Bacterial degradation of bisphenol analogues: an overview

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Abstract

Bisphenol A (BPA) is one of the most produced synthetic monomers in the world and is widespread in the environment. BPA was replaced by bisphenol analogues (BP) because of its adverse effects on life. Bacteria can degrade BPA and other bisphenol analogues (BP), diminishing their environmental concentrations. This study aimed to summarize the knowledge and contribute to future studies. In this review, we surveyed papers on bacterial degradation of twelve different bisphenol analogues published between 1987 and June 2022. A total of 102 original papers from PubMed and Google Scholar were selected for this review. Most of the studies (94.1%, n = 96) on bacterial degradation of bisphenol analogues focused on BPA, and then on bisphenol F (BPF), and bisphenol S (BPS). The number of studies on bacterial degradation of bisphenol analogues increased more than six times from 2000 (n = 2) to 2021 (n = 13). Indigenous microorganisms and the genera Sphingomonas, Sphingobium, and Cupriavidus could degrade several BP. However, few studies focussed on Cupriavidus. The acknowledgement of various aspects of BP bacterial biodegradation is vital for choosing the most suitable microorganisms for the bioremediation of a single BP or a mixture of BP.

Keywords Emerging contaminants · Endocrine disruptor · Xenobiotic · Consortium · Mixtures · Bacterial community

Introduction

Bisphenol A (BPA; 4-[2-(4-hydroxyphenyl)propan-2-yl]phenol) is one of the synthetic monomers most produced and used in the world (Michałowicz 2014). The wide use of BPA accounted for an annual production of 6 kilotons in 2021 (Mordor Intelligence 2022). It is estimated that BPA production will increase by 6% between 2022 and 2027 (Mordor Intelligence 2022). The Russian A. P. Dianin was the first to synthesize this compound in 1891. BPA was synthesized by the reaction of two molecules of phenol with acid-catalysed acetone (Alexander and Dill 1998; Rubin and Soto 2009). In 1930, BPA was a potential candidate for synthetic estrogen because of its estrogenic activity. However, the industry chose diethylstilbestrol (DES) instead of BPA (Dodds and Lawson 1938; Rubin and Soto 2009). In the 1940s and 1950s, BPA was introduced in the plastic industry (Rubin and Soto 2009). Nowadays, this compound can be used to manufacture polycarbonates, epoxy resins, flame retardants, and paper thermal, among other uses (Geens et al. 2012).

BPA is continuously released into the environment because of its widespread production. It was detected in all environmental compartments, including soil, air, sediment, and water (Corrales et al. 2015; Crain et al. 2007; Flint et al. 2012; Michałowicz 2014). However, BPA is a xenobiotic and endocrine disruptor. This means BPA is a synthetic substance foreign to the natural environment (Ramírez-García et al. 2018) and an exogenous chemical substance that interferes with the hormonal action of the body (Alexander and Dill 1998; Chen et al. 2002; Kawa et al. 2021; Tarafdar et al. 2022; Vandenberg et al. 2013). BPA was detected in urine, amniotic fluid, and tissues like blood, breast milk, and liver of humans (Calafat et al. 2008; Fernandez et al. 2007; Geens...
et al. 2012; Huang et al. 2018b; Mercogliano and Santonicolola 2018; Yamada et al. 2002; Ye et al. 2009). In vitro studies and animal experiments indicate that exposure to BPA can affect reproductive, cardiovascular, and thyroid function (Catenza et al. 2021; Ma et al. 2019). Moreover, BPA may be related to oxidative stress, and metabolic diseases, including obesity and diabetes (Catenza et al. 2021; Ma et al. 2019). Besides, several adverse effects of BPA in wildlife were reported, including inhibition of development, malformations, and changes in the reproductive system (Akram et al. 2021; Chen et al. 2017; Crain et al. 2007; Falcão et al. 2020; Flint et al. 2012; Huang et al. 2018a; Kim et al. 2020; Moreman et al. 2017; Mu et al. 2018; Yang et al. 2021; Zhang et al. 2022). Consequently, authority regulations restricted the use of BPA. Then, BPA was replaced by substances of a similar chemical structure called bisphenol analogues (BP) (Chen et al. 2016; Lee et al. 2015).

Most toxicity studies still focus on BPA (Pelch et al. 2019). However, bisphenol analogues are not safer than BPA. Several bisphenol analogues listed in Table 1 have estrogenicity and toxicity similar to or greater than BPA, including the BPAF, BPF, BPB, and BPS (Chen et al. 2016; Chen et al. 2002; Gao et al. 2022; Lee et al. 2019; Tišler et al. 2016; Yang et al. 2017). In humans, bisphenol analogues were associated with diseases such as cancer and diabetes, oxidative stress, and other complications, including obesity and asthma (Catenza et al. 2021; Pelch et al. 2019). Therefore, given the emergency of BP harming living organisms, it is essential to understand the fate of bisphenol analogues in the environment, their toxicity, and biodegradability.

**Bisphenol analogues (BP)**

The bisphenol analogues (BP) are composed of two phenol rings (Hu et al. 2019; Chen et al. 2016; Lee et al. 2015). They can be applied to several materials including polyamides, polyesters, and polymers (Chen et al. 2016; Noszczyńska and Piotrowska-Seget 2018). In the manufacture of poly-carbonates, BPS, BPF, and BPAF are the main substitutes for BPA (Chen et al. 2016). Consequently, BPA substitution resulted in the wide distribution of the analogues in environmental compartments including sludges and effluents from treatment plants, sediment, fresh and marine water, dust, and soil (Česen et al. 2018; Chen et al. 2016; Chen et al. 2020; Hu et al. 2019; Lalwani et al. 2020; Pérez et al. 2017; Sun et al. 2017; Wang et al. 2022a, b; Wang et al. 2022b; Xie et al. 2022; Yamazaki et al. 2015). For example, BPS was found in surface water at a concentration of 24.8 ng/L, BPAF at 10.1 ng/L, and BPF at 9.0 ng/L (Wang et al. 2022a, b).

The fate of bisphenol in the environment is dictated by its water solubility (Table 1). Consequently, the hydrophobic compounds adsorb in soil, and the hydrophilic ones dissolve in water (Česen et al. 2018; Wang et al. 2019a). Thereby, some methods to remove BP from the environment include adsorption, photocatalysis, ozonation, oxidation, electro-chemical methods, and membrane separation (Godiya and Park 2022). The advantages of these approaches include low cost, easy preparation, efficient process, and ability to remove contaminants. Nonetheless, they also have drawbacks such as high energy consumption, the generation of sludge and additional pollution, and the formation of hazardous intermediates (Godiya and Park 2022). Therefore, biodegradation is the best choice for removing organic pollutants from the environment (Zhang et al. 2013) and will be the centre of this review.

**Biodegradation of bisphenol**

Biodegradation is an eco-friendly method with higher efficiency than traditional ones (Godiya and Park 2022). Biodegradation involves the break of compounds into simpler molecules called metabolites (Godiya and Park 2022; Ramírez-García et al. 2018). Metabolites often have a molecular structure less recalcitrant than the original molecule. Ideally, the mineralization end product should be simple chemical compounds such as water and carbon dioxide (Godiya and Park 2022; Kumar et al. 2018).

Bacterial degradation of BPA is the most relevant in biodegradation studies of bisphenol analogues (Eltoukhy et al. 2020; Zhang et al. 2013, 2007). However, several organisms including fungi, algae, and plants can also degrade BPA in the environment (Eio et al. 2015; Im and Löfler 2016; Michałowicz 2014; Zhang et al. 2019a). In bacterial biodegradation, the process relies on bacteria metabolism (Zhang et al. 2013). The bacteria benefit from molecules as a carbon source and substrate for the generation of bacterial energy (Godiya and Park 2022; Kumar et al. 2018; Noszczyńska and Piotrowska-Seget 2018).

Likewise, the release of aromatic compounds in the environment promotes specific enzymatic pathways in bacteria to degrade and use these compounds as an energy source (Cydzik-Kwiatkowska et al. 2021; Ramírez-García et al. 2018). However, biodegradation rates can be slow and rely on the bioavailability of compounds and bacterial species (Godiya and Park 2022). Besides, environmental factors such as temperature, pH, and supply of nutrients can disturb biodegradation (Eltoukhy et al. 2020; Godiya and Park 2022; Kumar et al. 2018; Ren et al. 2016; Singh et al. 2020). To diminish or improve the removal of contaminants, biodegradation can be associated with bioaugmentation and other methods of degradation (Singh et al. 2020).

The number of studies approaching bisphenol analogues and their potential hazard to the biota is increasing over the years. Nevertheless, bacterial biodegradation of bisphenol A is still predominant (Björnsdotter et al. 2017; Naderi et al.
Table 1  Bisphenol analogues and their respective chemical structures, nomenclature, solubility, and CAS number (Björnsdotter et al. 2017; PubChem 2021a, b, c, d, e, f, g, h, i, j, k, l; Sigma-Aldrich 2021a, b, c, d, e, f, g, h, i, j, k)

| Compound | Chemical structure | Nomenclature (IUPAC) | Solubility in water (g/L) | CAS       |
|----------|--------------------|----------------------|---------------------------|-----------|
| BPA      | ![BPA structure](image) | 4-[(4-hydroxyphenyl)propy]phenol | 120-300 | 80-05-7 |
| BPF      | ![BPF structure](image) | 4-[(4-hydroxyphenyl)methyl]phenol | 190 | 620-92-8 |
| BPS      | ![BPS structure](image) | 4-[(4-hydroxyphenyl)sulfonyl]phenol | $1.1 \times 10^{-3}$ | 80-09-1 |
| BPAF     | ![BPAF structure](image) | 4-[1,1,1,3,3,3-hexafluoro-2-(4-hydroxyphenyl)propan-2-yl]phenol | Insignificant | 1478-61-1 |
| BPB      | ![BPB structure](image) | 4-[(4-hydroxyphenyl)butan-2-yl]phenol | < 1.0 | 77-40-7 |
| BPE      | ![BPE structure](image) | 4-[(4-hydroxyphenyl)ethyl]phenol | n.d | 2081-08-5 |
| BPC      | ![BPC structure](image) | 4-[(2,2-dichloro-1-(4-hydroxyphenyl)ethenyl]phenol | 4.7 | 14868-03-2 |
| BPP      | ![BPP structure](image) | 4-[(2-[4-(4-hydroxyphenyl)propan-2-yl]phenyl)propan-2-yl]phenol | n.d | 2167-51-3 |
| BPZ      | ![BPZ structure](image) | 4-[(4-hydroxyphenyl)cyclohexyl]phenol | n.d | 843-55-0 |
| BPAH     | ![BPAH structure](image) | 4-[(4-hydroxyphenyl)-1-phenylethyl]phenol | 1.1 | 1571-75-1 |
| BPPH     | ![BPPH structure](image) | 4-[(4-hydroxy-3-phenylphenyl)propan-2-yl]-2-phenylphenol | n.d | 24038-68-4 |
| BPM      | ![BPM structure](image) | 4-[(2,3-[2-(4-hydroxyphenyl)propan-2-yl]phenyl)propan-2-yl]phenol | n.d | 13585-25-0 |
2014; Noszczyńska and Piotrowska-Seget 2018; Tišler et al. 2016; Usman et al. 2019; Yang et al. 2017). BPA degrading bacteria have been isolated from compartments such as water, soil, sediment, and water treatment plants. However, studies regarding the environmental persistence and fate of bisphenol analogues are scarce (Chen et al. 2016). Therefore, this review aims to summarize the published information on bacterial biodegradation of bisphenol analogues. Furthermore, we hope this survey can contribute to the advance of this discussion, pointing out emerging trends that should be addressed in future studies.

**Methods**

This survey included papers published between 1987 and June 2022 and available in PubMed and Google Scholar (G-Scholar) databases. The searches included the keywords “biodegradation” AND “bisphenol A”, “bisphenol F” and “biodegradation”; “bisphenol S” AND “biodegradation”; “bisphenol E” AND “biodegradation”; “bisphenol B” AND “biodegradation”; “bisphenol P” AND “biodegradation”; “bisphenol Z” AND “biodegradation”; “bisphenol AP” AND “biodegradation”; “bisphenol PH” AND “biodegradation”; “bisphenol M” AND “biodegradation”; “bisphenol A analogues” AND “biodegradation”.

We analysed the titles and abstracts of the papers retrieved and selected those with adherence to the objective of this study. Reviews, books, preprints, and conference abstracts were excluded. Only original studies in English were selected for this survey. The inclusion criteria were experimental research comprising the ability of bacteria to degrade bisphenol analogues. Thus, we excluded studies about (i) biodegradation with fungi; (ii) biodegradation with algae; (iii) biodegradation with isolated enzymes; (iv) different compounds; (v) monitoring; (vi) sorption and desorption; (vii) detection and occurrence; (viii) ecotoxicology; (iv) bioreactor and wastewater treatment plant (WWTP) membranes; (x) biodegradation articles about different compounds together with some bisphenol analogue; (xi) wetlands; (xii) phytoremediation; (xiii) photodegradation; and (xiv) human health. After this trial, we excluded duplicate articles.

**Results and discussion**

**Overview of BP biodegradation publishing**

Table 2 also includes the total number (n) and percentage (%) of papers restricted to bacterial biodegradation, and the number (n) of duplicated articles. The keywords “biodegradation” AND “bisphenol A” retrieved 24,070 papers on the G-Scholar search. However, only 85 (0.4%) were papers about bacterial biodegradation, and 59 were repeated in the PubMed list (Table 2).

A total of 102 original papers on bacterial bioremediation were selected for this review. The selected papers were listed in Online Resource, sheet S1 (de Morais Farias and Krepsky 2022). Figure 1 reveals an increasing trend in the number of studies about BP biodegradation in the past years (p = 0.0044). The first study investigating the biodegradation of BPA and its analogues was published in 1992 (Lobos et al. 1992). A second study about the biodegradation of bisphenol analogues was published 14 years later (Ike et al. 2006). After this publishing hiatus, the number of studies on bacterial degradation of bisphenol analogues increased more than six times from 2000 (n = 2) to 2021 (n = 13) (Fig. 1). In 2020, there was an increase in studies concerning the biodegradation of different bisphenol analogues, including BPS, BPF, BPB, BPE, BPAP, BPZ, BPM, BPPH, and BPAP (Fig. 1). However, the years 2021 and 2019 were the most productive for BP. A total of 13 papers were published in 2021 and 2019 concerning all analogues of bisphenol, especially BPA, BPS, and BPF (Fig. 1).

The evidence of BPA risk to human and environmental health may have encouraged researchers to investigate ways of removing this compound from the environment. Furthermore, the increase in BPA interest also influenced relevant authority decisions. For example, in 2006 the European Food Safety Authority (EFSA) issued a risk assessment opinion on BPA use. The panel members established as tolerable a daily consumption of 50 μg kg\(^{-1}\) of BPA (EFSA 2015). In 2013, 7 years after this issue, studies started to establish a thorough assessment of BPA risk. The EFSA evaluation included the quantification of exposure from non-dietary sources. The most vulnerable groups of the population, including pregnant women, babies, and children, were the focus of these studies (EFSA 2015). In 2015, the maximum acceptable daily intake of BPA from dietary and non-dietary sources was diminished from 50 to 4 μg kg\(^{-1}\) (EFSA 2015). In August 2015, the US Environmental Protection Agency (US EPA) released a list of 19 replacements, including other BP, to replace BPA in thermal papers (US EPA 2015). In December 2016, the European Commission has added BPA to the list of restricted substances (EU 2018). Later, in January 2017, a decree limited the concentrations of BPA in thermal papers to 0.02% after January 2, 2020 (Björnsdotter et al. 2017). In 2018, emerged the first restrictions for the use of BPA in canned food coatings (EU 2018). In Brazil, the Brazilian National Health Surveillance Agency (ANVISA) also banned the manufacture and import of baby bottles with
BPA by a Resolution published in 2011 (Brazil 2011). Thus, increasing awareness may explain the increasing number of publications on bacterial biodegradation of BP in the years 2007, 2015, 2017, 2019, 2020, and 2021 (Fig. 1).

**Bisphenol analogues**

Most (94.1%, \( n = 96 \)) of the publications analysed investigated the bacterial biodegradation of BPA (Fig. 2). The wide use of BPA in several products and its constant release in nature turned BPA into a ubiquitous compound (Flint et al. 2012; Oehlmann et al. 2009). Consequently, research efforts on the biodegradation of bisphenol analogues are still focussed on BPA. Nevertheless, the bisphenol analogues BPF and BPS are among the main substitutes for BPA in the manufacture of polycarbonates, epoxy resins, and thermal papers (Björnsdotter et al. 2017; Chen et al. 2016). Therefore, the detection of BPF and BPS in different environments is increasing yearly (Chen et al. 2016; Noszczyńska and Piotrowska-Seget 2018).

The increased environmental detection of BPF and BPS encouraged research on the bacterial biodegradation of these compounds. Indeed, our survey data showed that 19.6% and 18.6% of the publications explored the biodegradation of BPF and BPS, respectively (Fig. 2). However, this is not enough. Studies investigating the biodegradation of other BP should be encouraged. For example, BPC, BPZ, BPP, BPAF, and BPB have higher levels of toxicity and estrogenicity when compared to other BP (Chen et al. 2002; Gao et al. 2022; Yang et al. 2017; Zühlke et al. 2016, 2020). Besides, BPAF is an important BPA substitute in industry and can persist in the environment (Chen et al. 2016; Choi and Lee 2017; Choi et al. 2019; Frankowski et al. 2020; Zhou et al. 2020). The biodegradation of BPZ, BPE, and BPB analogues by indigenous microorganisms is not efficient enough (Frankowski et al. 2020; Ike et al. 2006; Zhou et al. 2017).

### Table 2

| Keywords                          | Database of search | Total number of papers (n) | Papers about bacterial biodegradation | Number of duplicated papers (n) |
|----------------------------------|--------------------|----------------------------|--------------------------------------|---------------------------------|
| “biodegradation” AND “bisphenol A” | PubMed             | 403                        | 63                                   | 59                              |
|                                  | G-Scholar          | 24,070                     | 85                                   | 0.4%                            |
| “bisphenol F” AND “biodegradation” | PubMed             | 21                         | 11                                   | 11                              |
|                                  | G-Scholar          | 1,312                      | 13                                   | 1.0%                            |
| “bisphenol S” AND “biodegradation” | PubMed             | 20                         | 12                                   | 12                              |
|                                  | G-Scholar          | 1,406                      | 15                                   | 1.1%                            |
| “bisphenol E” AND “biodegradation” | PubMed             | 5                          | 4                                    | 4                               |
|                                  | G-Scholar          | 136                        | 6                                    | 4.4%                            |
| “bisphenol B” AND “biodegradation” | PubMed             | 8                          | 6                                    | 6                               |
|                                  | G-Scholar          | 369                        | 9                                    | 2.4%                            |
| “bisphenol P” AND “biodegradation” | PubMed             | 2                          | 1                                    | 1                               |
|                                  | G-Scholar          | 71                         | 3                                    | 4.2%                            |
| “bisphenol Z” AND “biodegradation” | PubMed             | 3                          | 2                                    | 2                               |
|                                  | G-Scholar          | 97                         | 4                                    | 4.1%                            |
| “bisphenol C” AND “biodegradation” | PubMed             | 2                          | 1                                    | 1                               |
|                                  | G-Scholar          | 106                        | 4                                    | 3.8%                            |
| “bisphenol AF” AND “biodegradation” | PubMed             | 6                          | 3                                    | 3                               |
|                                  | G-Scholar          | 531                        | 6                                    | 1.1%                            |
| “bisphenol AP” AND “biodegradation” | PubMed             | 2                          | 1                                    | 1                               |
|                                  | G-Scholar          | 107                        | 3                                    | 2.8%                            |
| “bisphenol PH” AND “biodegradation” | PubMed             | 291                        | 1                                    | 1                               |
|                                  | G-Scholar          | 16                         | 1                                    | 0.3%                            |
| “bisphenol M” AND “biodegradation” | PubMed             | 35                         | 1                                    | 1                               |
|                                  | G-Scholar          | 2                          | 1                                    | 0.5%                            |
| “bisphenol A analogues” AND “biodegradation” | PubMed             | 424                        | 2                                    | 2                               |

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et al. 2020). Nevertheless, there are few studies about the biodegradation of these analogues (Fig. 2).

Furthermore, studies on bacterial degradation of mixed BP are scarce. Figure 3 shows the studies where bacterial biodegradation of bisphenol analogues was considered. We represented in *purple* all studies focussing on a single BP published until June 2022. Studies focussing exclusively on bacterial biodegradation of BPA were listed in Online Resource, sheet SI2 (de Morais Farias and Krepsky 2022). Bisphenol analogues are not isolated in the environment. Notably, only three studies analysed the biodegradation of mixtures of BP (Fig. 3). Therefore, studies of BP mixtures can guide the bioremediation in environments with more than one bisphenol analogue.

**Environmental compartments**

Figure 4 summarized the environmental compartments studied on bacteria biodegradation of bisphenol analogues until June 2022. Water was the compartment most studied \((n=25)\), including different analogues (Fig. 4). Considering the bisphenol analogues and the water compartment, BPA \((n=24)\) was the compound most analysed, followed by BPS \((n=7)\) and BPF \((n=6)\) (Fig. 4). Some authors also investigated the bacterial degradation of BPB \((n=5)\) and BPE \((n=5)\) in BPA studies (Frankowski et al. 2020; Ike et al.
Inoue et al. 2008; Sakai et al. 2007; Zhou et al. 2020) (Figs. 3 and 4). However, studies with the other BP were rare. For example, the analogues BPP (Inoue et al. 2008; Ike et al. 2006), BPAF (Frankowski et al. 2020; Zhou et al. 2020), and BPZ (Zhou et al. 2020; Sakai et al. 2007) were included in two studies (Fig. 4). Moreover, one study evaluated the bacterial biodegradation of BPC (Sakai et al. 2007) or BPM (Zhou et al. 2020) (Figs. 3 and 4).

The popularity of BPA in bacterial biodegradation studies in various environmental compartments can be attributed to its ubiquity and higher environmental concentration than other BP (Caban and Stepnowski 2020; Chen et al. 2020; Flint et al. 2012; Lalwani et al. 2020; Ozhan and Kocaman 2019; Peteffi et al. 2019; Wang et al. 2022a, 2022b; Xie et al. 2022). The most researched analogues coupled with BPA in water were BPF and BPS (Fig. 3). These three analogues (BPF, BPS, and BPA) were found in 53 surface water samples collected from different regions of India (Lalwani et al. 2020). They were also detected in water samples from China, Poland, Japan, and Korea (Caban and Stepnowski 2020; Yamazaki et al. 2015; Wang et al. 2022a, 2022b; Xie et al. 2022). In China, those three analogues were detected in the Pearl River (Chen et al. 2020). BPS was the third predominant analogue in Chinese seawater (Xie et al. 2022), although, in surface water, BPS was the second most predominant after

| References          | BP | B | P | A | B | P | S | B | P | F | B | P | E | B | P | A | F | B | P | C | B | P | M | B | A | P | B | P | H |
|---------------------|----|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Lu et al. 2022      |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Moreira et al. 2021 |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Kovačič et al. 2021 |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Frankowski et al. 2021 |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Qian et al. 2021    |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Cao et al. 2020     |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Elthouky et al. 2020|    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Frankowski et al. 2020 |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Zhou et al. 2020    |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Zühlke et al. 2020  |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Takeo et al. 2020   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Choi et al. 2019    |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Huang et al. 2019   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Wang et al. 2019    |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Choi and Lee 2017   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Lu et al. 2017      |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Ren et al. 2016     |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Zühlke et al. 2016  |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Chang et al. 2014   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Ogata et al. 2013   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Toyama et al. 2013  |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Toyama et al. 2009  |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Danzl et al. 2009   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Inoue et al. 2008   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Sakai et al. 2007   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Ike et al. 2006     |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |
| Lobos et al. 1992   |    |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |   |

Fig. 3 Colour scheme showing the studies investigating bacterial biodegradation of the bisphenol analogues published between 1987 and June 2022. Purple indicates studies with single BP; orange indicates studies of bacterial biodegradation with mixtures of BP. Legend: BPA, bisphenol A; BPS, bisphenol S; BPF, bisphenol F; BPP, bisphenol P; BPC, bisphenol C; BPBH, bisphenol PH; BPM, bisphenol M; BPAP, bisphenol AP.
BPA (Wang et al. 2022a). In Poland, BPS was detected in concentrations similar to BPA found in Gdansk (Caban and Stepnowski 2020). Besides, BPF was also reported in higher concentrations than BPA in Southeastern Asian rivers, including Chinese surface waters (Wang et al. 2022b; Yamazaki et al. 2015).

Afterwards, water treatment plants were the second compartment most investigated (n = 21) in studies for biodegradation of BP (Fig. 4). In the water treatment plant category, we included studies on activated sludge and effluents. Activated sludge is the most popular technique for water treatment plants. This technique supports microbial communities able to remove nutrients and xenobiotics (Noszczyńska and Piotrowska-Seget 2018). Because of its high nutrient richness, the sludge can be used as a fertilizer. However, polluted fertilizers can contaminate the soil, and groundwater, and reach the food chain (Wang et al. 2019a, b). Consequently, some studies focused on the degradation of BP in sludge. Three authors studied the biodegradation of BPA and BPF (Frankowski et al. 2020; Lobos et al. 1992; Zühlke et al. 2016). Two papers included the biodegradation of the analogues BPAF and BPS (Frankowski et al. 2020; Choi et al. 2019), BPB (Lobos et al. 1992; Zühlke et al. 2016), BPZ (Lobos et al. 1992; Zühlke et al. 2016), and BPE (Frankowski et al. 2020; Zühlke et al. 2016), and one study included the biodegradation of BPC (Zühlke et al. 2016). One paper studied the biodegradation of the analogues BPF and BPS (Kovačič et al. 2021). Nevertheless, another paper investigated the biodegradation of BPF isolated (Lu et al. 2022) (Fig. 4).

Fig. 4 Heat map scheme of environmental compartments where bacteria biodegradation of bisphenol analogues was studied in papers published between 1987 and June 2022. The colour scale represents the number of published papers on the subject. Others include microorganisms from fermented food, landfill, and the human gut. Legend: BPA, bisphenol A; BPS, bisphenol S; BPF, bisphenol F; BPB, bisphenol B; BPE, bisphenol E; BPZ, bisphenol Z; BPAF, bisphenol AF; BPP, bisphenol P; BPC, bisphenol C; BPPH, bisphenol PH; BPM, bisphenol M; BPAP, bisphenol AP.

Effluents from treatment plants can be the primary source of BP in the environment, including BPA (Caban and Stepnowski 2020; Sun et al. 2017; Wang et al. 2022a, b; Zhang et al. 2019b). BPA, BPF, and BPS were predominant in water treatment plant sludge from Korea, China, and India (Lalwani et al. 2020; Hu et al. 2019; Sun et al. 2017). Besides, these three BP dominate water and sewage sludge samples from other countries (Wang et al. 2019a, b; Yamazaki et al. 2015). Lalwani et al. (2020) associated the highest concentration (14,800 ng L−1) of BPA in an Indian river with the discharge of untreated sewage effluent (Lalwani et al. 2020). Conversely, the analogues including BPB, BPZ, and BPAP were hardly reported in sludges (Hu et al. 2019). The analogues BPB, BPZ, BPAP, BPP, and BPAF were detected in sewage treatment plants in India (Karthikraj and Kannan 2017). In Slovenia, the analogue BPZ was detected in effluent samples in higher concentrations than other BP, including BPB, BPC, BPE, BPAP, and BPAF (Česen et al. 2018). In China, the sludge of water treatment plants in Xiamen City presented the highest concentrations of the analogues BPA, BPF, and BPS, including BPAF and BPE (Sun et al. 2017). Pieces of evidence report that the analogues BPB, BPE, BPAP, and BPAF are little degraded in treatment plants (Sun et al. 2017; Wang et al. 2019a; Česen et al. 2018). Therefore, research efforts to investigate the biodegradation of other BP and their ideal conditions, especially in sewage and industrial effluents, are imperative.

Nevertheless, Cydzik-Kwiatkowska et al. (2020) observed direct degradation and removal of bisphenol A...
in wastewater by aerobic granular sludge (AGS). AGS are more efficient than the activated sludge. It yields an operational economy of energy and a wastewater footprint reduction of up to 50% (Cydzik-Kwiatkowska et al. 2020). The AGS have round and dense granules with different layers of self-immobilized microbial communities. In each layer, there is a microenvironment with different oxygen conditions and bacterial species able to degrade distinct pollutants (Cydzik-Kwiatkowska et al. 2020). Therefore, AGS is a promising substitute technology for the activated sludge system (Cydzik-Kwiatkowska et al. 2020) and needs further attention for BP degradation.

Afterwards, sediment \((n = 19)\) was the third compartment most analysed on BP biodegradation (Fig. 4). Likewise, BPA was the most investigated analogue in sediment \((n = 18)\). Among the studies regarding BPA biodegradation in sediment, one included the BPF analogue (Chang et al. 2014) and the other the BPS (Moreira et al. 2021) (Fig. 3). Additionally, one single study (Fig. 3) focussed on the bacterial degradation of BPS analogue in sediments (Wang et al. 2019b). The presence of BPA in sediments was detected in various locations including India, Italy, and China (Li et al. 2019; Mukhopadhyay et al. 2020; Pignotti and Dinelli 2018; Wang et al. 2022b; Xie et al. 2022). A previous study reported higher concentrations of BPA in the sediment than in the water column (Flint et al. 2012). Conversely, Huang et al. (2012) related BPA concentrations in sediment with higher concentrations of BPA in water. Yamazaki et al. (2015) observed that BPF can deposit in the sediment, suggesting that more hydrophobic BP can accumulate in sediment. Nonetheless, the eventual release of bisphenol analogues from the sediment into the water can increase its concentration in the underlying water (Chen et al. 2020). Therefore, studies considering different BP should focus on sediments.

The soil was the fourth most researched compartment for BP biodegradation \((n = 15)\) and BPA was the most studied analogue \((n = 14)\). Nonetheless, after the water compartment, the soil was the second compartment with papers investigating biodegradation of different bisphenol analogues (Fig. 4). For example, coupled with BPA, four studies on bacterial degradation in soil included BPS (Choi and Lee 2017; Eltoukhy et al. 2020; Frankowski et al. 2020; Oshiman et al. 2007) and BPF analogues (Eltoukhy et al. 2020; Frankowski et al. 2020; Ren et al. 2016; Zühlke et al. 2020). Three papers studied the degradation of BPF analogue in soil (Eltoukhy et al. 2020; Oshiman et al. 2007; Zühlke et al. 2020). A single study (Fig. 4) focussed on the bacterial degradation of BPAF in soil (Choi and Lee 2017). Furthermore, few authors focused on the degradation of different bisphenol analogues in soil, excluding BPA from the investigation. Cao et al. (2020) studied the BPS. Lu et al. (2017) investigated the BPF. Likewise, Zühlke et al. (2020) analysed the biodegradation of seven bisphenol analogues, including BPB, BPC, BPE, BPF, BPZ, BPAP, and BPPH (Fig. 3). However, these authors published a previous paper addressing BPA biodegradation in soil under the same experimental conditions (Zühlke et al. 2017).

Bisphenol analogues can reach soil from the discharge of landfill leachate, application of sewage sludge and biosolids as fertilizers, irrigation with wastewater effluents, and waste disposal (Corrales et al. 2015; Flint et al. 2012). Pérez et al. (2017) detected BPA in higher concentrations than other BP in the soil. In addition, these authors detected BPF in samples of soil from an industrial source and agricultural land irrigated with water recycled from a treatment plant (Pérez et al. 2017). Conversely, the analogue BPAF was detected only in the samples from agricultural land, but not in industrial soils (Pérez et al. 2017). Consequently, given the different sources of soil contamination and potential crop production contamination, further studies considering the biodegradation of other bisphenol analogues in soil are vital for food safety.

Last but not least, twelve studies \((n = 12)\) investigated the bacterial biodegradation of bisphenol analogues in a bioreactor (Fig. 4). Again, BPA was the most popular bisphenol analogue in bioreactor studies. Eleven studies \((n = 11)\) focussed on BPA degradation. One paper included the analogues BPB and BPF (Chang et al. 2014) and another studied analogues BPA, BPF, BPS, BPB, BPE, BPZ, BPAP, and BPAF (Qian et al. 2021) (Fig. 3). Moreover, one single paper (Huang et al. 2019) focussed on BPS biodegradation (Figs. 3 and 4). The bioreactor is one of the methods approved for treatment stations. Bioreactor allows a longer retention time of the compost to be treated, higher biomass concentration, and, consequently, higher bacterial density, favouring biodegradation (Hu et al. 2019). Besides, bioreactors can be fed, for example, with water, sludge, and other matrices (Chang et al. 2014; Huang et al. 2019; Oh and Choi 2019; Sathyamoorthy et al. 2018). Therefore, it is important to identify the microorganisms capable of degrading bisphenols under the operating conditions characteristic of bioreactors, including the optimization of this process.

The number of papers published in each environmental compartment varied over time (Fig. 5). Water was the most studied compartment concerning the biodegradation of bisphenol analogues from 1987 to 2009 (Fig. 5). BPA has a water solubility of 120–300 mg/L at room temperature (Staples et al. 1997) and started to be detected in aquatic environments in the late 1990s (Corrales et al. 2015). Together, both factors may explain why the studies of bacterial biodegradation in water were carried out so
readily. Meanwhile, studies with other environmental compartments intensified after 2014, including sediment, soil, treatment plant, and bioreactor (Fig. 5). All these compartments are the source or destination of bisphenol analogues to the environment. Thus, approaching bacterial biodegradation of diverse bisphenol analogues in different compartments is imperative to understand this process and control their damage.

**Bacterial biodegradation**

One hundred and nine bacterial strains capable of BPA degradation were reported in the literature and listed in sheet SI3 of Online Resource (de Morais Farias and Krepsky 2022). Figure 6 summarizes the dominant groups of bacteria and indigenous microorganisms in studies on the biodegradation of BP between 1987 and June 2022.

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**Fig. 5** Number of papers per year considering bacterial biodegradation of bisphenol analogues in each environmental compartment published between 1987 and June 2022. Others include microorganisms from fermented food and laboratory culture.

**Fig. 6** Number of bisphenol analogues assessed in biodegradation studies by bacterial groups and indigenous microorganisms studied in papers published between 1987 and June 2022. Legend: BPA, bisphenol A; BPS, bisphenol S; BPF, bisphenol F; BPB, bisphenol B; BPE, bisphenol E; BPZ, bisphenol Z; BPAF, bisphenol AF; BPP, bisphenol P; BPC, bisphenol C; BPPH, bisphenol PH; BPM, bisphenol M; BPAP, bisphenol AP
Indigenous microorganisms were the most studied group regarding bacterial biodegradation (Fig. 6). The main genera investigated for BPA degradation were *Pseudomonas*, *Sphingomonas*, and *Bacillus* (Fig. 6). Besides BPA, *Sphingomonas* species could biodegrade up to six bisphenol analogues, including BPS, BPF, BPB, BPE, BPZ, and BPC (Fig. 6). Likewise, *Cupriavidus* also biodegraded six different BP, besides BPA (Fig. 6). Meanwhile, *Sphingobium* was able to degrade six analogues including BPA, BPS, BPF, BPB, BPE, and BPZ (Fig. 6). *Bacillus* biodegradation comprised degradation of five analogues—BPA, BPF, BPE, BPZ, and BPC (Fig. 6)—and *Pseudomonas* presented degradation of only four analogues—BPA, BPS, BPF, and BPB (Fig. 6).

**Pseudomonas**

Despite the degradation of a limited list of bisphenol analogues (Fig. 6), *Pseudomonas* was the most studied genus for the degradation of BPA. *Pseudomonas* is known for its ability to degrade a variety of organic molecules, including aromatic compounds like toluene, biphenyl, naphthalene, phthalates, and others (Diaz et al. 2008; Goldberg 2000; Kimura et al. 2018; Kim and Park 2018; Palleroni 2015, 2010; Yu et al. 2020). This genus is an aerobic, Gram-negative, rod-shaped bacteria that can be found in different environments and are flexible to environmental changes (Palleroni 2015). Consequently, it was observed that a *Pseudomonas* consortium had the best percentages of degradation of BPA, BPF, and BPB (Chang et al. 2014). Likewise, *Pseudomonas* was predominant in a consortium able to degrade BPS (Huang et al. 2019; Wang et al. 2019b) and was identified in a consortium that degraded BPF (Lu et al. 2017). Moreover, when the strain *Pseudomonas* sp HS-2 was isolated from this consortium, it was still efficient in BPF degradation (Lu et al. 2017).

*Pseudomonas* also benefits plants, although it can be pathogenic to humans, animals, and plants (Palleroni 2015; Goldberg 2000). For example, *P. aeruginosa* is an opportunistic pathogen that can cause several infections and is resistant to several antibiotics (Hao et al. 2021). *P. aeruginosa* species can be used to improve BPA degradation, either by isolating it in nanofibre membranes (Liu et al. 2015) or by using it in bioreactors (Mita et al. 2015). *P. aeruginosa* Gb30 was able to degrade 60% of BPA in a concentration of 3 mM in 4 days (Louati et al. 2019). However, in the same study, the strain *P. putida* G320 presented the highest BPA degradation efficiency than other strains isolated from arid and desert soil (Louati et al. 2019). The Mn(II) oxidizing strain *P. putida* GB-1 could not degrade BPA in the absence of Mn(II). However, this strain degraded and enhanced the BPA degradation in the presence of Mn(II). Besides, the degradation using *P. putida* GB-1 with 10 μM of Mn(II) was more efficient than the degradation with the chemically synthesized manganese dioxide (Shobnam et al. 2021). Another study showed that the strain *Pseudomonas putida* YC-AE1, isolated from a soil sample, can degrade BPA at high (50, 100, 200, 300, 400, 500, 600, 700, 800, 900, 1000 mg/L) and low (0.5, 1, 2, 4, 6, 8, 10, 12 mg/L) concentrations (Eltoukhy et al. 2020). This strain showed variable efficiency of degradation according to the BP tested. For example, in a concentration of 100 mg/L, biodegradation of BPA in 72 h was 100%, BPS 30%, BPF 67%, and BPB 60% (Eltoukhy et al. 2020). Conversely, *P. putida* isolated from river water degraded in 10 days 87% of BPA in the concentration of 1 mg/L (Kang and Kondo 2002b). Besides, *Pseudomonas* sp. BG12 degraded only 35% of BPA in a concentration of 100 mg/L at pH 7 and 60% at pH 8 within 15 days, reaching 52 ± 4.58% of degradation with the addition of sodium glutamate (Noszczyńska et al. 2021). In contrast, the strain *P. palleroniana* GBPI_508 was able to degrade 97% of BPA in a concentration of 270 mg/L in 96 h (Thathola et al. 2022).

**Sphingomonas**

*Sphingomonas* was the second genus most studied on BP biodegradation until June 2022. *Sphingomonas* consists of aerobic, rod-shaped or ovoid, Gram-negative bacteria that can be mobile or not (Yabuuchi and Kosako 2015). They can reside in natural or modified environments and can be opportunistic pathogens (Yabuuchi and Kosako 2015). *Sphingomonas* possesses a wide metabolic versatility (Yabuuchi and Kosako 2015) and can degrade several recalcitrant aromatic compounds, including biphenyl, chlorinated furan, carbazole, chlorinated phenols, polyethylene glycol, different herbicides, and pesticides, in addition to endocrine disruptors such as estradiol and nonylphenol (Asaf et al. 2020; Yabuuchi and Kosako 2015; Stolz 2009; Willison 2004). Likewise, this genus can degrade BPA and other BP such as BPS, BPF, BPB, BPE, BPZ, and BPC (Fig. 6).

Regarding bisphenol analogues, the *Sphingomonas* sp BP-7 could degrade approximately 90%, 100%, 94%, and 100% of BPE, BPB, BPC, and BPZ, respectively, at a concentration of 100 mg/L (Sakai et al. 2007). However, this strain was not able to degrade BPF and BPS. Sakai et al. (2007) suggested that the biodegradation of BP relies on the methyl or methylene group present between the two aromatic rings in some BP. Although present in BPA, neither methyl nor methylene group is present in the BPF or BPS analogues (Table 1). Consequently, *Sphingomonas* degradation of BPA was reported in many studies. For example, the strains isolated from soil *Sphingomonas* sp SO11, *Sphingomonas* sp SO1a, and *Sphingomonas* sp SO4a could metabolize BPA at a concentration of 115 mg/L in a period of 12 to 48 h (Matsumura et al. 2009). Additionally, *Sphingomonas* sp. SP-7 can degrade 95% of BPA in a concentration of 30 mg/L in
40 days (Sakai et al. 2007). However, total BPA degradation could be much faster by coupling Sphingomonas sp. SP-7 with Pseudomonas sp. BP-14. For example, in 7 days this consortium degraded 100% of BPA in the concentration of 100 mg/L (Sakai et al. 2007). Likewise, increased BPA degradation was observed in a consortium of Sphingomonas and Pseudomonas (Yu et al. 2019). Moreover, the S. bisphenolicum strain AO1 degraded between 30 and 100% of BPA at a concentration of 100 mg/L in 44 h, according to the glucose concentration in the growth medium (Oshiman et al. 2007). The metabolism of Sphingomonas bisphenolicum AO1 was also analysed in other studies (Sasaki et al. 2005a, b; Sasaki et al. 2008). As a result, this strain increased the community of indigenous microorganisms and the efficiency of BPA degradation (Matsumura et al. 2015).

**Bacillus**

*Bacillus* was the third bacteria genus most studied on BP degradation, after *Pseudomonas* and *Sphingomonas* (Fig. 6). The genus *Bacillus* comprised aerobic, or anaerobic facultative, rod-shaped, Gram-positive bacteria. These bacteria can occur individually, in pairs, in chains, or as long filaments (Logan and Vos 2015). They are present in several environments such as freshwater, seawater, soil, air, plants, and animals. *Bacillus* can form endospores and resist adverse environmental conditions, including desiccation, radiation, and chemical substances (Logan and Vos 2015). Besides, *Bacillus* can overcome extreme conditions including high temperatures, acidic environments, and extremes of salinity (Logan and Vos 2015; Maughan and Van Der Auwera 2011). Some studies reported that *Bacillus* can degrade BPA, BPF, BPE, BPZ, and BPC (Fig. 6). However, most research focused on the *Bacillus* efficiency of BPA degradation. For example, the strain isolated from soil *Bacillus* sp YA27 took 60 h to degrade 100% of BPA at a concentration of 50 mg/L (Matsumura et al. 2009). Nonetheless, the strain *Bacillus* sp. KU3, isolated from the marine environment, showed 61% of efficiency in the degradation of BPA at a concentration of 1000 mg/L in 15 days (Kamaraj et al. 2014). Therefore, the degradation efficiency decreased as the concentration of BPA increased (Li et al. 2012).

In lower concentrations of BPA (10 and 25 mg/L), the strain *Bacillus pumilus*, isolated from fermented Kimchi food, were able to degrade 100% of BPA in 16 h and 3 days, respectively. Likewise, *B. pumilus* grown in a medium increased by 10% of NaCl degraded BPA at a concentration of 10 mg/L in 2 days. However, this strain was not able to degrade BPA at the concentration of 50 mg/L and 10 mg/L with more than 12.5% NaCl (Yamanaka et al. 2007). Additionally, the probiotic bacterium *B. subtilis* degraded 51.9% of BPA at a concentration of 50 μg/mL in 96 h (Kyrila et al. 2021). Moreover, some research reported increased BPA biodegradation associating *Bacillus* species with the macrophyte *Dracaena sanderiana*. For example, *B. thuringiensis* isolated from the endosphere of the plant *D. sanderiana* degraded 95% of BPA at a concentration of 100 μM in 24 h (Suyamud et al. 2018a, b). Likewise, the association of *Bacillus cereus* NL, *D. sanderiana*, and endophyte strains yielded a removal rate of 100% BPA in 5 days (Suyamud et al. 2020). Both strains belong to the *Bacillus cereus* group. *B. thuringiensis* is entomopathic and is used in the production of pesticides (Ehling-Schulz et al. 2019). *B. cereus* is responsible for poisoning food and infections (Ehling-Schulz et al. 2019).

Some studies associated the cytochrome P450 monoxygenase system with the hydroxylation of BPA during biodegradation with *Bacillus* sp. GZB (Das et al. 2018). *Bacillus* sp. GZB could degrade in 96 h, 100% of BPA at a concentration of 10 mg/L under aerobic and anaerobic conditions (Li et al. 2012). Genes encoding the cytochrome P450 monoxygenase system were detected in the genome of *Bacillus* sp. GZB (Das et al. 2018). The cytochrome P450 monoxygenase system comprises cytochrome P450, ferredoxin, and ferredoxin reductase. Das et al. (2018) affirmed this enzyme complex is vital for BPA degradation. The same investigation was conducted with *Sphingomonas bisphenolicum* AO1, and the cytochrome P450 monoxygenase system was also involved in the BPA biodegradation (Sasaki et al. 2005a, b; Sasaki et al. 2008). Additionally, it was reported that *Bacillus* produces other enzymes including the spore-laccase enzyme, which was responsible for the BPA biotransformation into less complex molecules (Das et al. 2018). Indeed, the genus *Bacillus* produces laccase and this enzyme are capable of catalysing the oxidation of aromatic compounds (Lu et al. 2012), including phenol and other recalcitrant compounds (Das et al. 2018; Lu et al. 2012; Le et al. 2006; Held et al. 2005).

Regarding the other BP, the *B. amyloliquefaciens* strain was able to degrade 77% of BPA, 69% of BPF, and 77% of BPE in concentrations of 60 mg/L (Zühlke et al. 2016). Likewise, *B. amyloliquefaciens* degraded 95% of BPC at a concentration of 20 mg/L (Zühlke et al. 2016). The degradation of BPZ by *B. amyloliquefaciens* was also investigated (Zühlke et al. 2016). However, BPZ was poorly soluble and only a small portion (one-sixth) of the soluble phase could be detected at the high-performance liquid chromatography (Zühlke et al. 2016). The degradation of bisphenol analogues resulted in the formation of bisphenol and phosphate conjugates. The authors suggested that this may be a mechanism to reduce the toxicity of BP and thus avoid the growth inhibition of the strain. However, the transformation of BP in these products proved to be reversible because after the formation of the products they returned to the initial structure of BP (Zühlke et al. 2016).
Sphingobium

*Sphingobium* was another important genus in the biodegradation of BP (Fig. 6). *Sphingobium* is one of three new genera derived from *Sphingomonas*, including *Novosphingobium*, *Sphingomonas stricta* sensu, and *Sphingopyxis* (Takeuchi et al. 2001). A fifth genus *Sphingosinicella* was proposed later (Maruyama et al. 2006). *Sphingobium* is characterized by grouping strictly aerobic, rod-shaped, Gram-negative bacteria with glycosphingolipids in their cell envelope, and chemoorganotrophic (Takeuchi et al. 2001). Members of this genus are capable of degrading aromatic compounds such as naphthalene, biphenyl, m-xylene, phenanthrene, herbicides, and pesticides (Cai et al. 2015; Liang and Lloyd-Jones 2010; Pinyakong et al. 2003; Révész et al. 2018; Önneby et al. 2014).

Furthermore, *Sphingobium* could degrade most bisphenol analogues, including BPA, BPF, BPS, BPB, BPE, and BPP (Fig. 6). For example, the strains *Sphingobium fuliginis* TIK1 and *Sphingobium* sp. IT4 were both isolated from the rhizosphere of plants and degraded 0.5 mmol/L of those six BP with 100% degradation efficiency in 24 h. However, the biodegradation efficiency of BPP was 78% for *S. fuliginis* TIK1 and 91% for *Sphingobium* sp. IT4 (Toyama et al. 2013). *Sphingobium* sp. IT4 degradation of BP resulted in the hydroquinone and p-benzoquinone metabolites. Meanwhile, *Sphingobium fuliginis* TIK1 generated metabolites resulting from hydroxylation and meta cleavage of BP, which were consistent with the findings from BPA degradation by *Sphingobium fuliginis* OMI (Toyama et al. 2013). Figure 7 was modified from Ogata et al. (2013) and describes two metabolic pathways for BPA degradation by *Sphingobium fuliginis* IMO. The metabolites (a) 3-hydroxy BPA; (c) 3-(4-hydroxyphenyl)-3-methyl-2-butanone; (d) 2,2-bis(3,4-dihydroxyphenyl) propane; and (f) 2,2-bis and 3-(3,4-dihydroxyphenyl)-3-methyl-2-butanone indicated the hydroxylation of one or two aromatic rings and further meta-cleavage of this molecule (Ogata et al. 2013). Therefore, it was proposed from this metabolic pathway that bisphenol like BPF and BPS could be degraded regardless of the chemical group present in the connection between the aromatic rings (Ogata et al. 2013). In addition, *Sphingobium fuliginis* OMI, also isolated from the rhizosphere, showed 100% of degradation efficiency for almost all BP cited in a
concentration of 1 mM in 24 h. However, the BPP for this strain was the exception and obtained about 67% degradation (Ogata et al. 2013).

In another study with strains isolated from the rhizosphere, the Sphingobium strain yanoikuyae TYF-1 was able to degrade 90% and 92% of BPA and BPF, respectively, at a concentration of 25 mg/L, over a long period of 42 days (Toyama et al. 2009). The products that originated from BPF degradation were ditrimethylsilyl (4HB), hydroquinone (1,4-HQ), and p-benzoquinone (1,4-BQ). It was suggested that Sphingobium strain yanoikuyae TYF-1 had a BPF degradation pathway similar to the Sphingobium yanoikuyae FM-2 strain (Toyama et al. 2009). Indeed, the Sphingobium yanoikuyae FM-2 isolated from riverine water was able to degrade 100% of BPF at a concentration of 0.5 mM in 9 h when previously acclimated with the BPF. This efficiency decreased to 95% of BPF in 16 h when acclimated with glucose. Toyama et al. (2009) proposed a degradation pathway in which the connection between the BPF rings undergoes a rearrangement, releasing 1,4-hydroquinone and p-hydroxybenzoic acid. Then, both generated metabolites could be completely degraded later (Inoue et al. 2008). However, despite BPF degradation, the Sphingobium yanoikuyae FM-2 could not degrade other BP including BPA, BPE, BPB, and BPS (Inoue et al. 2008). Therefore, Inoue et al. (2008) suggested that the Sphingobium yanoikuyae FM-2 strain could only degrade the BP with no methyl groups on the connection between the aromatic rings or in the aromatic rings. Therefore, there were differences in metabolic pathways for both strains Sphingobium fuliginis TIK1 and Sphingobium fuliginis OMI (Ogata et al. 2013; Toyama et al. 2009).

Cupriavidus

Despite few studies with Cupriavidus, this bacteria genus was relevant to BP degradation (Fig. 6). Some members of the Cupriavidus genus can resist heavy metals, and synthesize polychlorinated biphenyls and degrade xenobiotics (Wang et al. 2017). For example, the Cupriavidus basilensis is capable of degrading xenobiotics including biphenyl, dibenzofuran, ochratoxin A, 9H-carbazol, and others (Becher et al. 2000; Ferenczi et al. 2014; Suenaga et al. 2015; Waldau et al. 2009; Wang et al. 2017). Likewise, Cupriavidus basilensis could degrade BPA, BPE, BPB, BPC, and BPS (Fig. 6).

Regarding BPA degradation, Cupriavidus basilensis JF1 showed slow degradation of BPA as the single source of carbon. However, BPA degradation was accelerated with the addition of phenol as a co-substrate. Phenol acted as a degradation biostimulant. Almost 66% of BPA in the concentration of 0.21 mM was degraded in 150 h when phenol was added (Fischer et al. 2010). Similarly, the strain C. basilensis SBUG 290 obtained higher BPA degradation efficiency when previously cultivated with biphenyl, achieving 78% degradation of 0.26 mM in 48 h (Zühlke et al. 2017). Thus, Zühlke et al. (2017) cultivated C. basilensis strains in biphenyl to carry out degradation experiments with BPF, BPE, BPP, BPC, BPAP, and BPPH (Zühlke et al. 2020). Cupriavidus basilensis SBUG 290 showed 98% efficiency in the degradation of BPC, 62% of BPB, 31% of BPE, and 6% of BPF, in 216 h, at a concentration of 60 mg/L (Zühlke et al. 2020). Conversely, the low solubility of BPZ, BPAP, and BPPH made it impossible to investigate the degradation efficiency in Cupriavidus basilensis SBUG 290 (Zühlke et al. 2020). Investigations regarding the metabolic pathways for BP cleavage by C. basilensis were published previously and will not be discussed in our review. To our knowledge, the BP metabolic pathway is related to its chemical structure and involves bacterial actions in the aromatic ring, such as hydroxylation, oxidation, and cleavages (Zühlke et al. 2020, 2017; Fischer et al. 2010).

Indigenous microorganisms

Finally, the group of indigenous microorganisms was also related to the biodegradation of BP (Fig. 6). The biodegradation of BPA, BPP, BPE, BPB, and BPF by indigenous microorganisms was previously observed in freshwater (Ike et al. 2000; Ike et al. 2006; Kang and Kondo 2002a; Klecka et al. 2001). Accordingly, indigenous microorganisms were able to degrade BPA, BPE, BPB, BPZ, and BPF in the concentration of 0.1 mg/L in freshwater (Zhou et al. 2020). However, each bisphenol analogue presented different biodegradation efficiencies. For example, the analogues BPA, BPE, BPF, and BPS showed 70% of degradation. Nevertheless, BPP and BPM degraded, respectively, 60% and 30% (Zhou et al. 2020).

Moreover, the biodegradation of BPA, BPF, BPS, BPP, and BPE by indigenous microorganisms was evaluated in riverine water and activated sludge (Frankowski et al. 2020). Regarding the aerobic biodegradation of BPS in water, it was not detected in riverine water (Frankowski et al. 2021; Ike et al. 2006), nor in lake water after 49 days (Zhou et al. 2020) or in marine water within 60 days (Danzl et al. 2009). Conversely, BPS was degraded in a wastewater treatment plant in the concentration range of 0.1 to 5 mg/L (Kovačič et al. 2021). Moreover, BPS degradation reached 91% efficiency within the 28 days of consortium enrichment (Moreira et al. 2021). Additionally, 99% of BPS at a concentration of 50 mg/L was degraded in 10 days by a consortium isolated from a sediment microbial community after 28 days of acclimation (Wang et al. 2019b).

The biodegradation of indigenous microorganisms from activated sludge samples reached approximately 100% efficiency for BPA and BPF at the concentration of 10 mg/L (Frankowski et al. 2020). At this same concentration, biodegradation of BPS was between 40 and 50% and around
40% for BPB and BPE. However, the degradation of bisphenol analogues by indigenous microorganisms in riverine water was inefficient (Frankowski et al. 2020). For example, the efficiency of BPAF biodegradation was less than 20% under the same conditions at the concentration of 10 mg/L (Frankowski et al. 2020). Likewise, BPAF showed high persistence in lake water and its concentration remained unchanged after 49 days of monitoring (Zhou et al. 2020). This can indicate that BPM, BPS, and BPAF were not easily degraded in the water. Nevertheless, increased adsorption of BPM and BPAF to humic acid can increase its degradability (Zhou et al. 2020). Indeed, BPAF showed the highest affinity for humic acid and activated sludge particles than BPA, BPS, BPB, and BPM (Zhou et al. 2020; Choi et al. 2019).

Conversely, the removal of eight bisphenol analogues was studied in a bioreactor for 23 days at different pHs (Qian et al. 2021). The analogues BPA, BPS, BPB, and BPE reached the highest degradation efficiency at pH 7.0 and 9.0. At pH 7.0 their degradation ranged approximately from 98.5 to 88.6% and at pH 9.0 from 98.7 to 88.1% (Qian et al. 2021). However, the removal efficiency of the analogues BPZ, BPAF, and BPAP at pH 7.0 shifted approximately from 71.3 to 45.3% and from 61.6 to 32.8% at pH 9.0 (Qian et al. 2021). Therefore, chemical differences in the analogue structure (Qian et al. 2021) or microbial community can influence their removal efficiency from the bioreactor (Chen et al. 2022; Huang et al. 2017; Tong et al. 2021; Wang et al. 2021; Xiong et al. 2017).

Indigenous microorganisms comprise the microbial community native to the environment. Consequently, the addition of pollutants like BPS or BPA can modify the composition, diversity, and abundance of these microbial communities. For example, BPA can favour bacterial groups resistant to xenobiotics and inhibit sensible ones (Chen et al. 2022; Huang et al. 2017; Tong et al. 2021; Wang et al. 2021; Xiong et al. 2017). The acclimatization of bacteria with BPS for 28 days selected tolerant bacteria, including *Hyphomicrobium*, *Pandoraea*, and *Cupriavidus*, that could use this BP as a substrate for growth (Wang et al. 2019a, b). A similar observation was found during biodegradation in a bioreactor (Huang et al. 2019). BPS was degraded in 10 days and the microbial community was modified over this period time. Accordingly, there was an increase in the abundance of bacteria associated with BPS degradation, including *Pseudomonas, Devosia, Delftia, Acidovorax*, and *Rhodobacter* (Huang et al. 2019).

In addition to the presence of xenobiotics, environmental conditions interfere with bacterial metabolisms (Eltoukhy et al. 2020; Godiya and Park 2022; Kumar et al. 2018; Ren et al. 2016; Singh et al. 2020).

| Compound | Matrix | Microorganism | $T$ (°C) | pH | Reference |
|----------|--------|---------------|---------|----|-----------|
| BPA      | River water | Indigenous microorganisms | 30      | 7  | Kang and Kondo 2002a |
| BPA      | Water marine sediment | *Pseudomonas sp. KU1* | n.d    | 7  | Kamaraj et al. 2014 |
| BPA      | Water marine sediment | *Pseudomonas sp. KU2* | n.d    | 7  | Kamaraj et al. 2014 |
| BPA      | Water marine sediment | *Bacillus sp. KU3* | n.d    | 7  | Kamaraj et al. 2014 |
| BPA      | Seawater | *Croceicoccus bisphenolivorans* sp. nov | 32      | 7  | Li et al. 2021 |
| BPA      | Soil | *Arthrobacter sp. YC-RL1* | 30      | 7  | Ren et al. 2016 |
| BPA      | Soil | *Pseudomonas putida YC-AE1* | 25–30   | 7.2 | Eltoukhy et al. 2020 |
| BPA      | Soil | *Pseudomonas sp. BG12* | n.d    | 8  | Noszczyńska et al. 2021 |
| BPA      | Soil | *Pseudomonas palleroniana GBPl_508* | 25      | 9  | Thathola et al. 2022 |
| BPA      | River sediment | *Sphingobium sp. YC-JY1* | 30      | 5.5–8 | Jia et al. 2020 |
| BPA      | River sediment | *Bacillus sp. GZB* | 37      | 7  | Li et al. 2012 |
| BPA      | Solid waste leachate | *Achromobacter xylosoxidans* | 35      | 7  | Zhang et al. 2007 |
| BPA      | Activated sludge, landfill, and water with sediment | *Acinetobacter sp. KIMN* | n.d    | 8  | Noszczyńska et al. 2021 |
| BPA      | Activated sludge, landfill, and water with sediment | *Pseudomonas sp. BG12* | n.d    | 8  | Noszczyńska et al. 2021 |
| BPF      | Soil | consortium | 35      | 7  | Lu et al. 2017 |
| BPS      | River sediment | consortium | 30      | 7  | Wang et al. 2019a, b |
| BPF      | Soil | consortium | 30      | 7  | Wang et al. 2019a, b |

Legend: BPA, bisphenol A; BPF, bisphenol F; BPS, bisphenol S
the available data on temperature and pH optimal for BP degradation by microorganisms and reference. However, investigations of the ideal conditions for bacterial degradation of bisphenols comprised only BPA, BPF, and BPS (Table 3). Papers that assess the ability of bacterial strains to degrade different concentrations of BP other than BPA are still scarce. Several studies have already identified differences in the efficiency of BPA degradation according to changes in compound concentration (Babatabar et al. 2019; Chang et al. 2011; Eltoukhy et al. 2020; Klecka et al. 2001; Li et al. 2012; Vijayalakshmi et al. 2018; Zhang et al. 2007). However, there are still few studies that investigate this difference in other bisphenol analogues (Table 3).

Conclusions

Most of the studies published until June 2022 focussed on the bacterial degradation of BPA, and then on the biodegradation of BPF and BPS. Indeed, BPA was the most studied analogue for all compartments and bacterial genera. Water was the preferred environmental compartment to access bacterial biodegradation of BPA and other analogues. Biodegradation studies of BPA on water treatment plants were the second most accessed after the water compartment. Furthermore, the soil was frequently accessed for the degradation of the other bisphenol analogues, whereas leachate was the compartment less studied. All the strains cited in published studies diminished bisphenol concentration in the environmental compartments. *Pseudomonas*, *Sphingomonas*, and *Bacillus* were the genera most investigated for biodegradation of BPA and other BP. Besides the number of publications, indigenous microorganisms, *Sphingomonas*, *Cupriavidus*, and *Sphingobium* showed the highest degradation variability. Those groups degraded more than six BP analogues, including BPA, BPB, and BPE. Meanwhile, *Pseudomonas* and *Bacillus* degraded up to three or four BP, respectively.

The degradation of BPA by bacterial consortia showed to be more efficient than with isolated strains (Chang et al. 2011; Kang and Kondo 2002b; Peng et al. 2015; Sakai et al. 2007; Sarma et al. 2019; Yu et al. 2019). However, only three studies evaluated the consortium degradation of an analogue different from BPA (Lu et al. 2017; Moreira et al. 2021; Wang et al. 2019b). Thereby, bacterial degradation efficiency varies greatly depending on the microorganisms involved in the process, the environmental conditions, and the chemical structure of the compound to be degraded. Accordingly, the biodegradation of bisphenol A does not guarantee the biodegradation of its analogue. Therefore, research comparing the degradation of isolated bacteria and consortia must be executed with other bisphenol analogues, considering the particularities of each microorganism and compartments.

Additionally, studies on the biodegradation of mixtures of bisphenol analogues are scarce. Nevertheless, these studies should combine other pollutants in the environment. Moreover, research analysing the degradation of bisphenol analogues with different chemical groups attached to the aromatic ring, such as BPAP and BPC, is rare. Consequently, studies covering various aspects of the bacterial biodegradation of bisphenol analogues are vital to better understanding its relationship with the chemical and molecular structures of each analogue. Future research efforts should focus on clarifying the capacity of bacterial strains or consortiums in degrading the different bisphenol analogues. This includes determining the optimum pH and temperature for biodegradation and the effect of previous bacteria acclimatization on each BP biodegradation, notably the hydrophobic analogues like BPAF and BPM in water compartments. All this knowledge could assist in the choice of the most suitable microorganisms for bioremediation whether it is a single BP or a mixture of BP.

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Data availability All datasets analysed in this study are available as spreadsheets in online resource (de Morais Farias and Krepsky 2022).

Declarations

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Consent to participate Not applicable.

Consent for publication Not applicable.

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