjugation of triclosan; glycosylated triclosan; triclosan with an acetylamino group sulfonated triclosan; hydroxylated sulfonated triclosan; glucuronidated triclosan	Wu et al. 2012

Full-scale wastewater treatment plants (WWTPs)

Widespread use of triclosan provides a number of pathways which all end in wastewater and thus finally reach the WWTPs. The influent concentrations of triclosan in WWTPs can vary dramatically according to usage pattern from below 1 μg/L to exceeding 100 μg/L (with 100 μg/L being very untypical for European wastewater see Table 3 and references within). Wastewater (inflow as well as effluent) concentrations are thus usually above its predicted-no-effects concentration (PNEC) of 0.05 μg/L, and even exceeded the low-observed-effect-concentration (LOEC) of 0.015 μg/L (Singer et al. 2002; Wilson et al. 2003).

Most WWTPs rely on microbial processes to remove and degrade organic pollutants from wastewater. As triclosan concentrations usually are in the μg/L range and the BOD is in the several hundred mg/L, it is most probable that triclosan removal occurs not as catabolic degradation but as co-degradation, i.e., the microbial communities take their energy and carbon to grow on from other substrates. In wastewater treatment, three types of biological treatment are dominating: activated sludge, trickling filters, and rotating biological contactors—with activated sludge being clearly the most widespread used technology. In WWTPs, triclosan can be removed by (a) mineralization, (b) transformation by oxidation or reduction, and (3) sorption to sludge (Latch et al. 2003; Bester 2003, 2005; Yu et al. 2006). In fact, both biodegradation and sorption to activated sludge contribute to the removal of triclosan in WWTPs.

In activated sludge, WWTPs about 30–40% of the triclosan partitions into sludge as triclosan is relatively lipophilic (log K_{ow} of 4.2–4.8; log K_{oc} of 4.3) (Bester 2003, 2005; Coogan et al. 2007; Heidler and Halden 2007; Kumar et al. 2010; Nakada et al. 2010). On the other hand, this means that most of the removal (30–50% of the total mass fraction) occurs due to biodegradation processes (Bester 2005; Heidler and Halden 2007; Kumar et al. 2010; Nakada et al. 2010). Total removal of triclosan is thus adding up to typically > 90% in classical activated sludge WWTPs.

The rate of biodegradation of triclosan is highly variable and dependent on the operational parameters of WWTPs such as exposure time (hydraulic retention time), oxygen concentration, as well as the microbial community (which is operationally influenced by sludge retention time). WWTPs with activated sludge treatment process generally have hydraulic retention times of 10 to 40 h and high dissolved oxygen levels (1.5–2.0 mg/L) in the BOD removal and nitrification part of the plants. These parts of the WWTP have been shown to be responsible for the high removal of triclosan of 76–99% (Table 3). However, in WWTPs with fixed-film biological processes

Table 3 Removal efficiencies of triclosan in full-scale WWTPs

Type of treatment plants	Location	Influent	Effluent	Percentage removal	Reference
Activated sludge	Switzerland	0.6–1.3	0.07–0.65	76–92	Lindström et al. 2002
		0.04–0.2	0.01–0.1	94	Singer et al. 2002
	UK	21.9	1.1	95	Sabaliunas et al. 2003
		0.67–5.12	ND-0.1	95–98	Thompson et al. 2005
	Germany	1	0.05	95	Bester 2003
		7.3 ± 1.5 4.8 ± 0.55	0.3 ± 0.1 0.62 ± 1.55	96 87	Bester 2005
	Greece	0.45	0.076	83	Pothitou and Voutsa 2008
		0.42–1.14	0.14–0.18	84–87	Villaverde-de-Saa et al. 2010
	Spain	0.94	<LOQ	99	Gonzalez-Marino et al. 2009
		0.31 ± 0.01	0.03	91	Regueiro et al. 2009
		0.73 ± 0.09	0.07–0.1	86–90	Montes-Hernandez et al. 2009
	USA	5.2–10.7	0.24–0.41	95–96	McAvoy et al. 2002
		1.1	0.027	97.5	Kanda et al. 2003
		3–3.6	0.028–0.072	97–99	Thomas and Foster 2005
		4.7 ± 1.6	0.07 ± 0.06	98 ± 1	Heidler and Halden 2007
		3.44	0.19	94	Fair et al. 2009
		18.85 13.7	1.04 0.18	95 99	Kumar et al. 2010
		86.16	5.37	94	
		32.64	0.27	99	
		4.53	0.12	97	Buth et al. 2011
		0.45	0.036	92	
	Japan	0.27–0.75	0.02–0.4	86–93	Nakada et al. 2006
		2.7 11.9	0.26 0.27	90 98	Nakada et al. 2010
	China	142.0 ± 16.5	22.5 ± 1.4	84	Wu et al. 2007
	Israel	0.4–2.8	ND-0.2	27–84	Dotan et al. 2016
	Australia	0.59–0.81	0.02–0.43	72–93	Ying and Kookana 2007
Trickling filter	UK	7.5	0.34	95.5	Sabaliunas et al. 2003
		0.99–3.7	ND-0.3	86–97	Thompson et al. 2005
	USA	3.80–16.6	1.61–2.7	58–87%	McAvoy et al. 2002
		3.7	0.13	96.5	Kanda et al. 2003
Rotating biological contactors	UK	1.28–3.89	0.081–1.12	58–96	Thompson et al. 2005

such as trickling filters and rotating biological contactors, the hydraulic retention time is shorter (1–4 h) and the dissolved oxygen levels are relative low (0.3 mg/L) (Mathys et al. 1997). These WWTPs showed relatively low removal of triclosan with a large variation (58–97%) (Table 3).

Pycke et al. (2014) investigated the biodegradation of triclosan in sludge from 3 different WWTPs with a broad range of wastewater flows. However, the hydraulic retention time (10–40 h), sludge retention time (14–20 days), and dissolved oxygen concentration (4–10 mg/L) are clearly influencing the overall efficiency of triclosan removal prior to parameters like temperature, pH, and loading rate (Stasinakis et al. 2010; Chen et al. 2015b; Inyang et al. 2016).

Though it is improbable that the triclosan removal is generally coupled to other processes than BOD removal, there are some indications towards the processing of triclosan might in some cases also be linked to ammonia-oxidizing bacteria in nitrifying activated sludge (Roh et al. 2009).

Generally, all biological treatment systems are able to metabolize triclosan, as it seems to be one of the easier to degrade xenobiotic compounds. Whether and to what extent this is effective depends on the reaction speed in the respective system and the residence time in it.

Degradation in sludge post-treatment (composting and reed beds)

Activated sludge WWTPs produce excess sludge that can be incinerated, used directly on land as fertilizer or used on land after post treatment, e.g., by composting or treatment in sludge readbeds.

Composting (aerobic) under well-controlled conditions is an effective method in humidifying the organic matter in the sludge and thus stabilizing the material; additionally, this process is suited to biodegrade organic micro-pollutants (Epstein 1997). For instance, degradation of triclosan in sewage sludge during thermophilic composting was investigated by Poulsen and Bester (2010) and Sadef et al. (2014b) who reported half-lives of 25 days for triclosan in full-scale sludge composting windrow and 33 days in sludge composting bench-scale measurements (Sadef et al. 2015). Pakou et al. (2009) suggested that the efficiency of composting is generally dependent on the operational conditions such as initial concentration of pollutants, oxygen availability, temperature, pH, and moisture content during the process. Sadef et al. (2014a) has reported that removal rates depended strongly on temperature and for triclosan an optimal temperature for removal is 30–50 °C. The results from Sadef et al. (2015) showed that micro-pollutant removal rates decreased when the oxygen concentration decreased or inorganic nitrogen content increased. In addition, changes in biosolids/wood shaving ratio have significantly impact on micro-pollutant removal in other experiments (Das and Xia 2008).

Reed beds are often used in small WWTPs for inexpensive sludge dewatering as well as in reducing volume, breaking down organic matter and increase the density of sludge (Nielsen 2003, 2005; Gustavsson et al. 2007). Chen et al. (2009a, b) have found degradation of triclosan in reed beds with half-life of 315–770 days in full-scale and 330–462 days in lysimeters. Considering 10-year production cycle of reed beds (Nielsen 2003), more than 95% of triclosan can be removed even though the reaction rate is relatively slow.

Degradation in sludge-amended soil

Triclosan has been found in soils due to irrigation with reclaimed water and application of sewage sludge (biosolids), which lead to adverse effects on soil organisms, crops and even human beings (Qin et al. 2015). Concentrations of triclosan in sludge (biosolids) typically range from 1 to 10 mg/kg (Bester 2003). In agricultural praxis, only a small fraction of sludge is amended into the soil.

Degradation of triclosan in soils generally followed first-order kinetics, with half-lives ranged from days to years. The degradation rate varied with soil properties and incubation conditions (Wu et al. 2009). For example, in biosolids (sludge)-amended soils, the half-lives of triclosan were 42 days in a silty clay loam, 50 days in a fine loam, 108 days in a coarse loam (Waria et al. 2011), while 107 days were determined in loam (Lozano et al. 2010, 2013), and 182 to 193 days in sandy clay loam (Walters et al. 2010). Ying et al. (2007) incubated triclosan in loamy soil with 1 mg/kg of triclosan, and found 92% removal of triclosan within 70 days under aerobic condition, while triclosan was persistent under anaerobic conditions. Butler et al. (2011) dosed 10–1000 mg/kg triclosan with three different soils (sandy loam, clay, and loamy sand) and indicated that triclosan removal was inversely correlated to triclosan dose, soil organic matter, and clay content. Butler et al. (2012) reported that degradation of triclosan in soil was relatively slow in winter period due to the low temperature and soil moisture. Moreover, Al-Rajab et al. (2010) evaluated the effect of depth of placement of biosolids in the soil profile on the triclosan dissipation rates under semi-field conditions. This group reported a faster dissipation of triclosan in subsurface (half-life of 17.3 days) than surface (half-life of 80 days) applied biosolids. Removal of triclosan in soil could be attributed to biodegradation, adsorption, leaching, and possibly the formation of non-extractable residues (Butler et al. 2012).

Degradation in sediments

Sediments could be an important sink for triclosan in aquatic environment. However, little is known about the fate and behavior of triclosan in sediment systems. Singer et al. (2002) investigated the 30-year-old sediment layer in lake Greifensee, indicating that triclosan degradation has to be very slow in these sediments. The fate of triclosan may also be affected by its bioavailability (Huang et al. 2014). Lin and Gan (2011) studied the adsorption of triclosan onto river sediments and indicated that the sorption capacity is highly dependent on content of organic matter, clay, BET surface area, and pH value. Degradation of triclosan was more rapid in water-sediment systems than in pure sediment systems: The half-life ranged from 32 to 62 days in water-sediment systems (Huang et al. 2014) and 55 to 239 days in sediment systems (Huang et al. 2015).

Research needs

This review reveals that triclosan can be removed in various systems via biodegradation, while the degradation rate is highly dependent on load of triclosan and operational conditions including composition of microbial community. Biological transformation products have been reported by several investigators; however, knowledge on biodegradation pathways need to be further developed to close the mass balance of triclosan. Further work should be focused on the investigation of the fate and toxicity of triclosan metabolites in the environment. In addition, a study on the effects of triclosan metabolites on plants, animals, and humans could be useful for health

risk assessment of triclosan. This is to some extent hampered by the fact that only a minor fraction of the triclosan metabolites are available.

Single organisms are known to degrade triclosan in axenic cultures; it is however unknown which species in a sludge community indeed are performing the degradation, and what is the ecology that steers the degradation processes.

Several laboratory studies determining the biodegradation of triclosan under a certain conditions are available. To continuously optimize the triclosan removal rates as well as detoxify triclosan and its metabolites, more research is needed especially in realistically scaled up pilot reactors.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no competing interests.

Ethical approval This paper does not contain any studies with human participants or animals performed by any of the authors.

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