Endocrine-disrupting chemicals (EDC) are a wide group of chemicals that interfere with the endocrine system. Their similarity to natural steroid hormones makes them able to attach to hormone receptors, thereby causing unfavorable health effects. Among EDC, bisphenol A (BPA), bisphenol S (BPS), and nonylphenol (NP) seem to be particularly harmful. As the industry is experiencing rapid expansion, BPA, BPS, and NP are being produced in growing amounts, generating considerable environmental pollution. White rot fungi (WRF) are an economical, ecologically friendly, and socially acceptable way to remove EDC contamination from ecosystems. WRF secrete extracellular ligninolytic enzymes such as laccase, manganese peroxidase, lignin peroxidase, and versatile peroxidase, involved in lignin deterioration. Owing to the broad substrate specificity of these enzymes, they are able to remove numerous xenobiotics, including EDC. Therefore, WRF seem to be a promising tool in the abovementioned EDC elimination during wastewater treatment processes. Here, we review WRF application for this EDC removal from wastewater and indicate several strengths and limitations of such methods.

Introduction

The past two decades have seen growing awareness of the possible adverse effects on human and animal health resulting from exposure to endocrine-disrupting chemicals (EDC). This group includes xenoestrogens, i.e., exogenous substances with estrogen activity, to which bisphenol A (BPA), bisphenol S (BPS), and nonylphenol (NP) belong (Pothitou and Voutsa 2008; Michalowicz 2014; Pookpoosa et al. 2014; Garcia-Morales et al. 2015; Guo et al. 2016; Catanese and Vandenberg 2017; Yan et al. 2017; Diao et al. 2017; Urriola-Muñoz et al. 2017; Česen et al. 2018; Wu et al. 2018b; Noszczyńska and Piotrowska-Seget 2018). EDC are associated with a wide variety of disorders (Ben-Jonathan 2004; Kandaraki et al. 2011; Schug et al. 2011; De Coster and Van Larebeke 2012). Despite the negative effects of EDC, they are widely used in industry (Noszczyńska and Piotrowska-Seget 2018; Rodriguez-Peña et al. 2019). As a result of the extensive production, processing, and transport of EDC-containing products and EDC themselves, these compounds often contaminate aquatic environments, as shown in Table 1 (Pothitou and Voutsa 2008; Terzić et al. 2008; Janex-Habibi et al. 2009; Kasprzyk-Hordern et al. 2009; Martin Ruel et al. 2010; Rosal et al. 2010; Yu et al. 2013; Yang et al. 2014a, b; Jin and Zhu 2016; Lu et al. 2019; Radwan et al. 2020; Singh and Thakur 2020). Currently, wastewater treatment systems are not able to cope with EDC removal, which are present in wastewater in trace amounts even at ng L⁻¹ (Niemuth and Klaper 2015; Bai and Acharya 2019; Lv et al. 2019). In response to this problem, various techniques of degradation, transformation, and/or removal of EDC from wastewater have been applied. Among them, white rot fungi (WRF) seem to be an efficient and ecologically friendly method with the potential to transform most of the xenobiotics. The majority of prior research has been conducted on EDC removal by WRF. Our aim is to summarize the existing knowledge and indicate gaps in the research that need to be filled.
Endocrine-disrupting chemicals

EDC are nonpersistent or persistent chemicals (Cajthaml 2015; Corrales et al. 2015). Nonpersistent EDC include chemicals that are rapidly degraded in the environment and are quickly metabolized in and eliminated from the human body (Nelson et al. 2020). Persistent EDC are stable in both the environment and the human body and undergo significant biomagnification for a short or long period (Song et al. 2014; de Voogt 2018). EDC are described as chemically synthesized or naturally existing compounds, absent within living organisms, that interfere with the endocrine system by imitating or inhibiting endogenous hormones, thus consecutively inducing hormonal dysfunctions, having a negative impact on living organisms (S. Environ. Prot. Agency 1997; Go re et al. 2015; Björnsdotter et al. 2017; Lauretta et al. 2019). On the one hand, EDC may show an affinity to specific nuclear receptors known as peroxisome proliferator-activated receptors (PPARs) (Cocci et al. 2013; Agarwal et al. 2017; Sharma et al. 2018). PPARs are normally involved in the binding of certain ligands such as steroid hormone molecules or fatty acids, acting as transcription factors, thus regulating the expression of genes associated with lipid metabolism in the organism (Urriola-Muñoz et al. 2014; Catanese and Vandenberg 2017; Gupta and Pushkala 2019). Therefore, the influence of EDC on PPARs contributes to an elevated adipocyte level in the body and the risk of obesity (Heindel et al. 2015; Ahn et al. 2020). On the other hand, the main targets of EDC are estrogenic receptors (ERα and ERβ), which can be stimulated or inactivated by appropriate conjunction of the ligand. Thus, EDC work either as antagonists or agonists of ERs, disrupting the estrogenic balance in organisms (Rogers et al. 2013; Sifakis et al. 2017). BPA and NP are among the best known xenoestrogens. However, due to the increasing use of BPS in industry and its widely demonstrated negative impact on human hormonal system, in the scientific literature, more and more attention is devoted to this compound (Viñas and Watson 2013; Catanese and Vandenberg 2017; Urriola-Muñoz et al. 2017; Qiu et al. 2018; Gupta and Pushkala 2019).

Bisphenol A has become one of the most intensively manufactured chemicals in the world due to demonstrating the finest properties for plastic production (Noszczyńska and Piotrowska-Seget 2018). Numerous studies have investigated BPA effects on the human body and animals (Zhu et al. 2015; Quesada et al. 2002; Braun et al. 2009; Izzotti et al. 2009; Pfeifer et al. 2015; Leung et al. 2017; Maćczak et al. 2017; Pinney et al. 2017; Tiante et al. 2018; Grandin et al. 2019; Özel et al. 2019; Gao et al. 2020; Rasdi et al. 2020; Tassinari et al. 2020; Wu and Seebacher 2020; Wu et al. 2020a; Pan et al. 2020). Since BPA has a comparable structure to that of natural estrogen 17β-estradiol, it can bind to ERα and ERβ, though with 1000-fold less affiliation than estradiol (Gray et al. 2004; vom Saal and Hughes 2005; Takayanagi et al. 2006). Despite this, BPA, even at low doses measured in ng L⁻¹, is capable of disrupting human cell function by interacting with extranuclear receptors (Michałowicz 2014). For instance, BPA binds to membrane estrogen receptors and GPR30 protein-coupled receptors and, hence, participates in nongenomic pathways (Rubin 2011; Cygankiewicz et al. 2015). The literature review shows that BPA is not only an endocrine-disrupting chemical, but it also causes damage to hepatocytes through oxidative stress (Kourouma et al. 2015; Elswefy et al. 2016; Li et al. 2017). BPA can modulate the
immune structure provides it both hydrophilic and hydrophobic character; hence, it acts as an effective uncharged surfactant (John et al. 2000; Soares et al. 2008). Therefore, NP is a suitable raw material in the production of paints, cosmetics, detergents, hair dyes, and pesticides. In addition, the presence of NP is observed in vinyl chloride (PVC), which can contaminate water passing through PVC plumbing (EPA 2005). Due to its high hydrophobicity, resistance to biodegradation, and low solubility, it is prone to accumulate in various environmental matrices (Krupiński and Długoński 2011). Consequently, NP was detected in water averaging 0.805 μg L\(^{-1}\) in China; 12.61 μg L\(^{-1}\), 12.2 μg L\(^{-1}\), and 6.08 μg L\(^{-1}\) in recreational water, wastewater discharges, and drinking water, respectively, in Mexico; 1.6 μg L\(^{-1}\) in Japan; and 0.22 μg L\(^{-1}\) in Ukraine (Hoai et al. 2003; Zhang et al. 2017; Vysata et al. 2018; Vargas-Berrones et al. 2020). However, as evaluated by the Water Framework Directive of the European Union, the maximum NP concentration in water in Europe is 2 μg L\(^{-1}\) (EU, Directive 2013/39/EU 2013), while in the USA, the Environmental Protection Agency (EPA U 2010) establishes this dose as 6.6 μg L\(^{-1}\) (EPA 2005). Owing to NP’s lipophilic properties, it can be deposited in adipose tissue (Yu et al. 2020). Also, NP is capable of binding to ER receptors by competing with natural estrogen (E2), although with lower affinity than the natural hormone (Noorimotlagh et al. 2017). As a result of the above mechanism, NP induces disorders in men, including a reduction in the level of circulating testosterone in the blood, decreased activity of antioxidant enzymes in sperm, and disturbed testicular structure as well as enhanced apoptosis of Sertoli cells (Cardinali et al. 2004; Gong et al. 2009; Aly et al. 2012; Hu et al. 2014; Urriola-Muñoz et al. 2014). On the other hand, a study showed that high exposure to NP of women in the second trimester of pregnancy led to reduced birth weight of the child and shortened the gestational age (Chang et al. 2013).

**White rot fungi**

In the forest ecosystem, wood decomposition is a key process in the carbon and nutrient cycle (Purahong et al. 2016). The rate of wood decay is determined by external factors such as substrate quality and climate as well as the diversity and activity of the organisms that contribute to degradation (Brischke et al. 2006). Moreover, wood contains a high lignin content, which significantly hinders the breakdown process (Purahong et al. 2016). WRF are among the best lignin degraders. Their name derives from a specific process of bleaching which occurs during the degradation of wood by fungi (Ten Have and Teunissen 2001). Interestingly, it was demonstrated that Fe\(_{3}\)O\(_{4}\) nanomaterials combined with the WRF *Phanerochaete chrysosporium* have promising potential for application in lignocellulose degradation (Huang et al.
WRF are primarily classified as *Basidiomycota* type; however, also a limited number represent *Ascomycota* (Patel et al. 2014). These fungi are common in nature, usually found in forest ecosystems, more often in deciduous than coniferous forests (Singh and Singh 2014). Besides a capacity for lignin degradation, WRF have remarkable versatility in breaking down a wide variety of complex and resistant environmental contaminants that pollute aquatic ecosystems, posing a potential threat to human and animal health. It is quite well proven that WRF have a biochemical ability to degrade sulfonamide antibiotics and important categories of toxic, organic xenobiotics such as polycyclic aromatic hydrocarbons (PAH), 1,1,1-trichloro-2,2-bis(4-chlorophenyl) ethane (DDT), synthetic textile dyes, polychlorinated biphenyls (PCB), pentachlorophenols (PCP), and trinitrotoluene (TNT). Furthermore, these organisms are capable of producing lignin-modifying enzymes, which, apart from their ability to degrade lignin, are active against xenobiotics, including EDC (Hashim et al. 2018). There are four main classes of LMEs: laccases, manganese peroxidases, lignin peroxidases, and versatile peroxidases (Cabana et al. 2007b; Cajthaml 2015). Although WRF are capable of producing all classes of enzymes, particular strains may not release all of them together (Yang et al. 2013a). LMEs are synthesized by fungi undergoing secondary metabolism, as lignin oxidation does not supply energy to them. The limited nutrient quantity in the medium, such as carbon or nitrogen, as well as hypoxia stimulates the synthesis of these enzymes (Niku-Paavola et al. 1990; Pointing 2001; Marco-Urrea et al. 2010; Mattila et al. 2020). Mixing of liquid fungi cultures generates laccase production but inhibits the synthesis of lignin and manganese peroxidase. On the other hand, high oxygen molecular pressure leads to increased secretion of lignin and manganese peroxidase. Frequently, several LME isoforms are produced by fungi depending on the fungus strain and culture conditions (Torres et al. 2003; Wesenberg et al. 2003; Levin et al. 2004; Yang et al. 2013b; Kinnunen et al. 2016). Temperature, pH, agitation, or the presence of inorganic salts or heavy metals affects the breakdown of endocrine-disrupting chemicals by LMEs. These parameters influence the activity of enzymes, their stability, and substrate specificity (Kim and Nicell 2006; Soares et al. 2006; Auriol et al. 2007; Kinnunen et al. 2016).

The advantage of fungi over bacteria in lignin mineralization results from the production and secretion of LMEs outside the cell. In addition, fungi can operate over a wide range of temperatures and pH values, while enzymes are synthesized during nutrient deficiency (Robinson et al. 2001; Arora and Gill 2005; Urek and Pazarlioglu 2007; Dhakar and Pandey 2013; Hariharan and Nambisan 2013). Expanding fungal hyphae can make it possible to reach contaminants inaccessible to bacteria (Cabana et al. 2007b). Moreover, WRF enzymes are nonspecific so that the fungi can transform compounds resembling lignin in their chemical structure. Such compounds may include pesticides, alkanes, aromatic hydrocarbons, or bisphenol A (Harms et al. 2011). The secretion of LMEs outside the cell gives fungi access to nonpolar and insoluble substances (Llorca et al. 2017). Meanwhile, the presence of functional groups such as amine, hydroxyl, or alkyl groups in chemical compounds, acting as electron donors, makes these compounds more susceptible to electrophilic oxygenase attack. Therefore, WRF effectively remove phenolic compounds such as BPA and NP (Tadkaew et al. 2011; Yang et al. 2013b).

### Manganese peroxidase

The peroxidase most frequently produced by WRF is manganese peroxidase (MnP). MnP is a glycoprotein containing a prosthetic group in the form of a heme molecule (an iron complex with protoporphyrin IX). There are existing multiple MnP isoforms with a molecular weight between 32 and 62.5 kDa (Qiu et al. 2019). This enzyme was discovered for the first time in *P. chrysosporium* almost 30 years ago, and it is the only heme peroxidase with a single-electron mechanism of Mn$^{3+}$ oxidation reaction (Pollegioni et al. 2015). MnP catalyzes the oxidation of...
Mn$^{2+}$ to Mn$^{3+}$ via hydrogen peroxide (H$_2$O$_2$) required as an electron acceptor (Dashtban et al. 2010).

\[ 2\text{Mn}^{2+} + 2\text{H}^+ + \text{H}_2\text{O}_2 \rightarrow 2\text{Mn}^{3+} + 2\text{H}_2\text{O} \]

The reaction catalyzed by MnP begins with the conversion of the native enzyme through hydrogen peroxide to the first transitional compound [Cpd-I], which constitutes the Fe$^{4+}$ radical complex (Fig. 1) (Manavalan et al. 2015). At the same time, the Mn$^{2+}$ ion is oxidized to Mn$^{3+}$, and a second transitional compound [Cpd-II] is formed. Mn$^{3+}$ ion is then separated from the surface of the enzyme and is linked to carboxylic acids, in particular, oxalate and malate. The chelated Mn$^{3+}$ complex acts as an oxidant of phenolic rings, reducing to the Mn$^{2+}$ ion and producing a transitional phenoxyl radical, resulting in the formation of various breakdown products (Pollegioni et al. 2015). The native enzyme is created from the Cpd-II, through electron release and oxidation of Mn$^{2+}$ to the Mn$^{3+}$ complex. The chelated Mn$^{3+}$ can restore the phenoxyl radical, which oxidizes sequential phenolic rings (Manavalan et al. 2015). The Mn$^{3+}$ complex is restricted exclusively to the oxidation of phenolic compounds such as simple phenols, amines, dyes, and lignin phenolic compounds. In relation to nonphenolic compounds, the complex remains inactive due to deficient redox potential (Manavalan et al. 2015; Żygo and Prochoń 2017). Besides, the action of MnP is entirely inhibited by inhibitors such as Hg$^{2+}$, Pb$^{2+}$, Ag$^+$, Na$_3$N, lactate, or ascorbic acid (Manavalan et al. 2015).

**Lignin peroxidase**

Lignin peroxidase (LiP) is a glycoprotein with a molecular weight between 38 and 46 kDa, which contains heme as a prosthetic group, whereas the entire enzyme is stabilized via 4 disulfide bridges. The LiP structure is very akin to MnP since it is a globular protein composed of 11–12 α-helixes containing the central cavity with a heme group (Manavalan et al. 2015; Pollegioni et al. 2015). Such a considerable analogy of both enzymes may point to divergent selection (Pollegioni et al. 2015). Despite the structural resemblance, LiP exhibits significantly greater redox potential [$E_0'$ ≈ 1.2 V] in comparison with MnP [≈ 0.8 V], due to a higher deficit of ferrous atom electrons in the porphyrin ring (Abdel-Hamid et al. 2013; Pollegioni et al. 2015). This advantage allows LiP to oxidize, along with phenolic compounds, even nonphenolic xenobiotics and lignin components, regardless of the presence of a mediator. Nevertheless, an elevated concentration of hydrogen peroxide or compounds such as acetone and diethyl ether as well as dioxane functions as LiP inactivators in many fungi (Manavalan et al. 2015).

LiP disintegrates lignin and xenobiotics in three stages, involving hydrogen peroxide (Fig. 2) (Pollegioni et al. 2015). The catalytic reaction is initiated by oxidation of the native LiP enzyme to the transient compound [Cpd-I], which forms the radical complex Fe$^{4+}$. Crucial in this reaction is H$_2$O$_2$, serving as an electron acceptor. In a further stage, the transitional compound [Cpd-I] is reduced by a xenobiotic such as EDC to a second transitional compound [Cpd-II] (Abdel-Hamid et al. 2013; Falade et al. 2017). Simultaneously, the xenobiotic molecule converts into a radical form through electron depletion, followed by nonenzymatic reactions leading to the formation of the final degradation product (Dashtban et al. 2010). In order to complete the enzymatic cycle and regain the native form, LiP must be reduced anew, with the consequent occurrence of the subsequent xenobiotic radical (Abdel-Hamid et al. 2013). Concerning lignin decomposition, LiP favors veratryl alcohol (VA) as a nonphenolic substrate providing electrons for redox reactions. As a natural metabolite of

![Fig. 1 MnP catalytic cycle (Pollegioni et al. 2015, modified)](image1)

![Fig. 2 LiP catalytic cycle during degradation of xenobiotics (Abdel-Hamid et al. 2013, modified)](image2)
fungi in contact with lignin, VP increases the catalytic properties of the enzyme and the velocity of lignin breakdown (Muszyńska et al. 2017). As a result of VA oxidation, a radical cation of this compound is formed and acts as a direct lignin oxidant (Fig. 3) (Abdel-Hamid et al. 2013).

**Versatile peroxidase**

Similar to previous peroxidases, versatile peroxidase (VP) also presents a glycoprotein structure with a molecular weight varying between 38 and 45 kDa, with heme in the central region, functioning as an enzyme cofactor. VP has been originally reported in the *Pleurotus eryngii* species, whereas at this point, the presence of VP has been only confirmed in the species of *Pleurotus* and *Bjerkandera* fungi (Abdel-Hamid et al. 2013). The versatility of this peroxidase is achieved by combining the catalytic properties of MnP and LiP, through the ability to oxidize Mn$^{+2}$ and due to high redox potential (Abdel-Hamid et al. 2013). Hence, VP is able to degrade both nonphenolic and phenolic components of lignin and xenobiotics, as well as numerous dyes (e.g., Reactive Black 5 nonphenolic and phenolic components of lignin and xenobiotics) (Pollegioni et al. 2015). Moreover, a hybrid VP provides multiple binding sites for substrates. The catalytic efficiency of VP in the oxidation of Mn$^{+2}$ ions is comparable to MnP. However, in the case of oxidation of phenolic and nonphenolic components of lignin, this enzyme is ten times less productive than LiP (Pollegioni et al. 2015).

The mechanism of phenolic compound breakdown by VP is analogous to MnP. At the first stage, the cofactor Fe$^{4+}$ complex of the native enzyme is oxidized to the transient compound [Cpd-I] radical in the presence of H$_2$O$_2$ (Fig. 4). Simultaneously, Mn$^{2+}$ is converted into Mn$^{3+}$, and then the oxidized ion combines with carboxylic acids to maintain its stability (Pollegioni et al. 2015). The Mn$^{3+}$ complex functions as an oxidant of phenolic compounds leading to the formation of a transient phenoxyl radical and, consequently, to the generation of final breakdown products (Manavalan et al. 2015). As a result of manganese ion oxidation, a second transient compound [Cpd-II] is formed, which can revert to the initial enzyme form by gaining an electron. Electron loss allows the Mn$^{3+}$ to oxidize subsequent phenolic rings (Manavalan et al. 2015).

On the other hand, VP employs an identical mechanism as LiP for the elimination of both nonphenolic compounds and lignin polymer. The native enzyme is oxidized to a transient compound (Cpd-I) radical involving hydrogen peroxide. Cpd-I is further reduced by a single electron delivered from a nonphenolic compound (xenobiotic, VA) to a second transition compound (Cpd-II) (Fig. 5) (Abdel-Hamid et al. 2013; Falade et al. 2017). Hence, a radical form of xenobiotic molecule is created, which is exposed to nonenzymatic reactions (coupling, polymerization, side-chain splitting, demethylation, regrouping) (Dashtban et al. 2010). Termination of a cycle by VP is possible by continued reduction of the Cpd-II compound, as well as the simultaneous generation of a new nonphenolic radical molecule (Abdel-Hamid et al. 2013).

**Laccase**

Laccase (Lac) is the most commonly occurring enzyme in the environment among LMEs. Lac has been primarily detected in the Asian tree *Toxicodendron vernicifluum* species. Currently, this enzyme is identified in numerous species of plants and microorganisms such as bacteria and fungi, including a majority of WRF (e.g., *P. eryngii*, *Trametes versicolor*, *P. chrysosporium*). Lac, together with the rest of LMEs, belongs to glycoproteins, although it has a greater molecular weight, reaching even up to 150 kDa, as well as a distinctive blue color. In the central region of the enzyme, 4 copper cations are located, divided into 3 types (Manavalan et al. 2015; Pollegioni et al. 2015). Type 1 (T1) exhibits a high level of absorption at 600 nm, which is responsible for the unique pigmentation of an enzyme. Copper type 2 (T2) is deprived of color, though it possesses paramagnetic properties, whereas type 3 (T3) is composed of two interconnected diamagnetic cations exhibiting peak absorbance equal to 330 nm (Strong and Claus 2011). Lac belongs to the oxidases group; therefore, it participates in the 4 electron transition from distinct substrate molecules to O$_2$, which is subsequently reduced to H$_2$O$_2$ (Fig. 6) (Muszyńska et al. 2017).

The catalytic cycle of this enzyme is initiated by the progressive oxidation process of 4 separate substrate particles and simultaneously the passage of 4 subsequent electrons to the copper cations in the active center, resulting in a state of full Lac reduction (Pollegioni et al. 2015). In the second stage, a single O$_2$ molecule joins the T3 and T4 copper cations, rapidly transforming into a transition peroxide by obtaining two individual electrons from both T3 ions. However, this condition does not persist long since oxygen falls apart into an
oxyradical, engaging 2 additional electrons from copper molecules, which split the oxygen bonds. This is accompanied by the release of the first water particle. Completion of a Lac catalytic cycle is achieved by total oxidation of each of the four copper ions and release of a second water molecule (Pollegioni et al. 2015). The above reaction mechanism allows Lac to degrade phenols and phenolic components of lignin, as well as nonphenolic compounds, but only in the presence of redox mediators (Abdel-Hamid et al. 2013).

Potential of WRF to remove BPA, BPS, and NP from wastewater

Due to the increasing urbanization, EDC are increasingly produced by many branches of industry. As a consequence, these substances penetrate the soil and water, which causes significant pollution affecting these ecosystems. Despite EDC being present in the environment mainly at low concentrations in the order of ng L\(^{-1}\), they can be a serious threat both for aquatic animals and humans (Solé and Schlosser 2015). Therefore, such recalcitrant compounds have to be removed from wastewater. Since traditional sewage treatment plants using activated sludge processes eliminate EDC only to a limited extent, there is a need to look for other effective methods for their removal (Ahmed et al. 2017; Cecconet et al. 2017). Numerous attempts, including adsorption, filtration, chlorination, coagulation/flocculation, Fenton/photo-Fenton degradation, sonochemical degradation, photochemical/photocatalytic oxidation, ozonation, and hybrid processes with physical and thermal approaches, have been made to remove EDC from water (Yoon et al. 2007; Sharma et al. 2009; Zhang and Li 2014; Ahmed et al. 2017). However, these procedures are costly and often result in equally toxic secondary impurities. Alternatively, the use of WRF for remediation of contaminated water is cost-effective and sustainable. WRF compared to other potential bioremediation bacteria are not adversely affected by the antibiotics commonly found in wastewater (Boer 2018). On the other hand, WRF need a second source of carbon, as the abovementioned EDC degradation takes place as part of the secondary metabolism. Despite that, in contrast to bacteria, WRF are able to decompose EDC even at low concentrations (Mir-Tutusaus et al. 2018). Many different studies have been conducted on the effectiveness of removing EDC from the environment (Kim et al. 2007; Toyama et al. 2009; Huang et al. 2014; Zielinska et al. 2016; Csuros et al. 2018; Li et al. 2020; Oh et al. 2020; Stenholm et al. 2020; Suyamud et al. 2020; Zhang et al. 2020). Much of this research has been devoted to the use of both whole WRF cells and extracted enzymes in EDC degradation, although tests on the former were more repeatedly reported. As this review focused on BPA, BPS, and NP removal using WRF, therefore, in the description below, particular emphasis has been placed on the use of these organisms in the removal of the abovementioned compounds.

For research applications, WRF systems are constructed in the form of bioreactors, providing a constant substrate supply, thus maintaining controlled environmental conditions (Tadkaew et al. 2011; Ahmed et al. 2017). Examples of the efficiency of EDC removal by whole-cell WRF cultures are shown in Table 2. The results vary among studies since the degradation capacity depends on multiple factors such as the molecular structure of the xenobiotic, the species of the applied fungus, and the type of secreted enzymes.
Several authors have shown that the first step of organic pollutant biodegradation by WRF may be sorption of these compounds to the fungal mycelium caused by the high surface to volume ratio of WRF (Zafar et al. 2007; He et al. 2010; Ding et al. 2013; Nguyen et al. 2014). On the other hand, it was revealed that crude or purified LME solutions were able to catalyze EDC biodegradation in the absence of sorption to fungal mycelium (Yang 2012). It results from the hydrophobic character of these compounds ($\log K_{ow} \geq 3.2$), which determines the adsorption behavior of EDCs (Krupadam et al. 2011). Most of the studies have mainly revealed EDC removal from the aqueous phase without monitoring the extent of biosorption (Pezzella et al. 2017; Mtbáa et al. 2018; Brazkova 2019). It creates difficulties in assessing the relative

![Lac catalytic cycle](https://example.com/lac-catalytic-cycle.png)

**Fig. 6** Lac catalytic cycle (Pollegioni et al. 2015, modified)

| WRF species     | Culture conditions                          | EDC               | Initial concentration (mg/L) | Incubation time | Removal efficiency (%) | References                     |
|-----------------|---------------------------------------------|-------------------|------------------------------|-----------------|------------------------|--------------------------------|
| *T. versicolor* | Bubble column/internal loop airlift bioreactor<br>Temp 28 °C <br>Temp 28 °C<br>Batch bioreactor<br>Temp 25 °C<br>pH 4.5 | Bisphenol A 22.83<br>Nonylphenol 22.04<br>Bisphenol A 500<br>Nonylphenol 0.0017 | 8 days<br>8 days<br>6 h<br>2 days | 100<br>84<br>98.7<br>52.9 | | Pezzella et al. (2017) | Llorca et al. (2017) |
| *P. ostreatus*  | Continuous flow trickle-bed bioreactor<br>Temp 28 °C | Bisphenol A 2<br>Nonylphenol 2 | 12 days<br>12 days | >90<br>90 | | Křesinová et al. (2018) |
| *P. chrysosporium* | Bubble column/internal loop airlift bioreactor<br>Temp 28 °C | Nonylphenol 22.04<br>Bisphenol A 22.83 | 8 days<br>8 days | 65<br>60 | | Pezzella et al. (2017) |
contribution of biosorption and biodegradation to the general removal of the highly hydrophobic EDC. Only a few studies on biosorption effects alone on EDC treatment have been performed. Among them are the studies performed by Nguyen et al. (2014) who observed not higher than 30% efficiency of BPA sorption to inactivated T. versicolor biomass, and Yonten et al. (2016) who gained up to 90% of BPA removal by adsorption to Pleurotus eryngii immobilized on polymeric resin. Immobilization greatly facilitates biosorption by increasing the mechanical strength of the biosorbent and reusability (Wu and Yu 2007). Additionally, factors such as pH or volume of the sample solution can influence the course of the sorption treatment. Increasing biosorption of BPA was observed in the pH range of 7–11, with maximum adsorption at pH 11, while a decreasing trend was noted at the lower pH of 2–7 (Yonten et al. 2016). The same authors also revealed that BPA is removed from the solution exponentially only up to a specific moment, followed by a constant value, due to the complete saturation of absorbent by BPA. Besides biosorption, the participation of intracellular and/or mycelium-associated enzymes in EDC biodegradation cannot be excluded. Therefore, more comprehensive studies answering the contribution of these enzymes should be performed. Until now, the main role as an alternative oxyreductase to LMEs has been assigned to intracellular cytochrome P450 (Marco-Urrea et al. 2006). This was confirmed by Wang et al. (2013), who analyzed loss of BPA in nonligninolytic conditions with Phanerochaete sordida. Weekly treatment showed 80% BPA reduction, while the use of cytochrome P450 inhibitor decreased the degradation efficiency to under 40%. On the other hand, the interplay of intracellular cytochrome P450 and LMEs may strongly influence EDC elimination, although the entire mechanism still remains undiscovered (Haroune et al. 2017). Therefore, LMEs are considered as a main mechanism for EDC elimination by WRF.

Each WRF can secrete a distinct type of LME depending on the species or even strain (Torres et al. 2003; Wesenberg et al. 2003; Levin et al. 2004; Yang et al. 2013b; Kinnunen et al. 2016). The enzymatic pathways of living WRF undergo the control of gene promoters, which are stimulated by an appropriate environmental factor (Suetomi et al. 2015; Toyokawa et al. 2016; Daly et al. 2020). The triggering factor for LMEs is primarily the balance of nitrogen and carbon in the medium. A high carbon/nitrogen ratio in the environment enhances the expression of enzymatic genes similar to the presence of phenolic compounds, improving WRF efficiency in the removal of contaminants (Keyser et al. 1978; Soares et al. 2005). On the other hand, the lack of sufficient trigger affects the activity of LME synthesis pathways, significantly lowering the EDC elimination rate (Janusz et al. 2013). T. versicolor has been the object of most studies, due to its proven high efficiency in EDC removal. The vast majority of these fungal strains secrete up to three extracellular enzymes involved in EDC decomposition (Bending et al. 2002; Takamiya et al. 2008). It can be noted from Table 2 that T. versicolor reached a substantial reduction (> 80%) for most tested EDC (Llorca et al. 2017; Pezzella et al. 2017; Brazkova 2019) and up to 100% for BPA (Pezzella et al. 2017). However, the remaining Pleurotus ostreatus and P. chrysosporium species, despite having a different combination of LMEs, also achieved high removal rates from 60 to over 90% (Pezzella et al. 2017; Kršinová et al. 2018). Unfortunately, due to different culture conditions and various incubation times, the presented data is hard to compare.

Despite the high productivity of the WRF on a laboratory scale under sterile and controlled conditions, such results do not provide much knowledge about fungal activity and their capacity for mycoremediation in highly variable wastewater conditions (Accinelli et al. 2010; Strong 2010; Anastasi et al. 2011; Ntougias et al. 2012; Zhang and Geißlen 2012; Cruz-Morató et al. 2014). Fungi have to confront autochthonous organisms as well as multiple microcontaminants at low concentrations. Therefore, intensified research in nonsterile conditions has recently been conducted, with a view to their future industrial application (Blánquez et al. 2008; Lu et al. 2009; Cruz-Morató et al. 2013, 2014; Badia-Fabregat et al. 2015). Nonetheless, this approach faces several limitations. It has been found that the microflora naturally existing in wastewater interfere to some extent with the decomposition processes undertaken by WRF (Svobodová and Novotný 2018). Bacteria compete with fungi for nutrients and carbon sources, influencing fungal growth and synthesis of extracellular enzymes. On the other hand, bacteria decompose substances harmful to the WRF and enhance the level of nitrogen required for fungal growth (Válková et al. 2017; Mir-Tutusaus et al. 2018). In order to reduce the competition between bacteria and fungi, various strategies are applied to ensure that the culture conditions are favorable for fungi. One method is to adjust the acidic pH, optimal for fungi (Libra et al. 2003). Low pH will suppress the growth of bacteria that prefer a neutral environment, thus increasing WRF activity. However, such an approach of supporting fungal growth does not work for a long period because the bacteria are capable of adapting to acidic conditions (Mir-Tutusaus et al. 2018). Moreover, too acidic pH could result in a decrease of enzyme secretion by the WRF. Another solution implies the replacement of existing fungal biomass during degradation, due to its aging over time. The access of young mycelium allows the degradation time to be extended, also increasing the activity of the WRF (Blánquez et al. 2006; Dhouib et al. 2006; Badia-fabregat et al. 2017). Attempts have also been made to restrict the access of nitrogen to the medium, causing limited bacterial growth, though it is effective just at the beginning of the degradation since during the process, the bacteria start to absorb nitrogen from the fungi (Libra et al. 2003; Asif et al. 2017; Svobodová and Novotný 2018). This problem may be
overcome by the application of extracted LMEs. Compared to the whole WRF cell, isolated enzymes are more specific to the degraded xenobiotic as well as capable of operating across a wide range of environmental conditions, thus simplifying the control of the entire process (Gassara et al. 2013; Becker et al. 2017; Falade et al. 2017). Nevertheless, the enzymes remain less efficient in degradation than the WRF due to the synergic interactions between the extracellular enzymes and mycelium (Yang et al. 2013a). In addition, fungi can secrete low molecular weight redox mediators, which can expand the range of degradable compounds (Abdel-Hamid et al. 2013; Asif et al. 2017). The next issue related to the application of enzymes includes high production and purification costs, as well as instability and no possibility of reuse (Gassara et al. 2013; Bilal et al. 2017a; Pezzella et al. 2017; Voběrková et al. 2018). Therefore, it is becoming increasingly common to implement methods of enzyme immobilization. They are based on linking the catalyst with the carrier in order to keep it in limited space and maintain its structure (Voběrková et al. 2018). The carrier should feature no toxicity, easy accessibility, and strong biological integrity with the enzyme. As the particle creates bonding with the enzyme, its structure and properties have a significant influence on the enzymatic activity of the immobilized catalyst. Both organic polymers (cellulose, starch, chitin, chitosan, silica alginate) and chemically synthesized inorganic molecules are used in the immobilization process (Al-Adhami et al. 2002; Wang et al. 2011; Kampmann et al. 2014; Verma et al. 2020). The organic ones, owing to their natural source, exhibit enhanced biological compatibility toward the enzyme. However, nowadays, non-organic particles are gaining increasing interest (Acevedo et al. 2010; Hou et al. 2014; Ji et al. 2017). The advantage of synthetic materials is their great stiffness and highly specific surface zone, which can be easily modified through suitable functional groups according to the requirements of the situation (Barcelos et al. 2016). The catalyst can also be stabilized without supporting carrier through the construction of cross-linked enzyme conglomerates (Asgher et al. 2014). Immobilization significantly increases the stability of the enzyme thereby improving resistance to chemical and thermal denaturation. As a result, production costs are reduced due to the regenerative potential of the enzyme and the possibility of reuse (Boer 2018; Voběrková et al. 2018). Moreover, reactions involving immobilized enzymes take place in a broad spectrum of environmental conditions (Asgher et al. 2014). Since the late nineteenth century, as research on enzyme immobilization has progressed, multiple diverse methods have been developed. A distinction can be made between physical (adsorption, entrapment) and chemical (covalent bonding, cross-linking) methods (Li et al. 2012; Kim et al. 2016; Wu et al. 2018a). Physical methods do not require additional reagents and show simplicity, though the link between the carrier and the enzyme remains weak. These are mainly hydrogen

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### Table 3

| Enzyme type | Fungal species | Immobilization type | Incubation time | EDC Initial concentration (mg/L) | Removal efficiency (%) | References |
|-------------|----------------|---------------------|-----------------|---------------------------------|------------------------|------------|
| Laccase     | T. versicolor | H. communis spore-based scaffolds | 24 h | Bisphenol A 2 | 56 | Zádarta et al. (2018) |
|             |                |                     |                 | Bisphenol S 2 | 55 | Maryskova et al. (2019) |
|             |                |                     |                 | Bisphenol F 2 | 95 | Bilal et al. (2019) |
|             |                |                     |                 | >99              | >99              | Brugnani et al. (2018) |
|             | M. thermophila | Silica nanoparticles | 150 min | Bisphenol A 10 | >99              | Bilal et al. (2017b) |
|             | P. ostreatus   | MANAE-agarose       | 60 min          | Bisphenol A 100 | 100             | Tobecka-Pojg et al. (2011) |
|             |                |                     |                 | Nonylphenol 5 | 96 | Gassam et al. (2013) |
| Manganese peroxidase | G. lucidum | Cross-linked enzyme aggregates (CLEAs) | 150 min | Nonylphenol 10 | 100             | Bilal et al. (2019) |
| Versatile peroxidase with glucose oxidase | B. adusta | Cross-linked enzyme aggregates (CLEAs) | 10 min | Bisphenol A 10 | 100             | Table et al. (2011) |
| Manganese peroxidase, lignin peroxidase, and laccase | P. chrysosporium | Encapsulation in polyacrylamide (PA) microgel | 8 h | Bisphenol A 10 | 100             | Gassam et al. (2013) |
bonds, hydrophobic interactions, or van der Waals forces. By contrast, in chemical methods, a stronger covalent bond is formed between the molecules. Unfortunately, the strength of the connection creates the risk of interfering with the enzyme activity (Vobèrková et al. 2018; Bilal et al. 2019). Among the well-known methods of immobilization, frequently used are cross-linking, encapsulation, entrapment, or covalent bonding (Vobèrková et al. 2018). So far, it is considered that the most effective technique is covalent linking, in which the enzyme is strongly attached to the carrier by covalent bonds (Gasser et al. 2014; Zhu et al. 2020). Due to the possibility of forming multiple solid connections, the stability and activity of the immobilized enzyme increases significantly. Cross-linking appears to be an equally efficient solution owing to the high stability and restoration capacity of the catalyst, as well as the economic advantage of industrial use. Additionally, this method enables two or more proteins to be immobilized in one aggregate, allowing many independent degradation processes to be conducted (Guisan 2013; Asgher et al. 2014; Bilal et al. 2017b; Vobèrková et al. 2018). The choice of a suitable method is essential for the immobilization process as it determines the subsequent activity of the enzyme along with the properties of the aggregate, whereas there is no universal solution for each protein (Mohamad et al. 2015; Vobèrková et al. 2018). Table 3 presents the results of the degradation efficiency of various EDCs by selected immobilized enzymes obtained by differing techniques. The majority of performed studies are focused on immobilized laccase due to its prevalence among WRF, as well as its versatility enabling numerous technological applications (Asgher et al. 2014). Research on immobilized laccase has shown a very high degree of EDC reduction (> 85%) (Gamallo et al. 2018; Zdarta et al. 2018; Bilal et al. 2019; Maryskova et al. 2019), which reached even 100% in the case of laccase from *P. ostreatus* (Brugnari et al. 2018). Nevertheless, in single studies using universal and manganese peroxidase, a sufficient degradation rate, exceeding 95%, has also been achieved (Taboada-Puig et al. 2011; Bilal et al. 2017b). In addition, an experiment involving several LMEs proved to be equally productive, with a 90% decrease in BPA (Gassara et al. 2013). Promisingly, in all mentioned studies involving immobilized enzymes (Table 3), there were observed comparable or improved degradation results as whole-cell WRF (Table 2) in a significantly shorter incubation time not exceeding 24 h. Thus, the immobilized enzymes exhibit the potential for future industrial use upon improved optimization and reduced production costs.

**Conclusions**

EDC are a global problem in the environmental and health field. These compounds are constantly used in many production processes, hence negatively affecting human and animal health. Research carried out so far has shown that usage of WRF is a promising alternative for traditional wastewater treatment plants (WWTP) using activated sludge allowing for EDC removal from water. Despite the many advantages of WRF application, some challenges before using this technique on an industrial scale need to be solved. Actual WWTP are not designed for the new technology, while the adaptation is very expensive. Furthermore, regulation of the enzyme pathway in EDC degradation by WRF requires better understanding. Additionally, more comprehensive experiments should be performed on real wastewater aimed at gaining better insight into possible use of WRF in natural conditions, while prior studies have explored the degradation effectiveness of xenobiotics by specific LMEs on the laboratory scale. These studies recognized immobilized enzymes as having the greatest potential for industrial-scale use, so further tests should be undertaken in this direction.

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**References**

Abdel-Hamid AM, Solbiati JO, Cann IKO (2013) Insights into lignin degradation and its potential industrial applications. Adv Appl Microbiol 82:1–28. https://doi.org/10.1016/B978-0-12-407679-2.0001-6

Acchinelli C, Saccà ML, Batissou I et al (2010) Removal of oseltamivir (Tamiflu) and other selected pharmaceuticals from wastewater using a granular bioplastic formulation entrapping propagules of *Phanerochaete chrysosporium*. Chemosphere 81:436–443. https://doi.org/10.1016/j.chemosphere.2010.06.074

Acevedo F, Pizzul L, Castillo M et al (2010) Degradation of polycyclic aromatic hydrocarbons by free and nanoclay-immobilized manganese peroxidase from *Anthracophilum discolor*. Chemosphere 80:271–278. https://doi.org/10.1016/j.chemosphere.2010.04.022

Agarwal S, Yadav A, Chaturvedi RK (2017) Peroxisome proliferator-activated receptors (PPARs) as therapeutic target in neurodegenerative disorders. Biochem Biophys Res Commun 483:1166–1177. https://doi.org/10.1016/j.bbrc.2016.08.043

Ahmed MB, Zhou IL, Ngo HH et al (2017) Progress in the biological and chemical treatment technologies for emerging contaminant removal from wastewater: a critical review. J Hazard Mater 323:274–298. https://doi.org/10.1016/j.jhazmat.2016.04.045
Harms H, Lapenna S, Bremer S (2012) Weak estrogenic transcriptional activities of bisphenol A and bisphenol S. Toxicol in Vitro 26:727–731. https://doi.org/10.1016/j.tiv.2012.03.013

Guisan JM (2013) Immobilization of enzymes and cells. Methods Mol Biol 1051:1–375. https://doi.org/10.1007/978-1-62703-550-7

Guo H, Li H, Liang N et al (2016) Structural benefits of bisphenol S and its analogs resulting in their high sorption on carbon nanotubes and graphite. Environ Sci Pollut Res 23:8976–8984. https://doi.org/10.1007/s11356-016-6040-7

Guo X, Peng Z, Huang D et al (2018) Biotransformation of cadmium-sulfamethazine combined pollutant in aqueous environments: *Phanerochaete chrysosporium* bring cautious optimism. Chem Eng J 347:74–83. https://doi.org/10.1016/j.cej.2018.04.089

Gupta P, Pushkala A (2019) Increasing woman’s health concern due to xenoestrogens and parabens: a review. Cell Tissue Res 19:6829–6832.

Harirhan S, Nambisan P (2013) Optimization of lignin peroxidase, manganese peroxidase, and laccase production by *Ganoderma lucidum* under solid state fermentation of pineapple leaf. BioResources 8:6832–6841. https://doi.org/10.15376/biores.8.4.6832-6841

Hashim N, Kassim M, Yusof N, Sharifuddin S (2018) Bioremediation of hexabromocyclododecane in birds from an E-waste region in South China: influence of diet on diastereoisomer- and enantiomer-specific distribution and trophodynamics. Environ Sci Technol 52:335–342. https://doi.org/10.1021/acs.est.9b03945

Hashim, N., Kassim, M., Yusof, N., Sharifuddin, S. (2018) Bioremediation of xenoestrogens and parabens: a review. Cell Tissue Res 19:6829–6832.

Hashim N, Kassim M, Yusof N, Sharifuddin S (2018) Bioremediation of endocrine disruptors: the role of microbial enzymes. Asian J Biotechnol Bioreosur Technol 2:1–9. https://doi.org/10.1038/nmicr2519

Haroune L, Saibi S, Cabana H, Bellenger JP (2017) Intracellular enzymes contribution to the biocatalytic removal of pharmaceuticals by *Trametes hirsuta*. Environ Sci Technol 51:897–904. https://doi.org/10.1021/acs.est.6b04409

Hashin M, Kassim M, Yusof N, Sharifuddin S (2018) Bioremediation of endocrine disruptive chemicals: the role of microbial enzymes. Asian J Biotechnol Bioreosur Technol 2:1–9. https://doi.org/10.1038/nmicr2519

He MJ, Luo XJ, Yu LH et al (2010) Tetrabromobisphenol-A and hexabromocyclododecane in birds from an E-waste region in South China: influence of diet on diastereoisomer- and enantiomer-specific distribution and trophodynamics. Environ Sci Technol 44:5748–5754. https://doi.org/10.1021/es101503r

Heindel JJ, Newbold R, Schug TT (2015) Endocrine disruptors and obesity. Nat Rev Endocrinol 11:653–661. https://doi.org/10.1038/nrendo.2015.163

Hélias-Toussaint C, Peyre L, Costanzo C, Chagnon MC, Rahmani R (2014) Is bisphenol S a safe substitute for bisphenol A in terms of metabolic function? An in vitro study. Toxicol Appl Pharmacol 280:224–235

Hirano T, Honda Y, Watanabe T, Kuwahara M (2000) Degradation of bisphenol A by the lignin-degrading enzyme, manganese peroxidase, and laccase, produced by the white-rot basidiomycete, *Pleurotus ostreatus* sp. UHH 5-1-03. Appl Microbiol Biotechnol 100:2381–2399. https://doi.org/10.1007/s00253-015-7113-0

Hou J, Dong G, Ye Y, Chen V (2014) Enzymatic degradation of bisphenol-A with immobilized laccase on TiO2 sol-gel coated PVDF membrane. J Membr Sci 469:19–30. https://doi.org/10.1016/j.memsci.2014.06.027

Hu Y, Wang R, Xiang Z et al (2014) Antagonistic effects of a mixture of low-dose nonylphenol and di-N-butyl phthalate (monobutyl phthalate) on the Sertoli cells and serum reproductive hormones in prepupal male rats in vitro and in vivo. PLoS One 9:1–9. https://doi.org/10.1371/journal.pone.0093425

Huang R, Fang Z, Fang X, Tsang EP (2014) Ultrasonic Fenton-like catalytic degradation of bisphenol A by ferroferric oxide (Fe3O4) nanoparticles prepared from steel pickling waste liquor. J Colloid Interface Sci 436:258–266. https://doi.org/10.1016/j.jcis.2014.08.035

Huang D, Li T, Xu P et al (2019) Deciphering the Fenton-reaction-aid lignocellulose degradation pattern by *Phanerochaete chrysosporium* with ferroferric oxide nanomaterials: enzyme secretion, straw humification and structural alteration. Bioresour Technol 276:335–342. https://doi.org/10.1016/j.biotech.2019.01.013

Izzotti A, Kanitz S, D’Agostini F et al (2009) Formation of adducts by bisphenol A, an endocrine disruptor, in DNA in vitro and in liver and mammary tissue of mice. Mutat Res-Genet Toxicol Environ Mutagen 679:28–32. https://doi.org/10.1016/j.mrgentox.2009.07.011

Janex-Habibi ML, Huyard A, Esperanza M, Bruchet A (2009) Reduction of endocrine disruptor emissions in the environment: the benefit of wastewater treatment. Water Res 43:1565–1576. https://doi.org/10.1016/j.watres.2008.12.051

Janusz G, Kucharzyk KH, Pawlik A et al (2013) Fungal laccase, manganese peroxidase and lignin peroxidase: gene expression and regulation. Enzym Microb Technol 52:1–12. https://doi.org/10.1016/j.enzmicro.2012.10.003

Ji C, Nguyen LN, Hou J et al (2017) Direct immobilization of laccase on titaonia nanoparticles from crude enzyme extracts of *P. ostreatus* culture for micro-pollutant degradation. Sep Purif Technol 178:223–233. https://doi.org/10.1016/j.seppur.2017.01.043

Jin H, Zha L (2016) Occurrence and partitioning of bisphenol analogues in water and sediment from Liaohe River Basin and Taihu Lake, China. Water Res 103:343–351. https://doi.org/10.1016/j.watres.2016.07.059

John DM, House WA, White GF (2000) Environmental fate of nonylphenol ethoxylates: differential adsorption of homologs to components of river sediment. Environ Toxicol Chem 19:293–300. https://doi.org/10.1002/etc.5620190207

Kachlishvili E, Asatiani M, Kakhidze A, Elisashvili V (2016) Trinitrotoluene and Mandarin peels selectively affect lignin-modifying enzyme production in white-rot basidiomycetes. SpringerPlus 5:1–9. https://doi.org/10.1186/s40064-016-1895-0

Kampmann M, Boll S, Kossuch J et al (2014) Efficient immobilization of mushroom tyrosinase utilizing whole cells from *Agaricus bisporus* and its application for degradation of bisphenol A. Water Res 57:295–303. https://doi.org/10.1016/j.watres.2014.03.054

Kandaraki E, Chatzigeorgiou A, Livadas S et al (2011) Endocrine disruptors and polycystic ovary syndrome (PCOS): elevated serum levels of bisphenol A in women with PCOS. J Clin Endocrinol Metab 96:480–484. https://doi.org/10.1210/jc.2010-1658

Kasprzyk-Hordern B, Dinsdale RM, Grant MA (2009) The removal of pharmaceuticals, personal care products, endocrine disruptors and illicit drugs during wastewater treatment and its impact on the quality of receiving waters. Water Res 43:363–380. https://doi.org/10.1016/j.watres.2008.10.047

Keyser P, Kirk TK, Zeikus JG (1978) Ligninolytic enzyme system of *Phanerochaete chrysosporium*: synthesized in the absence of lignin in response to nitrogen starvation. J Bacteriol 135:790–797. https://doi.org/10.1128/jb.135.3.790-797.1978
electrooxidation on BDD anode and oxidation by H₂O₂ in a continuous flow electrochemical reactor. Int J Electrochem Sci 14:4409–4419. https://doi.org/10.20964/2019.05.21

Rogers JA, Metz L, Yong VW (2013) Endocrine disrupting chemicals and immune responses: a focus on bisphenol-A and its potential mechanisms. Mol Immunol 53:421–430. https://doi.org/10.1016/j.molimm.2012.09.013

Rosal R, Rodriguez A, Perdigón-Melón JA et al (2010) Occurrence of emerging pollutants in urban wastewater and their removal through biological treatment followed by ozonation. Water Res 44:578–588. https://doi.org/10.1016/j.watres.2009.07.004

Rubin BS (2011) Bisphenol A: an endocrine disruptor with widespread exposure and multiple effects. J Steroid Biochem Mol Biol 127:27–34. https://doi.org/10.1016/j.jsbmb.2011.05.002

Rykowska I, Wasiak W (2006) Properties, threats, and methods of analysis of bisphenol A and its derivatives. Acta Chromatogr 7:27–37

S. Environmental Protection Agency (1997) Special report on environmental endocrine disruption: an effects assessment and analysis. Rep. EPA/630/R-96/012, https://archive.epa.gov/raf/web/pdf/endocrine. Accessed February 1997

Saito T, Kato K, Yokogawa Y et al (2004) Detoxification of bisphenol A and nonylphenol by purified extracellular laccase from a fungus isolated from soil. J Biosci Bioeng 98:64–66. https://doi.org/10.1016/S1389-1723(04)70243-1

Schug TT, Janesick A, Blumberg B, Heindel JJ (2011) Endocrine disrupting chemicals and disease susceptibility. J Steroid Biochem Mol Biol 127:204–215. https://doi.org/10.1016/j.jsbmb.2011.08.007

Seachrist DD, Bonk KW, Ho SM et al (2016) A review of the carcinogenic potential of bisphenol A. Reprod Toxicol 59:167–182. https://doi.org/10.1016/j.reprotox.2015.09.006

Services-GLOBAL MT (2013) Bisphenol A: In: Rep. code CPE/018/13, Tech. Update

Sharma VK, Anquandah GAK, Yngard RA et al (2009) Nonylphenol, octylphenol, and bisphenol-A in the aquatic environment: a review on occurrence, fate, and treatment. J Environ Sci Heal Part A Toxic/Hazardous Subst Environ Eng 44:423–442. https://doi.org/10.1080/10934520902719704

Sharma S, Ahmad S, Khan MF et al (2018) In silico molecular interaction of bisphenol analogues with human nuclear receptors reveals their stronger affinity vs. classical bisphenol A. Toxicol Mech Methods 28:660–669. https://doi.org/10.1080/15376516.2018.1491663

Shin EH, Choi HT, Song HG (2007) Biodegradation of endocrine-disrupting bisphenol A by white rot fungus Irpex lacteus. J Microbiol Biotechnol 17:1147–1151

Sifakis S, Androustoupoulos VP, Tsatsakis AM, Spandidos DA (2017) Human exposure to endocrine disrupting chemicals: effects on the male and female reproductive systems. Environ Toxicol Pharmacol 51:56–70. https://doi.org/10.1016/j.etap.2017.02.024

Singh AP, Singh T (2014) Biotechnological applications of wood-rotting fungi: a review. Biomass Energy 62:198–206. https://doi.org/10.1016/j.biombioe.2013.12.013

Singh R, Thakur IS (2020) Cancer treatment drugs and endocrine-disrupting chemicals release and fate in hospital wastewater. In: Current developments in biotechnology and bioengineering

Soares A, Jonasson K, Terrazas E et al (2005) The ability of white-rot fungi to degrade the endocrine-disrupting compound nonylphenol. Appl Microbiol Biotechnol 66:719–725. https://doi.org/10.1007/s00253-004-1747-7

Soares A, Guieysse B, Mattiasson B (2006) Influence of agitation on the removal of nonylphenol by the white-rot fungi Trametes versicolor and Bjerkandera sp. BOL 13, Biotechnol Lett 28:139–143. https://doi.org/10.1007/s10529-005-5326-5

Soares A, Guieysse B, Jefferson B et al (2008) Nonylphenol in the environment: a critical review on occurrence, fate, toxicity and treatment in wastewaters. Environ Int 34:1033–1049. https://doi.org/10.1016/j.envint.2008.01.004

Sole M, Schlosser D (2015) Xenobiotics from human impacts. Ecol Biochem Environ Interspecies Interact 258–276. https://doi.org/10.1002/9783527680633.ch13

Song S, Song M, Zeng L et al (2014) Occurrence and profiles of bisphenol analogues in municipal sewage sludge in China. Environ Pollut 186:14–19. https://doi.org/10.1016/j.envpol.2013.11.023

Song P, Fan K, Tian X, Wen J (2019) Bisphenol S (BPS) triggers the migration of human non-small cell lung cancer cells via upregulation of TGF-β1. Toxicol in Vitro 54:224–231. https://doi.org/10.1016/j.tiv.2018.10.005

Sridhar M (2016) Versatile peroxidases: super peroxidases with potential biotechnological applications—a mini review. Journal of Dairy, Veterinary & Animal Research 4(2)

Stasinakis AS, Mermigka S, Samaras VG et al (2012) Occurrence of endocrine disrupters and selected pharmaceuticals in Axios River (Greece) and environmental risk assessment using hazard indexes. Environ Sci Pollut Res 19:1574–1583. https://doi.org/10.1007/s11356-011-0661-7

Stella T, Covino S, Cvan M et al (2017) Bioremediation of long-term PCB-contaminated soil by white-rot fungi. J Hazard 324:701–710. https://doi.org/10.1016/j.jhazmat.2016.11.044

Stenholt M, Hedeland M, Arvidsson T, Pettersson CE (2020) Removal of nonylphenol polyethoxylates by adsorption on polyurethane foam and biodegradation using immobilized Trametes versicolor. Sci Total Environ 724:138159. https://doi.org/10.1016/j.scitotenv.2020.138159

Strong PJ (2010) Fungal remediation of Amlania distillery wastewater. World J Microbiol Biotechnol 26:133–144. https://doi.org/10.1007/s11274-009-0152-x

Strong PJ, Claus H (2011) Laccase: a review of its past and its future in bioremediation. Crit Rev Environ Sci Technol 41:373–434. https://doi.org/10.1080/10643380902945706

Sutonti M, Sakamoto T, Tokunaga Y et al (2015) Effects of calmodulin on expression of lignin-modifying enzymes in Pleurotus ostreatus. Cur Genet 61:127–140. https://doi.org/10.1007/s00294-014-0460-z

Suyumud B, Thiravetyan P, Gadd GM et al (2020) Bisphenol A removal from a plastic industry wastewater by Dracaena sanderiana endophytic bacteria and Bacillus cereus NI. Int J Phytoremediation 22:167–175. https://doi.org/10.1080/15226514.2019.1652563

Svobodová K, Novotný Č (2018) Bioreactors based on immobilized fungi: bioremediation under non-sterile conditions. Appl Microbiol Biotechnol 102:39–46. https://doi.org/10.1007/s00253-017-8575-z

Taboada-Puig R, Junghanss C, Demarche P et al (2011) Combined cross-linked enzyme aggregates from versatile peroxidase and glucose oxidase: production, partial characterization and application for the elimination of endocrine disruptors. Bioreourc Technol 102:6593–6599. https://doi.org/10.1016/j.biotech.2011.03.018

Tadkaew N, Hai FI, McDonald JA et al (2011) Removal of trace organics by MBR treatment: the role of molecular properties. Water Res 45:2439–2451. https://doi.org/10.1016/j.watres.2011.01.023

Takamiya M, Magan N, Warner PJ (2008) Impact assessment of bisphenol A on lignin-modifying enzymes by basidiomycete Trametes versicolor. J Hazard Mater 154:33–37. https://doi.org/10.1016/j.jhazmat.2007.09.098

Takayanagi S, Tokunaga T, Liu X et al (2006) Endocrine disruptor bisphenol A strongly binds to human estrogen-related receptor y (ERRy) with high constitutive activity. Toxicol Lett 167:95–105. https://doi.org/10.1016/j.toxlet.2006.08.012

Tassinari R, Narciso L, Tait S et al (2020) Corrigendum to: “Juvenile toxicity rodent model to study toxicological effects of bisphenol A (BPA) at dose levels derived from Italian children biomonitoring study”. Toxicol Sci 175:143. https://doi.org/10.1093/toxsci/kfaa016
Ten Have R, Teunissen PJM (2001) Oxidative mechanisms involved in lignin degradation by white-rot fungi. Chem Rev 101:3397–3413. https://doi.org/10.1021/cr0001151
Terzić S, Senta I, Ahel M et al (2008) Occurrence and fate of emerging wastewater contaminants in Western Balkan region. Sci Total Environ 399:66–77. https://doi.org/10.1016/j.scitotenv.2008.03.003
Tian Y, Zhou X, Miao M et al (2018) Association of bisphenol A exposure with LINE-1 hydroxymethylation in human serum. Int J Environ Res Public Health 15:1–10. https://doi.org/10.3390/ijerph15081770
Torres E, Bustos-Jaimes I, Le Borgne S (2003) Potential use of oxidative enzymes for the detoxification of organic pollutants. Appl Catal B Environ 46:1–15. https://doi.org/10.1016/S0926-3373(03)00228-5
Toyama T, Sato Y, Inoue D et al (2007) Enhanced production of manganese oxidizing enzymes from white-rot fungi in cross-linked aggregates. Environ Sci Technol 41:1033–1038. https://doi.org/10.1021/es0700977
Toyokawa C, Shobu M, Tsukamoto R et al (2016) Effects of overexpression of PKAc genes on expressions of lignin-modifying enzymes by *Pleurotus ostreatus*. Biosci Biotechnol Biochem 80:1759–1767. https://doi.org/10.1080/09168451.2016.1158630
Urek RO, Pazarlioglu NK (2007) Enhanced production of manganese peroxidase by *Phanerochaete chrysosporium*. Braz Arch Biol Technol 50:913–920. https://doi.org/10.1590/S1516-89132007000001
Urríola-Muñoz P, Lagos-Cabré R, Moreno RD (2014) A mechanism of male germ cell apoptosis induced by bisphenol-A and nonylphenol involving ADAM17 and p38 MAPK activation. PLoS One 9:1–27. https://doi.org/10.1371/journal.pone.0113793
Urríola-Muñoz P, Li X, Maretzky T et al (2017) The xenoeestrogens bisphenol-A and nonylphenol differentially regulate metalloprotease-mediated shedding of EGFR ligands. Cell Physiol 233:2247–2256
Válková H, Novotný Č, Malachová K et al (2017) Effect of bacteria on the degradation ability of *Pleurotus ostreatus*. Sci Total Environ 584–585:1114–1120. https://doi.org/10.1016/j.scitotenv.2017.01.071
Vargas-Berrones K, Díaz de León-Martínez L, Bernal-Jácome L et al (2020) Rapid analysis of 4-nonylphenol by solid phase microextraction in water samples. Talanta 209:120549. https://doi.org/10.1016/j.talanta.2019.120546
Verma ML, Kumar S, Das A et al (2020) Chitin and chitosan-based support materials for enzyme immobilization and biotechnological applications. Environ Chem Lett 18:315–323. https://doi.org/10.1007/s10311-020-00942-5
Vilas R, Watson CS (2013) Bisphenol S disrupts estradiol-induced nongenomic signaling in a rat pituitary cell line: effects on cell functions. Environ Health Perspect 121:352–358. https://doi.org/10.1289/ehp.1205826
Voběrková S, Solčány V, Vršanská M, Adam V (2018) Immobilization of ligninolytic enzymes from white-rot fungi in cross-linked aggregates. Chemosphere 202:694–707. https://doi.org/10.1016/j.chemosphere.2018.03.088
Vondrlik J, Vondrlička J, Pemovský P et al (2011) Synthesis, curing kinetics and thermal properties of bisphenol-AP-based benzoxazine. Eur Polym J 47:2158–2168. https://doi.org/10.1016/j.eurpolymj.2011.08.005
Wang J, Fang X, Wu MQ et al (2011) Synthesis, curing kinetics and thermal properties of bisphenol-AP-based benzoxazine. Eur Polym J 47:2158–2168. https://doi.org/10.1016/j.eurpolymj.2011.08.005
Wang J, Yamamoto R, Yamamoto Y et al (2013) Hydroxylation of bisphenol A by hyper lignin-degrading fungus *Phanerochaete sordida* YK-624 under non-ligninolytic conditions. Chemosphere 93:1419–1423. https://doi.org/10.1016/j.chemosphere.2013.07.026
Wang S, Wu W, Liu F et al (2015) Spatial distribution and migration of nonylphenol in groundwater following long-term wastewater irrigation. J Contam Hydrol 177–178:85–92. https://doi.org/10.1016/j.jconhyd.2015.03.013
Wiesenberg D, Kyniakkides I, Agathos SN (2003) White-rot fungi and their enzymes for the treatment of industrial dye effluents. Biotechnol Adv 22:161–187. https://doi.org/10.1016/j.biotechadv.2003.08.011
Wiersielis K, Samuels B, Roepeck T (2020) Perinatal exposure to bisphenol A at the intersection of stress, anxiety, and depression. Pharmacol Res 155:104743. https://doi.org/10.1016/j.phrs.2020.104743
Wu NC, Seebacher F (2020) Effect of the plastic pollutant bisphenol A on the biology of aquatic organisms: a meta-analysis. Glob Chang Biol: 3821–3833. https://doi.org/10.1111/gcb.15127
Wu J, Yu HQ (2007) Biosorption of 2,4-dichlorophenol by immobilized white-rot fungus *Phanerochaete chrysosporium* from aqueous solutions. Bioresour Technol 98:253–259. https://doi.org/10.1016/j.biortech.2006.01.018
Wu D, Feng Q, Xu T et al (2018a) Electrospray blend nanofiber membrane consisting of polyurethane, amidoxyime polycyclonitril, and B-cyclodextrin as high-performance carrier/support for efficient and reusable immobilization of laccase. Chem Eng J 331:517–526. https://doi.org/10.1016/j.cej.2017.08.129
Wu L, Zhang X, Wang F et al (2018b) Occurrence of bisphenol S in the environment and implications for human exposure: a short review. Sci Total Environ 615:87–98. https://doi.org/10.1016/j.scitotenv.2017.09.194
Wu F, Zhao J, Zhang E et al (2020a) Bisphenol A affects ovarian development in adolescent mice caused by genes expression change. Gene 740:–144535. https://doi.org/10.1016/j.gene.2020.144535
Wu W, Li M, Liu A et al (2020b) Bisphenol A and the risk of obesity: a systematic review with meta-analysis of the epidemiological evidence. Dose-Response 18:1–10. https://doi.org/10.1177/1559358190916949
Xiao CH, Kondo R (2020) Biodegradation and biotransformation of pentachlorophenol by wood-decaying white rot fungus *Phlebia acanthocystis* TMIC34875. J Wood Sci 66:2. https://doi.org/10.1590/s1516-848820190063
Xue J, Kannan K (2019) Mass flows and removal of eight bisphenol analogs, bisphenol A diglycidyl ether and its derivatives in two wastewater treatment plants in New York State, USA. Sci Total Environ 648:442–449. https://doi.org/10.1016/j.scitotenv.2018.08.047
Yamazaki E, Yamashita N, Taniyasu S et al (2015) Bisphenol A and other bisphenol analogues including BPS and BPF in surface water samples from Japan, China, Korea and India. Ecotoxicol Environ Saf 122:565–572. https://doi.org/10.1016/j.ecoenv.2015.09.029
Yang S, Liu Y, Yan K et al (2017) Bisphenol analogues in surface water and sediment from the shallow Chinese freshwater lakes: occurrence, distribution, source apportionment, and ecological and human health risk. Chemosphere 184:318–328. https://doi.org/10.1016/j.chemosphere.2017.06.010
Yang S (2012) Removal of micropollutants by a fungus-augmented membrane bioreactor.
Yang S, Hai FL, Nghiem LD et al (2013a) Understanding the factors controlling the removal of trace organic contaminants by white-rot fungi and their lignin modifying enzymes: a critical review.
Yang S, Hai FI, Nghiem LD et al (2013b) Removal of trace organic contaminants by nitrifying activated sludge and whole-cell and crude enzyme extract of 
Trametes versicolor. Water Sci Technol 67:1216–1223. https://doi.org/10.2166/wst.2013.684
Yang Y, Lu L, Zhang J et al (2014a) Simultaneous determination of seven bisphenols in environmental water and solid samples by liquid chromatography-electrospray tandem mass spectrometry. J Chromatogr A 1328:26–34. https://doi.org/10.1016/j.chroma.2013.12.074
Yang J, Li H, Ran Y, Chan K (2014b) Distribution and bioconcentration of endocrine disrupting chemicals in surface water and fish bile of the Pearl River Delta, South China. Chemosphere 107:439–446. https://doi.org/10.1016/j.chemosphere.2014.01.048
Yonten V, Ince M, Tanyol M, Yildirim N (2016) Adsorption of bisphenol A from aqueous solutions by Pleurotus eryngii immobilized on Amberlite XAD-4 using as a new adsorbent. Desalination 57:22362–22369. https://doi.org/10.1080/19443994.2015.1130659
Yoon Y, Westerhoff P, Snyder SA et al (2007) Removal of endocrine disrupting compounds and pharmaceuticals by nanofiltration and ultrafiltration membranes. Desalination 202:16–23. https://doi.org/10.1016/j.desal.2005.12.033
Yu Y, Wu L, Chang AC (2013) Seasonal variation of endocrine disrupting compounds, pharmaceuticals and personal care products in wastewater treatment plants. Sci Total Environ 442:310–316. https://doi.org/10.1016/j.scitotenv.2012.10.001
Yu J, Li W, Tang L et al (2020) In vivo and in vitro effects of chronic exposure to nonylphenol on lipid metabolism. Environ Sci Eur 32:87. https://doi.org/10.1186/s12302-020-00364-z
Zafar S, Aqil F, Ahmad I (2007) Metal tolerance and biosorption potential of filamentous fungi isolated from metal contaminated agricultural soil. Bioresour Technol 98:2557–2561. https://doi.org/10.1016/j.biortech.2006.09.051
Zdarta J, Antecka K, Frankowski R et al (2018) The effect of operational parameters on the biodegradation of bisphenols by Trametes versicolor laccase immobilized on Hippospongia communis sponge scaffolds. Sci Total Environ 615:784–795. https://doi.org/10.1016/j.scitotenv.2017.09.213
Zenata O, Dvorak Z, Vrzal R (2017) Profiling of bisphenol S towards nuclear receptors activities in human reporter cell lines. Toxicol Lett 281:10–19. https://doi.org/10.1016/j.toxlet.2017.09.006
Zhang Y, Geißen SU (2012) Elimination of carbamazepine in a non-sterile fungal bioreactor. Bioresour Technol 112:221–227. https://doi.org/10.1016/j.biortech.2012.02.073
Zhang A, Li Y (2014) Removal of phenolic endocrine disrupting compounds from waste activated sludge using UV, H2O2, and UV/H2O2 oxidation processes: effects of reaction conditions and sludge matrix. Sci Total Environ 493:307–323. https://doi.org/10.1016/j.scitotenv.2014.05.149
Zhang R, Liu R, Zong W (2016) Bisphenol S interacts with catalase and induces oxidative stress in mouse liver and renal cells. J Agric Food Chem 64:6630–6640. https://doi.org/10.1021/jacs.jacs.6b02656
Zhang L, Wei C, Zhang H, Song M (2017) Criteria for assessing the ecological risk of nonylphenol for aquatic life in Chinese surface fresh water. Chemosphere 184:569–574. https://doi.org/10.1016/j.chemosphere.2017.06.035
Zhang Y, Ni Z, Yao J (2020) Enhancement of the activity of electrochemical oxidation of BPS by Nd-doped PbO2 electrodes: performance and mechanism. Water 12:1–16
Zhu J, Jiang L, Liu Y et al (2015) MAPK and NF-κB pathways are involved in bisphenol A-induced TNF-α and IL-6 production in BV2 microglial cells. Inflammation 38:637–648
Zhu Y, Qiu F, Rong J et al (2020) Covalent laccase immobilization on the surface of poly(vinylidene fluoride) polymer membrane for enhanced biocatalytic removal of dyes pollutants from aqueous environment. Colloids Surf B: Biointerfaces 191:111025. https://doi.org/10.1016/j.colsurfb.2020.111025
Zielińska M, Bulkowska K, Cydzik-Kwiatkowska A et al (2016) Removal of bisphenol A (BPA) from biologically treated wastewater by microfiltration and nanofiltration. Int J Environ Sci Technol. https://doi.org/10.1007/s13762-016-1056-6
Żygo M, Prochoń M (2017) Enzymatyczne metody otrzymywania nanowłókien celulozowych. Wybrane pełne teksty z tego czasopisma 1:26–29

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