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Mechanisms and Potential for Fungal Bioremediation of Polycyclic Aromatic Hydrocarbons and Other Environmental Contaminants: A Comprehensive Review

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Abstract

Polycyclic aromatic hydrocarbons (PAHs) are present in soil, water, and air. The existence of PAHs in the environment is dangerous as they pose significant impacts on humans, plants, and animals. Fungal bioremediation is a sustainable approach to overcome environmental pollution by utilizing the metabolic capabilities of fungi to degrade or convert a wide range of pollutants, including PAHs. It is cost-effective, safer, and environmentally friendly. The review describes PAHs' structure, classes, sources, and toxicity. Different PAHs remediation techniques are also compared with mycoremediation. Because of their adaptability, fungi can be used for remediation in various environmental contexts, such as soil, water, and air. Fungi can degrade PAHs, converting them into various intermediate products, by secreting extracellular enzymes such as laccase, manganese peroxidase, and lignin peroxidase. Ligninolytic and non-ligninolytic fungi used in mycoremediation are also discussed. Many factors affect the efficiency of the biodegradation process of PAHs, including temperature, nutrient availability and pH of the media. This review also discusses a comprehensive overview of the applications of fungi in diverse contaminants such as dyes, heavy metals and xenobiotics, and *in-situ* and *ex-situ* bioremediation approaches. Finally, it discusses strategies, highlighting the limitations and challenges of fungal bioremediation.

Keywords – Anthracene – Degradation – Fungi – Mycoremediation – *Phanerochaete chrysosporium*

Introduction

Worldwide Polycyclic aromatic hydrocarbons (PAHs) Pollution

Over the last few decades, human initiatives have contributed to the significant release of hydrocarbons into the environment, making them some of the most prevalent and impactful pollutants globally (Ameh et al. 2023). Due to the incomplete combustion of organic materials, most importantly petroleum gas, oil, wood, and municipal and urban waste, Polycyclic Aromatic Hydrocarbons (PAHs) are formed (Agrawal et al. 2018, Zhang et al. 2024). An oil refinery can contribute to environmental contamination of PAHs through municipal and industrial discharge from pipelines, equipment and pipeline leaks, tanker accidents, and runoff from its operations (Balaji et al. 2014, Bayat et al. 2015, Abdullah et al. 2020). Through a variety of processes,

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including transportation, combustion of fossil fuel, biomass burning, incineration, and accidental spills, oil and its derivatives that contain PAHs damage the environment (Bayat et al. 2015). PAH pollution is a widespread environmental concern affecting both developed and developing countries, with significant implications for ecosystems and human health. Various studies have highlighted the presence and risks associated with PAH contamination in different environmental matrices across the world. A study by Meng et al. (2019) examined PAH pollution levels in water bodies and sediments in China. The findings indicated that while PAH concentrations in water were at moderate levels, naphthalene was the most dominant compound, posing a considerable risk to aquatic organisms in lakes. Although PAH levels in sediments were relatively low, certain compounds, such as acenaphthene, fluorene, and dibenz(a,h)anthracene, were identified as potentially hazardous (Meng et al. 2019). Similarly, Singh et al. (2025) investigated PAH distribution in soil samples from urban cities in Bihar, India, revealing that ingestion and dermal exposure pathways posed a greater risk than inhalation. Their study reported elevated cancer risk values, particularly for children (1.16×10^{-5}) and adults (1.03×10^{-5}) during the winter season, underscoring the health hazards associated with PAH contamination in urban environments (Singh et al. 2025).

In Bangladesh, a study conducted by Hasan et al. (2025) assessed PAH concentrations in the Shitalakshya River in Dhaka. The results identified significant ecological risks, with benzo(b)fluoranthene, benzo(k)fluoranthene, and indeno(1,2,3-cd)pyrene posing the greatest threat to aquatic life (Hasan et al. 2025). Further, Adeniran et al. (2025) investigated PAH contamination in dust samples collected from twenty vehicle parks in Lagos, Nigeria. The study found that four-ring PAHs were the most prevalent in road dust from Ibadan, with fluoranthene being the dominant compound. Among the carcinogenic PAHs analyzed, benzo[a]pyrene was identified as the most prevalent, highlighting the potential health risks associated with PAH exposure in urban traffic areas (Adeniran et al. 2025).

These studies collectively underscore the global extent of PAH pollution and its varying impacts across different environmental compartments, emphasizing the need for effective monitoring and mitigation strategies.

Definition of PAHs and Chemical Structure of PAHs

PAHs are organic compounds consisting of several aromatic rings fused (Suresh et al. 2025). PAHs are pale yellow solids or white solids that are organic. They are both natural and manmade and are often produced through the incomplete burning of biomass or fossil fuels (Silalahi et al. 2021, Suresh et al. 2025). In recent years, PAHs in the environment have drawn global concern due to their inherent properties and potential environmental and health risks (Abdullah et al. 2020, Ameh et al. 2023).

Carbon (C) and hydrogen (H) atoms grouped in two or more fused benzene rings make up the semi-volatile organic pollutants known as PAHs, which exist in large quantities in the environment (Godoy et al. 2016, Agrawal et al. 2018, Li et al. 2024). These compounds can be organized in linear, angular, or clustered configurations (Fig. 1) (Iwuoha & Oritsebinone 2024, Rezagholizade-shirvan et al. 2024). The chemical structure of PAHs reduces their bioavailability and biodegradability (Godoy et al. 2016, Okafor et al. 2024). As an example, PAHs with an angular arrangement tend to be more stable in comparison to a linear arrangement (Li et al. 2024, Rezagholizade-shirvan et al. 2024). Apart from C and H, some PAHs contain nitrogen (N), sulphur(S), and oxygen(O) atoms in their chemical structure (Agrawal et al. 2018, Rezagholizade-shirvan et al. 2024).

PAHs can exist in the environment for a long time due to their high hydrophobicity and low volatility, accumulating in the environment (Agrawal et al. 2018, Berríos-Rolón et al. 2024). The United States Environmental Protection Agency (USEPA) has categorized 16 PAHs as major pollutants (Agrawal et al. 2018, Ameh et al. 2023, Zhang et al. 2024). These PAHs have also been categorized as major pollutants by other major regulatory bodies, such as the European Union

(EU), the International Agency for Research on Cancer (IARC), and the Environmental Protection Agency (Okafor et al. 2024).

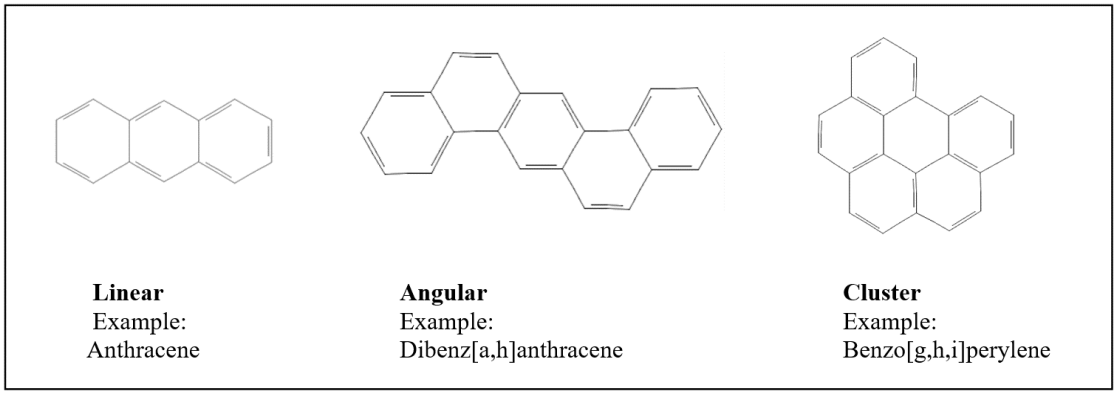


Fig. 1 – Linear, angular, and clustered structures of PAH

Classes of PAHs

PAHs can be categorized into two classes based on their molecular weight as below (Alao & Adebayo 2022, Dey et al. 2023, Barbosa et al. 2023, Femi-Oloye et al. 2024).

Low Molecular Weight Polyaromatic Hydrocarbons (LMW-PAHs)

PAHs that contain 2-3 benzene rings are considered low molecular weight polyaromatic hydrocarbons (LMW PAHs) (Alao & Adebayo 2022, Dey et al. 2023, Barbosa et al. 2023, Femi-Oloye et al. 2024). They cause inflammation and cytotoxicity, carcinogenic effects, asthma and respiratory effects (Sherris et al. 2024, Takam et al. 2024). Acenaphthene, acenaphthylene, anthracene, fluorine, naphthalene, and phenanthrene are some examples of LMW-PAHs whose structures are indicated in Table 1.

High Molecular Weight Polyaromatic Hydrocarbons (HMW-PAHs)

PAHs with 4-7 benzene rings are high molecular weight polyaromatic hydrocarbons (HMW PAHs) (Alao & Adebayo 2022, Dey et al. 2023, Femi-Oloye et al. 2024). Benzo[a]anthracene, chrysene, fluoranthene, and pyrene are some examples, as shown in Table 1(Godoy et al. 2016, Gupte et al. 2016).

PAHs are listed as hazardous by the USEPA due to the widespread contribution to the ecosystem and to human health that they make (Godoy et al. 2016, Barbosa et al. 2023). Table 1 represents the physical properties and chemical structures of the 16 USEPA priority PAHs. There are other PAH derivatives available in nature. The number of rings in PAHs' structure influences their molecular weights (MWs) and complexity. High molecular weight PAHs have significantly low vapour pressure and tend to attach to airborne particles. Low molecular weight PAHs have a higher vapour pressure, making them primarily present in the gaseous state. Compared to LMW PAHs, HMW PAHs are less volatile. Polycyclic aromatic hydrocarbons are resistant to degradation by heat and light. Therefore, LMW PAHs have lower boiling and melting points when compared to HMW PAHs. While they are largely insoluble in water, they dissolve readily in organic solvents.

PAH Pollution: Sources and Environmental Distribution

Fig. 2 illustrates the various sources and pathways through which PAHs are introduced into the environment, as well as their distribution across different environmental compartments. There are natural events and anthropogenic activities as sources for PAHs. Both dry deposition, such as settling particles, and wet deposition (rain) allow PAHs to reach the environment. These contaminants attach to particles in sediments and water bodies. Polycyclic aromatic hydrocarbons have health concerns for humans when they are consumed.

Table 1 Physical properties and chemical structures of the 16 USEPA priority PAHs (Patel et al. 2020, Alao & Adebayo 2022, Berríos-Rolón et al. 2024).

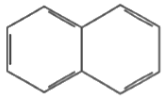
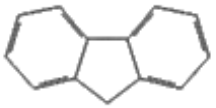
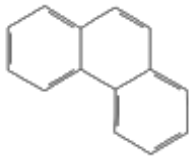
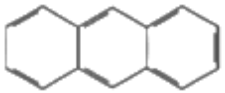
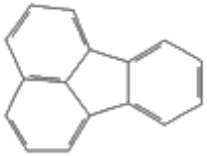
PAHs	Chemical formula	Chemical structure	Number of Aromatic rings	Classification	MW (g mol^{-1})	VP (Pa)	BP (ºC)	MP (ºC)	WS at 25 ºC (mg L^{-1})
Naphthalene	$C_{10}H_6$		2	LMW	128.17	11.866	209	80	31.0
Acenaphthylene	$C_{12}H_8$		3	LMW	152.19	0.500	290	124	16.1
Acenaphthene	$C_{12}H_{10}$		3	LMW	154.21	3.866	252	108	3.8
Fluorene	$C_{14}H_{10}$		3	LMW	166.22	0.432	276	119	1.9
Phenanthrene	$C_{14}H_{10}$		3	LMW	178.23	9.07×10^{-2}	326	136	1.1
Anthracene	$C_{14}H_{10}$		3	LMW	178.23	3.40×10^{-3}	326	136	0.045
Fluoranthene	$C_{20}H_{12}$		4	HMW	202.25	1.08×10^{-3}	369	166	0.26

Table 1 Continued.

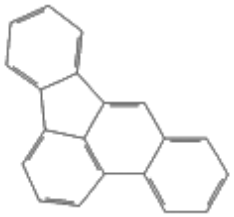
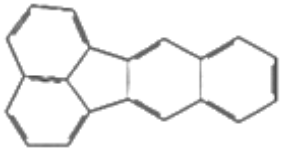
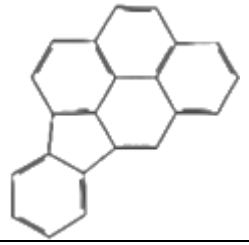
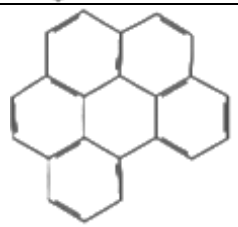
PAHs	Chemical formula	Chemical structure	Number of Aromatic rings	Classification	MW (g mol^{-1})	VP (Pa)	BP (ºC)	MP (ºC)	WS at 25 ºC (mg L^{-1})
Pyrene	$C_{16}H_{10}$		4	HMW	202.25	5.67×10^{-4}	369	166	0.132
Benzo[a]anthracene	$C_{20}H_{12}$		4	HMW	228.30	2.05×10^{-5}	400	177	0.011
Chrysene	$C_{18}H_{12}$		4	HMW	228.30	1.04×10^{-6}	400	177	0.0015
Benzo[b]fluoranthene	$C_{20}H_{12}$		5	HMW	252.30	1.07×10^{-5}	461	209	0.0015
Benzo[k]fluoranthene	$C_{20}H_{12}$		5	HMW	252.30	1.28×10^{-8}	430	194	0.0008

Table 1 Continued.

PAHs	Chemical formula	Chemical structure	Number of Aromatic rings	Classification	MW (g mol^{-1})	VP (Pa)	BP (ºC)	MP (ºC)	WS at 25 ºC (mg L^{-1})
Benzo[a]pyrene	$C_{20}H_{12}$		5	HMW	252.30	6.52 x 10^{-7}	461	209	0.0038
Indeno[1,2,3-cd] Pyrene	$C_{22}H_{12}$		5	HMW	276.30	1.87 x 10^{-8}	498	233	0.062
Dibenz[a,h]anthracene	$C_{22}H_{14}$		6	HMW	278.30	2.80 x 10^{-9}	487	218	0.0005
Benzo[g,h,i]perylene	$C_{22}H_{12}$		6	HMW	276.30	1.33 x 10^{-8}	467	218	0.00026

BP - Boiling point, HMW-PAHs - High molecular weight polycyclic aromatic hydrocarbons, LMW-PAHs - Low molecular weight polycyclic aromatic hydrocarbons, MP - Melting point, MW - Molecular weight, VP - Vapor pressure at 25 ºC, WS - Water solubility at 25 ºC

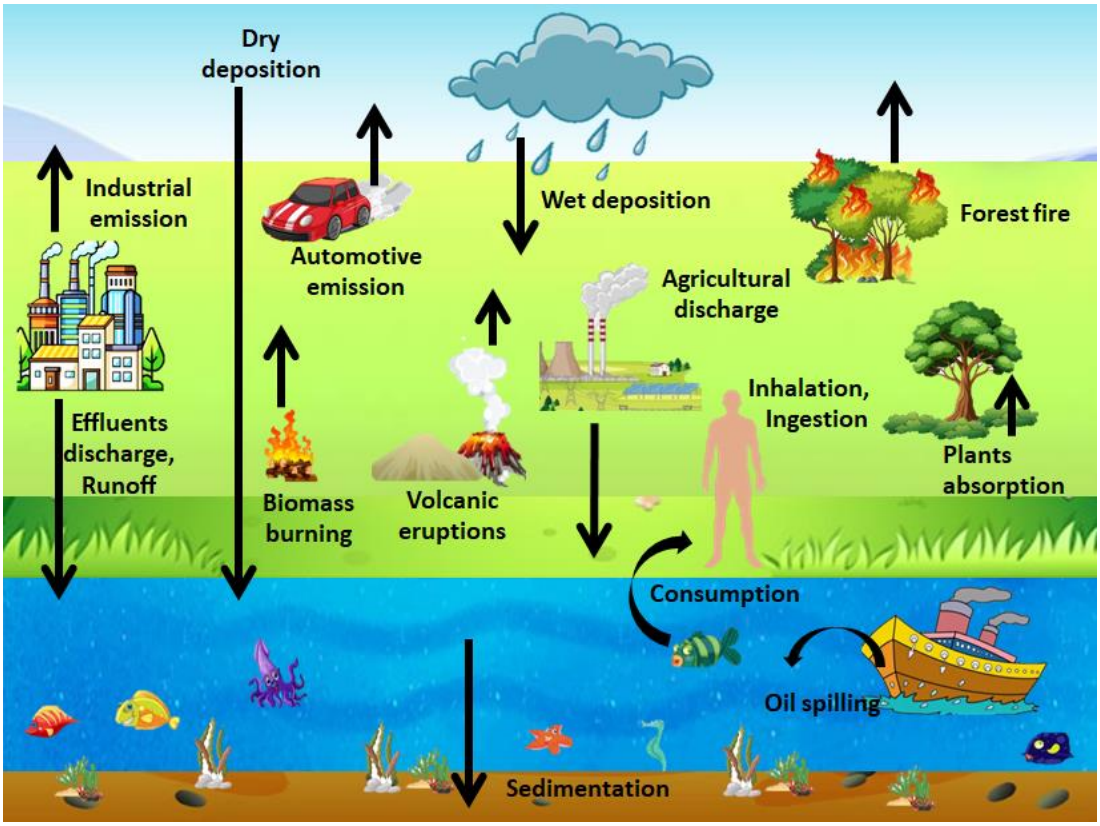


Fig. 2 – Sources of PAHs and distribution of PAHs in air, terrestrial, and aquatic environments

Sources of PAH Pollution

In the environment, PAHs are widely spread, e.g. in soil (Bayat et al. 2015, An et al. 2022), air (Ma et al. 2020) and water bodies (Okafor et al. 2024, Zhang et al. 2024). When PAHs are disposed into the environment, humans can be exposed to them through drinking water or direct contact with the skin (Okafor et al. 2024). Polycyclic aromatic hydrocarbons occur in natural and anthropogenic sources (Okafor et al. 2024) (Fig. 2).

Natural sources of PAH pollution are generally less significant than those caused by human activities. Natural sources include forest fires (Campos et al. 2019, Campos & Abrantes 2021, Gong & Wang 2021), volcanic eruptions (Guiñez et al. 2020, Jiao et al. 2024), and plants uptake and decomposition of PAHs and runoff from sediments (Witter & Nguyen 2016, Rocha & Palma 2019, Schwab & Dermody 2021).

PAHs are generated during wildfire combustion and released into the atmosphere. They can be re-distributed between the vapour and particulate phases. After transportation, PAHs deposit in aquatic as well as terrestrial systems through dry or wet deposition. PAHs can affect surface and groundwater bodies by leaching into the soil profile (Campos & Abrantes 2021, Gong & Wang 2021). PAHs are derived from frequent high-temperature volcanic combustion events (Jiao et al. 2024). Volcanic ashes also contain nitrated and oxygenated PAHs (Guiñez et al. 2020).

Plants uptake PAH from the soil through roots to shoots, from the soil through the air to shoots, and from the soil through airborne particles to shoots. Accumulation of PAHs in plants can cause ecological and health effects, impacting the whole food chain (Schwab & Dermody 2021). Root uptake and atmospheric deposition, followed by translocation, lead to the accumulation of PAHs in plants, contaminating foods (Abo-State et al. 2021, Jia et al. 2021, Liu et al. 2024).

Anthropogenic activities, including industrial processes, transportation, waste combustion, and domestic practices, are major contributors to environmental pollution. These activities release various pollutants, including polycyclic aromatic hydrocarbons (PAHs), which originate from the incomplete combustion of organic materials. PAHs are persistent organic pollutants that accumulate in air, soil, and water, posing significant environmental and health risks.

Anthropogenic-related sources of PAHs can be categorized as indoor and outdoor sources. Indoor sources primarily include household activities such as cooking and heating, which contribute to PAH emissions. Cooking with gas, kerosene, coal, or oils can significantly impact indoor air quality (Munyeza et al. 2020, Li et al. 2024). A study by Li et al. (2024) investigated PAH exposure from domestic heating and cooking combustion among elderly individuals in China. The study found that indoor PAH levels were higher than outdoor levels in houses using solid fuels. Additionally, while liquefied petroleum gas (LPG) users did not show significantly lower PAH concentrations compared to solid fuel users, biogas users exhibited the lowest PAH emissions, with reductions ranging from 64% to 82% (Li et al. 2024). Munyeza et al. (2020) examined indoor air PAH levels in Kenyan households, revealing that urban stoves emitted the highest PAHs, while wood-burning three-stone stoves contributed the most in rural areas. LPG was identified as a cleaner fuel since gas stoves produced no emissions, and kerosene stoves emitted fewer PAHs compared to conventional stoves. Toxic PAHs such as anthracene, benzo[a]anthracene, benzo[b]fluoranthene, and benzo[a]pyrene were predominant, posing significant health risks, particularly in poorly ventilated environments (Munyeza et al. 2020).

Outdoor sources of PAHs primarily stem from industrial and vehicular activities. PAH emissions from outdoor activities include e-waste combustion (Sahle-Demessie et al. 2021, Ghulam & Abushammala 2023), as well as agricultural and automotive emissions (Okafor et al. 2024, Berríos-Rolón et al. 2024). These sources contribute significantly to environmental PAH contamination, impacting both urban and rural ecosystems.

Research by Witter and Nguyen (2016) examined oxygenated PAHs and PAHs-containing sulfur and nitrogen in stream deposits collected from the Conodoguinet Creek watershed in South-Central Pennsylvania, USA. Their study demonstrated that metropolitan areas exhibit higher PAH concentrations due to industrialization, industrial transport, residential activities, vehicle traffic, runoff from impervious surfaces, and construction. These findings highlight how urban land use practices contribute to the persistence, bioavailability, and toxicity of PAHs containing oxygen, nitrogen, and sulfur in urban sediments (Witter & Nguyen 2016).

During the combustion of electronic waste, such as laptops, mobiles, and computers, PAHs can be emitted (Hoa et al. 2020, Sahle-Demessie et al. 2021, Wei et al. 2021a, Ghulam and Abushammala, 2023). Unsubstituted and methylated PAHs were identified by Hoa et al. (2020) in samples, which are surface soil and river sediment collected from a small village in northern Vietnam that had unofficial e-waste recycling operations. Workshop soil (870–42,000 ng g^{-1}) was followed by open burning soil (840–4200 ng g^{-1}), paddy field soil (530–6700 ng g^{-1}), and river sediment samples (370–2500 ng g^{-1}) in order of decreasing total levels of PAHs and methylated PAHs. Approximately 60% of the soil samples analysed in this investigation had elevated levels of PAH contamination. In several soil samples, the levels of anthracene, benz[a]anthracene, benzo[a]pyrene, fluoranthene, and phenanthrene were higher than the allowable level, suggesting the possibility of ecotoxicological impacts (Hoa et al. 2020).

Wei et al. (2021a) investigated the levels of PAHs in the air, soil, and grown plants at Taizhou, Zhejiang Province in China e-waste disposal sites. The plant samples had PAH values ranging from 29.7 to 2170.0 ng g^{-1}. Peeled taproot had the lowest concentration of PAHs. The differences in PAH concentration between the plant roots and shoots indicated many mechanisms for PAHs to enter the plants. It proved that the HMW (five- and six-ring) PAHs were less easily absorbed than the three- and four-ring PAHs (Wei et al. 2021a). This proved that root discharge and the peel can keep HMW PAHs in soil from getting to plants, especially taproots. Elevated concentrations of PAHs were identified in the leaves of most plants, suggesting that air deposition may have a considerable influence on PAH concentrations in airborne plant components (Wei et al. 2021a).

PAHs are released into the environment by automotive emissions (ships, cars, aeroplanes, and trains), and industrial and agricultural emission sources (Arias et al. 2022, Okafor et al. 2024, Berríos-Rolón et al. 2024). Off-road heavy and light transport systems also contribute to PAH emissions (Alao & Adebayo 2022). In Colombia, Arias et al. (2022) investigated a study to

evaluate the uncontrolled emissions of carbonyl and PAHs in both the solid and gaseous phases of an automobile diesel engine running in two typical urban driving scenarios. Totally 18 PAHs (16 found by the US EPA along with dibenzo[a,l]pyrene and benzo[e]pyrene) were identified. Up to 70% of the PAHs were in the gaseous phase, with the majority being those with three to four rings. It was noted that certain local combustion conditions may raise PAH emissions from both renewable fuels, primarily from LMW fuels (Arias et al. 2022).

PAHs are predominantly released into the environment due to incomplete combustion processes in urban and industrial settings. These emissions originate from various sources, including vehicular exhaust, industrial operations, and the burning of agricultural residues, all of which contribute significantly to environmental contamination.

Urban environments, particularly areas with high vehicular traffic, are major contributors to PAH pollution. A study conducted by Riccio et al. (2016) in a tunnel near Naples, Italy, investigated the mass of PAHs and particulate matter. Samples were collected from both sides of the tunnel at hourly intervals, representing the daily cycle. Chemical analysis revealed high PAH concentrations, reaching 1450 ng m^{-3}, with benzo(a)pyrene—a known carcinogen—measured at 69 ng m^{-3}. Emission factors were calculated based on fuel type and distance travelled, demonstrating that enclosed tunnels experience significantly higher PAH emissions due to limited dispersion. These findings raised concerns about human exposure to PAHs in densely populated urban areas.

Industrial activities are a major source of PAHs, particularly from processes that involve incomplete combustion. Key contributors include the production of agricultural chemicals, burning of agro-wastes, and steel and iron manufacturing. PAH emissions from steel and iron production have been widely studied. Wei et al. (2021b) analyzed PAH contamination in dust and soil samples near a steel and iron company in Laiwu, North China. Their study identified 16 USEPA-listed PAHs, with concentrations in road dust ranging from 0.460 to 46.970 μg g^{-1}, in roof dust from 0.670 to 17.140 μg g^{-1}, and in bare soil averaging 13.990 ± 1.203 μg g^{-1}. The most abundant PAHs in atmospheric dust were benzo[b]fluoranthene, benzo[k]fluoranthene, and fluorene, while benzo[b]fluoranthene, benzo[k]fluoranthene, and chrysene were dominant in settled dust. The study confirmed that industrial activities significantly elevate PAH levels in surrounding environments.

A study by Baek et al. (2022) in Pohang, Korea's largest iron-steel industrial site, investigated hazardous air pollutants (HAPs), including PAHs, at four locations—three in urban areas and one in a suburban area. The study found significantly higher PAH concentrations near the industrial complex compared to suburban sites, highlighting the influence of industrial emissions. Seasonal variations were also observed, with PAH levels being 4–5 times higher in winter than in summer. Among the detected PAHs, benzo[a]pyrene and dibenz[a,h]pyrene were found at all monitoring sites, indicating a potential health risk for local populations.

Open burning of agricultural residues is another major contributor to PAH emissions, particularly in developing countries. Fakinle et al. (2022) studied PAH emissions from the combustion of rice husks, bean chaff, maize cobs, and maize husks in Nigeria. The residues were burned in a furnace at 400°C, releasing various PAHs, including acenaphthylene, acenaphthene, anthracene, benzo[ghi]perylene, chrysene, fluorene, fluoranthene, pyrene, benzo[a]anthracene, dibenzo[a,h]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indene[1,2,3-cd]pyrene, naphthalene, and phenanthrene. Among these, dibenzo[a,h]anthracene had the lowest mean concentration (0.01 μg mg^{-1}), while benzo[b]fluoranthene had the highest (1.30 μg mg^{-1}). These findings highlight the environmental and health risks associated with agricultural waste burning.

PAHs from vehicular emissions, industrial operations, and agricultural waste combustion contribute significantly to environmental pollution. High concentrations of PAHs in urban tunnels, industrial areas, and regions affected by biomass burning pose severe health risks, particularly due to the presence of carcinogenic PAHs such as benzo[a]pyrene and dibenz[a,h]pyrene. These studies emphasize the need for stringent regulatory measures to control PAH emissions and mitigate their environmental and health impacts.

Distribution of PAHs in the environment

PAHs in the soil move within the environment by runoff or weathering. These processes finally lead to their transport into water systems (Hussain et al. 2018, Berríos-Rolón et al. 2024). PAHs in water systems bind to sediments, remain in the dissolved phase, or are adsorbed onto particulate matter or sediment. Both forms pose an ecological impact on aquatic life and microorganisms and can affect human health (Hussain et al. 2018).

According to the study by Frapiccini et al. (2020), PAHs in the tissue of red mullet (*Mullus barbatus*) were analyzed. Using a bottom trawl net, specimens were gathered monthly from an offshore fishing area in the Central and Northern Adriatic Sea. The study revealed that their physicochemical traits, catch season, and a few fish biological parameters (age, sex, reproduction, body size, total lipid content) were all linked to PAH bioaccumulation. Seasonality, reproductive stage and lipid content have little effect on LMW PAHs. While age and total lipid content appear to have little impact, no effect on body size. That study showed that reproduction and seasonality were significant factors in the formation of HMW PAH (Frapiccini et al. 2020).

Molbert et al. (2021) examined the effects of urbanization and pollution (phthalates, pyrethroid pesticides, organochlorine, polybromodiphenylethers, polychlorobiphenyls, PAHs) on the European chub (*Squalius cephalus*), which were taken from urban and agricultural rivers of the Marne hydrographic network, in France. Chub from urban rivers had shorter telomeres, and this reduction was associated with elevated liver phthalate metabolites. Speeding telomere degradation, urbanization, and pollution may jeopardize the survival of *Squalius cephalus* (Molbert et al. 2021)**.**

PAHs enter aquatic systems mainly through atmospheric dry and wet deposition, sewage discharges, industrial effluents, and runoff from urban or agricultural areas (Montuori et al. 2022). Sediment-bound PAHs can resuspend into the water column, increasing exposure risks for aquatic organisms. They can also undergo long-range transport through watersheds before reaching the ocean (Bralewska & Rakowska 2020). Kumar et al. (2024) examined surface water from India's urbanized water bodies, such as rivers and drains. The concentration ranged from 1.64 to 73.4, 1.98 to 93.0, and 1.99 to 67.4 mg L^{-1} in the Hindon, Yamuna, and drainage rivers. Naphthalene, acenaphthylene, fluorene, and chrysene proportions were 62.3% to 74.4% and 17.7% to 24.8%, respectively. Their dominance indicated emissions from burning wood, grass, and industrial oil. The concentration was lower in the upper part of the rivers than in the lower portion. This revealed that it originated from urban emission sources. The source estimation analysis identified pollutants, and long-range atmospheric transport of PAHs, including emissions from accidental spills that affect aquatic life (Kumar et al. 2024).

PAHs reach the atmosphere in particle-bound or gas forms once they are formed (Bralewska & Rakowska 2020). In the atmosphere, LMW PAHs typically stay in the vapour phase, while HMW PAHs adsorb to particulate matter, aiding in their long-range transport (Zhang et al. 2020). Eventually, PAHs in the atmosphere enter terrestrial or aquatic environments through wet or dry deposition (Abdel-Shafy & Mansour 2016).

Xia et al. (2022) used the entire month of averaged dry and wet deposition samples from Chongqing, a city in Southwest China, to find parent, oxygenated, and nitrated PAHs (PAHs, OPAHs, and NPAHs). In wet depositions, the majority of PAH species were linked to the particulate phase, whereas OPAHs and NPAHs were primarily found in the dissolved phase. This is most likely because N/OPAHs have a lower octanol-water partitioning coefficient than PAHs. Vehicle emissions (48.6%), coal combustion (13.4%), petroleum (5.9%), and secondary formation (32.1%) were the sources found using principal component analysis (Xia et al. 2022).

Toxicity and Health Risks of PAHs

PAHs are long-lasting in the environment and cause severe impacts on ecosystems and human health because of their toxic nature and ability to accumulate in living organisms. Exposure to PAHs can result in various acute symptoms, including nausea, eye irritation, and vomiting (Agrawal et al. 2018, Barbosa et al. 2023). In higher concentrations, PAHs can lead to more severe health impacts like asthma (Yu et al. 2022, Sherris et al. 2024), liver and kidney damage, skin

inflammation (Agrawal et al. 2018, Zhang et al. 2024), and immune suppression (Seise et al. 2020, Yu et al. 2022).

Sherris et al. (2024) analyzed 1,081 parent-child dyads at birth from five USA cities. Midway through pregnancy, the amounts of the maternal urine mono-hydroxylated PAH metabolites were assessed. Ten per cent of the study population had asthma. There was evidence of a negative correlation between asthma or wheeze trajectories and hydroxylated PAH concentrations during pregnancy. Although significant connections were noted among girls, there was no strong evidence linking prenatal exposure to PAHs to childhood asthma (Sherris et al. 2024).

According to Seise et al. (2020), in the USA, male offspring of Sprague Dawley rats ingested different amounts of 2-aminoAnthracene (2AA) contaminated diet for 12 weeks. It was shown that Type 1 diabetes is an autoimmune disease brought by a loss of insulin-secreting B-cells, which can be increased by exposure to PAHs, especially 2AA (Seise et al. 2020). Exposure to PAHs causes several health issues, including reproductive and developmental disorders, skin and lung cancer, respiratory complications, and genetic damage (Okafor et al. 2024, Moubarz et al. 2023).

Moubarz et al. (2023) studied workers in aluminium companies in Cairo, Egypt's Helwan and El Tebbin neighborhoods. Blood samples were taken to measure the squamous cell carcinoma antigen, anti-Cyclin-B1 marker, and Benzo(a)pyrene diol epoxide albumin adduct. Alpha-1-anti-trypsin gene mutation and human apurinic/apyrimidinic endonuclease gene polymorphism were found. It was discovered that exposure to PAHs may cause a risk for lung cancer. Even though the PAH in the atmosphere were within acceptable bounds, exposed workers' elevated BPDE-albumin adduct levels showed signs of DNA damage (Moubarz et al. 2023).

The study conducted on fish by Jafarabadi et al. (2020) examined the accumulation of PAHs in the muscles and livers of fish from the Persian Gulf, specifically *Lethrinus microdon*, *Lutjanus argentimaculatus*, and *Scomberomorus guttatus* from Kharg Island, Iran. The results of the human health risk assessment showed that there was a substantial possibility of consuming fish that included PAHs, indicating lifetime cancer risk is more than 1×10^{-5}, and excess cancer risk is more than 1×10^{-6} (Jafarabadi et al. 2020). The results reveal that long-term monitoring and thorough management are required.

Grmasha et al. (2023) investigated 16 PAHs over six months in surface water and the sediments close to the Tigris River's oil refineries and its near estuaries in Iraq. The overall findings indicated that most sites showed an incremental lifetime cancer risk (Grmasha et al. 2023). PAH exposure causes numerous health impacts on humans.

PAHs Remediation Techniques

A variety of treatment techniques have been utilized for the remediation of PAHs in the environment, addressing the persistence and toxicity of PAHs. These methods are designed to break down, remove, or neutralize PAHs in contaminated water, soil, and air. There are physical, chemical, biological, and nano-based techniques in PAH remediation.

Physical Remediation

The process of physically removing or separating PAHs from contaminated media, such as soil or water, is known as physical remediation (Zhang et al. 2024). PAHs are soluble in organic solvents due to their hydrophobic nature. Consequently, appropriate solvents such as acetone, dichloromethane, hexane, alcohol, methyl ethyl ketone, and toluene can effectively extract PAHs from sludge, soil, and water (Peng et al. 2018). Soil washing with solvent effectively addresses the challenge of removing HMW PAHs from the soil, as these compounds are difficult to eliminate due to the low bioavailability and high affinity for soil (Kuppusamy et al. 2017). Soil washing represents a highly effective technique that can be combined with other approaches to achieve optimal degradation of PAHs. The usage of surfactants has the potential to improve the effectiveness of soil washing through alterations in the solubility of PAHs (Gitipour et al. 2018, Zhang et al. 2024).

Mirzaee & Sartaj (2022) evaluated surfactant-enhanced soil washing with adsorption for PAHs in an accurate contaminated soil removal employing synthetic magnetic granular activated carbon as the adsorbent and Tween 80 as the surfactant. According to the findings, the repeated washing cycles had PAH removal efficiency of 68.6, 70.7, 70.3, 61.6, 55.5, 50.2, and 39.4%, respectively. Five per cent Tween 80, a liquid-to-soil ratio of 10:1, a 72-hour washing period, and a temperature of 20 ºC were the ideal ranges of the operational parameters for Tween 80-enhanced soil washing. The average percentage of PAHs removed from the polluted soil under these circumstances was 67.6% (Mirzaee & Sartaj 2022). In China, Zhang et al. (2022) explored biosurfactant-enhanced soil washing using rhamnolipid to remediate soil co-contaminated with phenanthrene and cadmium (Cd). Optimal conditions (pH 9, 5 g L^{-1} rhamnolipid, 15 ºC, 0.01 M ionic strength) achieved removal rates of 72.4% for Cd and 87.8% for phenanthrene, highlighting biosurfactant-enhanced soil washing efficacy (Zhang et al. 2022).

Different membrane-based filtration techniques, including ultrafiltration, reverse osmosis, and micro/nano-filtration, can be utilized to remove PAHs from water (Celebioglu et al. 2019, Li et al. 2019). Gong (2017) examined the electrocoagulation integrated membrane filtration method for removing PAHs from paper-making wastewater, noting a 94% removal rate of PAH achieved through electrocoagulation with low-pressure reverse osmosis membranes at 25 ºC. A variety of adsorbents, including activated carbon, biochar, modified clay, charcoal, nano-sulfonated graphene, magnetic nanomaterials, graphene oxide, and electro-spun nanofibers, have been employed to eliminate PAHs from water and soil (Eeshwarasinghe et al. 2018, Patel et al. 2020, Alao & Adebayo 2022, Queiroz et al. 2022).

Thermal remediation is a type of physical remediation (Patel et al. 2020). Among the significant factors affecting the remediation of contaminants, temperature exerts a noticeable effect. Recently, thermally enhanced biodegradation in soil and groundwater remediation has received high attention due to the rapid development of thermal treatment technologies (Liu et al. 2022, Ali et al. 2023).

Elevated temperatures enhance the permeability and fluidity of microbial cell membranes, thereby boosting the metabolic activities of microbes and their ability to biodegrade hazardous contaminants such as PAHs (Ali et al. 2023). Based on the temperature conditions, thermal remediation can reduce the levels of PAHs. Nonetheless, the fundamental properties of the soil could be modified (Liu et al. 2022). Temperature plays a significant role in bioremediation, as extremely high or low temperatures can diminish microbial quantity and activities, thereby affecting the bioremediation process (Ali et al. 2023).

Following natural air-dry ageing, the thermal remediation study was performed by Liu et al. (2022) in China utilizing a tube furnace and thermal gravimetry-Fourier transform infrared apparatus for phenanthrene, benzo(a)pyrene, and pyrene over two hours. Over 84% of phenanthrene and pyrene were decomposed during the ageing phase. The concentration of PAHs diminished a little below 100 ºC. However, upon elevating the temperature to 200 ºC, the level of benzo(a)pyrene decreased from 38% to 23%, while phenanthrene and pyrene each declined by approximately 1%. At 400 ºC, the concentrations of benzo(a)pyrene, phenanthrene, and pyrene diminished to around 1%, whereas no notable variation was seen at 800 ºC (Liu et al. 2022).

Ali et al. (2023) examined the negative effect of temperature on bioavailability and biodegradation of the production of reactive oxygen species through thermal biodegradation in co-contamination scenarios involving benzene and benzo(a)pyrene. The rates of biodegradation benzene rose from 57.4% to 88.7% and 84.9%, while the biodegradation efficiency of benzo(a)pyrene climbed from 15.8% to 34.6% and 28.6%, as the temperature elevated from ambient 15 ºC to 45 ºC and 30 ºC, respectively (Ali et al. 2023).

Physical remediation is an effective method for promptly mitigating PAH pollution. Established techniques such as incineration and thermal desorption are proficient in eliminating contaminants and can be utilized in diverse settings, including soil and water. Nonetheless, these procedures may incur substantial expenses owing to the necessity for specialized apparatus, considerable energy consumption, and transportation demands. They may also emit greenhouse

gases and generate secondary waste, such as ash or contaminated water, necessitating further treatment. Moreover, HMW PAHs may provide more significant treatment challenges, and specific treatments may result in residual contamination.

Chemical Remediation

Physical methods alone are inadequate for effectively removing PAHs from the environment because they do not alter the chemical structure of these persistent pollutants. Techniques such as sedimentation, adsorption, and filtration can temporarily remove PAHs from contaminated sites but do not lead to their degradation, allowing for potential recontamination (Ali et al. 2023, Zhang et al. 2024). Furthermore, these physical remediation methods are often time-consuming, require continuous maintenance, and have limited long-term efficiency. Due to these limitations, researchers have explored alternative approaches, with chemical methods being considered a more effective strategy for enhancing the bioremediation of PAHs (Jafari et al. 2018). Chemical treatments, such as oxidation and advanced chemical degradation techniques, can break down PAHs into less toxic or more biodegradable forms, thereby improving the efficiency of subsequent biological remediation processes (Kuppusamy et al. 2017, Peng et al. 2018, Gitipour et al. 2018, Ali et al. 2023).

Chemical oxidation processes involve general oxidants such as fenton reagent and ozone, as well as oxidants like potassium permanganate, peroxy-acid, hydrogen peroxide, and activated sodium persulfate (Peng et al. 2018). In the process of oxidation using the Fenton reagent, unstable hydroxyl radicals are produced in the presence of ferrous iron, which is employed to break down PAHs (Gong 2017, Gitipour et al. 2018).

Photocatalysis is a significant technique for eliminating PAHs from the environment. Photocatalytic materials like silica-functionalized graphene oxide/ZnO (Silica-GO/ZnO) nanocomposite have shown the ability to degrade 87% of pyrene (Jafari et al. 2018). Song et al. (2019) examined the removal of PAHs at an *in-situ* pilot scale using various zero-valent iron (ZVI)--activated persulfate oxidation methods. They documented PAH removal efficiencies of 82%, 63%, and 69% for nanostructured ZVI, stearic-coated nanostructured ZVI, and micron-sized ZVI-activated persulfate oxidations, respectively, over a treatment period of 104 days.

Chemical remediation is an effective technique for breaking down PAHs into less harmful substances, making it suitable for controlling water, soil, and sediment contaminations (Gong 2017, Gitipour et al. 2018). It operates efficiently and can be utilized with other techniques, whether implemented directly on-site or remotely.

Xu et al. (2021) used sulfate radical advanced oxidation processes to remediate PAH-contaminated soil. Ferrous sulfate with citric acid, citric acid, ferrous sulfate with surfactant IGEPAL CA-720 and sodium persulfate with ferrous sulfate were among the treatments in contrast to natural attenuation. The most outstanding PAH elimination was achieved by ferrous sulfate, citric acid, with surfactant IGEPAL CA-720, which is 80.8% during 15 days (Xu et al. 2021).

This method, while effective, can cause significant costs (Mohammadi et al. 2020) and may pose risks, including the potential generation of secondary pollutants or residual contamination (Gitipour et al. 2018, Song et al. 2019). The success is frequently impacted by variables, such as soil properties and environmental factors (Gong 2017, Gitipour et al. 2018).

Phytoremediation

The implementation of plants and root systems has taken attention because of their cost-effective nature, along with the adaptability of the treatment method, which can be implemented either *in-situ* or on-site (Sbani et al. 2020). This approach employs green plants to facilitate the degradation of PAHs in contaminated soils, water, and sediments (Allamin & Shukor 2021, Jani et al. 2024, Khuder 2024).

Various factors, including plant species, bioavailability, pollutant type, and soil conditions, influence the mechanism and performance (Allamin & Shukor 2021, Khuder 2024). Certain plant species are identified as "hyperaccumulators," making them particularly effective in

phytoremediation processes such as *Azolla pinnata, Cynodon dactylon,* and *Rhizophora mangle*. (Verâne et al. 2020, Song et al. 2022, Khuder 2024).

They are capable of tolerating and accumulating metals or organic substances found in the soil, including PAHs (Allamin & Shukor 2021). In phytoremediation, plants can absorb or directly adsorb PAH and decompose PAH through the release process. Ultimately, the degradation occurring in the rhizosphere has the potential to promote the microbiome (Gitipour et al. 2018).

Aquatic plants have the potential to generate significant biomass while demonstrating effective bioremediation properties. These aquatic plants may serve as a practical and economical approach to bioremediation (Khuder, 2024).

A study carried out by Khuder (2024) examined the impact of different pyrene concentrations (1, 5, 10, 15, 25, and 50 mg L^{-1}) on the aquatic plant *Azolla pinnata* collected from the Euphrates River in Iraq over 14 days. The outcome demonstrated that the maximum pyrene removal rate was 99% at concentrations of 25 and 50 mg L^{-1} by the 14th day (Khuder 2024). Verâne et al. (2020) investigated the phytoremediation of PAHs in mangrove sediments utilizing *Rhizophora mangle*, a species of red mangrove in the River estuary in Todoso Santo Bay in Brazil. With remediation rates of 60.76% and 49.57%, respectively, the results show that the average concentrations of total PAHs after 90 days dropped from 10,540.33 ng g^{-1} to 4,135.82 ng g^{-1} in phytoremediation and to 5,315.45 ng g^{-1} in natural attenuation (Verâne et al. 2020). By Song et al. (2022) in China, the phytoremediation ability of Bermuda grass (*Cynodon dactylon*) was assessed in soils that were co-contaminated with Cd (23.1 mg kg^{-1}) and PAHs (1238.62 mg kg^{-1}). In comparison to unplanted soils, it showed higher rates of PAH elimination (41.5–56.8%) and a high Cd bioconcentration factor (>1). These findings demonstrated that Bermuda grass was viable for PAH degradation (Song et al. 2022).

Phytoremediation represents a sustainable and eco-friendly strategy that employs plants to address PAH contamination. This approach is cost-effective, non-invasive, and enhances soil quality and biodiversity. This approach can be tailored to different settings, including soil, sediment, and water, while improving polluted areas' aesthetic quality and offering supplementary benefits, such as biomass generation. Plants can absorb and occasionally degrade PAHs, rendering this approach efficient for on-site remediation with limited maintenance requirements. Nonetheless, its operation is slow, it is confined to shallow soil environments, and it faces challenges with HMW PAHs. Seasonal constraints, plant vulnerability, and risks of secondary contamination present further obstacles.

Nanotechnology-based Remediation

Nanotechnology represents a novel advancement with promising uses in the remediation of polluted soil and water. A range of materials, such as nanocomposite membranes, zeolites, nano adsorbents, carbon nanotubes, and carbon-based nanomaterials, were utilized in the remediation of PAHs (Paszkiewicz et al. 2018, Borji et al. 2020).

A study by Huang et al. (2019) demonstrated that magnetic graphene oxide exhibited outstanding adsorption capability for the removal of PAHs from water. The study investigated multiple environmental factors to improve the removal process. The results indicated that the nanomaterial reached a PAH removal efficiency of 98–100% at a pH 5 and an initial 100 mg L^{-1} concentration.

Paszkiewicz et al. (2018) conducted a study in which multi-walled carbon nanotubes were employed to selectively extract PAHs from an aqueous solution. The study revealed that helical multi-walled carbon nanotubes exhibited exceptional sorption abilities, attaining as much as 99% removal of PAHs, even at minimal concentrations. The structural and electrical characteristics of carbon nanotubes, which improve interactions with PAHs through non-covalent forces such as π–π interactions, are responsible for the observed efficacy (Paszkiewicz et al. 2018). Graphene oxide nanoparticles have attracted significant interest for their capacity to eliminate various pollutants from water via an adsorption mechanism, including PAH (Hussain et al. 2019). The findings

demonstrate how appropriate modern nanomaterials are for efficiently eliminating PAHs from water.

Bioremediation

Bioremediation employs living organisms to decompose toxic chemicals into less harmful substances (Mohsenzadeh et al. 2012, Abdullah et al. 2020, Obukohwo et al. 2020, Ojha et al. 2021, Ameh et al. 2023). This approach utilizes biological agents such as plants, fungi, algae, bacteria, and termites to mitigate pollutants in soil, water, or sediments, converting harmful contaminants into less harmful compounds (Obukohwo et al. 2020). Various remediation strategies have been explored, particularly those involving microbial applications with fungi, bacteria, and algae, to address PAH-contaminated settings (Lazzem et al. 2022, Ameh et al. 2023).

The research by Roszak et al. (2021) in Poland indicates that the occurrence of an indigenous microbial consortium taken from long-term PAH-contaminated soil can improve the bioremediation process. The *in vitro* hydrocarbon degradation achieved a remarkable 93.5% reduction in total PAH content within the first day of incubation, increasing to 99.22% after eight days. Experiments on *in-situ* bioremediation conducted in the polluted area revealed an average reduction in total PAH concentration of 33.3% after five months, and a significant decrease of over 72% after 13 months (Roszak et al. 2021).

A separate study by Dhar et al. (2022) in Australia found that after 28 days, 80% of 16 PAHs were eliminated by the methylotrophic enrichment culture from the inoculated gas plant soil in Newcastle, New South Wales, Australia. Interestingly, bioaugmentation removed 90% of four-ringed PAHs and 60% of five and six-ringed PAHs. Almost all phenanthrene and pyrene, as well as 65% of benzo(a)pyrene, were eliminated from the augmented laboratory waste soil (Dhar et al. 2022).

Bioremediation generally requires a longer duration compared to physical or chemical methods. Some HMW PAHs present challenges for microbial degradation, which diminishes the overall efficiency of the method. The process is significantly impacted by environmental factors, including pH, temperature, and oxygen levels, which may require adjustments in substances (Obukohwo et al. 2020, Ojha et al. 2021, Ameh et al. 2023).

Cost-efficient, environmentally safe, and practical, mycoremediation uses fungus or its components to remediate organic, inorganic, and developmental poisons. Akpasi et al. (2023) evaluate mycoremediation approaches using fungi to biosorb, precipitate, biotransform, and sequester environmental pollutants. Metals, persistent organic pollutants, and other contaminants were eliminated using mycoremediation. *Pleurotusdryinus*, *Trameteshirsuta*, and *Aspergillus niger* breakdown pesticides or azo dyes 91.0%, 94.0%, and 98.4%. Also proposed was mycoremediation to remove emerging pollutants from streams and soils. Metals, drugs, PAHs, pesticides, and weedicides can be mycoremediated by fungi (Akpasi et al. 2023).

Mycoremediation

Fungi play a significant role in biosphere dynamics by controlling the flow of energy and nutrients through their mycelial networks, acting as natural engineers of their surroundings (Agrawal et al. 2018, Obukohwo et al. 2020). Fungi demonstrate superior effectiveness compared to bacteria in bioremediation (Gupta et al. 2017, Abdullah et al. 2020). Their growth rate is accelerated, leading to increased biomass production, and their hyphae facilitate a broader environmental spread (Agrawal et al. 2018, Obukohwo et al. 2020). They function as degraders in particular niches due to the high surface-to-cell ratio found in filamentous fungi (Obukohwo et al. 2020).

Apart from petroleum and PAHs, white rot fungi were also discovered to degrade pesticides, explosives, synthetic dyes, polychlorinated biphenyls, and even creosote (Pointing 2001, Yang et al. 2017, Russo et al. 2019, Danouche et al. 2021). Bumpus et al. (1985) analyzed the preliminary studies and found that white rot fungi can degrade persistent xenobiotics with carbon structures analogous to lignin. The study investigated the chemical degradation of polychlorinated

lindane, dioxins, biphenyls, and trinitrotoluene by *Phanerochaete chrysosporium* (Bumpus et al. 1985). After discovering the records of fungal species and the contaminants they degrade, a new field emerged, and the term "mycoremediation" was coined in 2005 by Paul Stamets (Stamets 2005). Mycoremediation is a specialized approach within bioremediation, employing fungi to restore and remediate contaminated ecosystems effectively (Fayyad et al. 2020, Gupta & Pathak 2020, Alao & Adebayo 2022, Dinakarkumar et al. 2024).

Phanerochaete chrysosporium was the prominent white-rot fungal species employed initially for PAH remediation (Bumpus et al. 1985). According to Bumpus et al. (1985), in 10 mL of cultures containing 1.25 nmol of substrate, 116.8 pmol of benzo[a]pyrene was converted into carbon dioxide within 30 days by *P. chrysosporium*, due to extracellular lignin-degrading enzymes (Bumpus et al. 1985). According to the study conducted by Manjunatha et al. (2025) in India, during a 30-day period, *Aspergillus niger* degraded anthracene, acenaphthene, fluoranthene, and fluorene with degradation rates of 77.8%, 65.0%, 60.9%, and 52.5%, respectively. *Trichoderma harzianum* and *Fusarium solani* strains isolated from the leaf samples collected from highly contaminated metropolitan zones in Sri Lanka showed the best degradation capability in phenanthrene, anthracene, pyrene, and naphthalene (Dharmasiri et al. 2024). Khajehzadeh et al. (2024) investigated the ligninolytic enzyme activities and bioremediation potential of a number of native Iranian white-rot fungi. *Trametes versicolor, Trametes hirsuta,* and *Oxyporus* sp. were among the isolated strains that efficiently degraded 400 mg/L of anthracene by producing enzymes during 28 days.

Numerous studies have demonstrated that various fungal taxa can achieve at least partial degradation of lignin and associated hydrocarbons, including the mycorrhizal *Glomus caledonium*, the coprophilic *Agaricus bisporus*, the black yeast *Exophiala xenobiotica*, the airborne *A. fumigatus*, among other species (Have et al. 2003, Bento et al. 2005, Isola et al. 2013). Several fungal species, including *Aspergillus, Chrysosporium, Cladosporium, Coriolopsis, Cunninghamella, Mucor, Penicillium*, and *Trichoderma,* have shown significant potential in breaking down PAHs (Gupta et al. 2017, Agrawal et al. 2018, Abdullah et al. 2020, Stoyanova et al. 2022, Ameh et al. 2023).

Types of Fungi Involved in PAH Degradation

The degradation of PAHs by fungi is mainly done by two types of enzymes: ligninolytic and non-ligninolytic enzymes (Agrawal et al. 2018, Babu et al. 2019, Abdullah et al. 2020, Ameh et al. 2023).

Ligninolytic Fungi

Most white rot fungi are ligninolytic, producing extracellular enzymes essential for breaking down lignin and other organic materials. These enzymes include lignin peroxidase, manganese-dependent peroxidase, phenol oxidases (such as laccases and tyrosinases), and enzymes that secrete hydrogen peroxide (H_2O_2) (Godoy et al. 2016, Babu et al. 2019). These enzymes oxidize lignin in wood and facilitate the mineralization of aromatic ring structures, allowing for the degradation of PAHs (Anastasi et al. 2012, Agrawal et al. 2018). Common white rot fungi, such as *Armillaria* sp.**,** *Bjerkandera adusta, Crinipellis stipitaria***,** *P. chrysosporium***,** *Pleurotus ostreatus***,** and *Trametes versicolor* are frequently utilized for the degradation of PAHs or for lignin pretreatment, which is achieved through their production of ligninolytic enzymes (Agrawal et al. 2018, Abdullah et al. 2020, Ameh et al. 2023). For instance, *Trametes hirsuta* achieved a degradation of 91.26% of phenanthrene and 87.72% of benzo[a]pyrene within 12 days, alongside a biosorption of 12.35% phenanthrene and 18.36% benzo[a]pyrene (Rathankumar et al. 2020).

Similarly, *Marasmiellus* sp., a marine-derived fungus, recorded 100% pyrene degradation within 48 hours under saline conditions (Vieira et al. 2018). *Trametes hirsuta* demonstrated a degradation rate of 0.8%, 0.17%, and 0.46% for phenanthrene, chrysene, and benzo[a]pyrene, respectively, per mg of dry-weight biomass (Hidayat & Yanto 2018). *Ganoderma lucidum* has been shown to degrade 99.65% of phenanthrene and 99.58% of pyrene when grown in mineral salt broth

(Agrawal et al. 2018). These findings highlight the significant potential of ligninolytic fungi in the biodegradation of PAHs and their application in bioremediation processes.

Non-ligninolytic Fungi

Non-ligninolytic fungi primarily break down PAHs through Phase I and Phase II detoxification, which involve cytochrome P450 monooxygenase and epoxide hydrolase-mediated reactions (Babu et al. 2019). Additionally, a particular fungus generates extracellular enzymes like laccase (Godoy et al. 2016, Babu et al. 2019). Non-ligninolytic fungi that contribute to PAH degradation include strains of *Aspergillus*, *Cladosporium*, *Cunninghamella*, *Fusarium*, *Mucor*, *Penicillium*, and *Trichoderma* (Godoy et al. 2016, Babu et al. 2019, Abdullah et al. 2020, Stoyanova et al. 2022).

According to a study by Mao & Guan (2016), after 30 days of incubation, the *Scopulariopsis brevicaulis* strain isolated from PAH-polluted soil was able to degrade fluoranthene (62%), phenanthrene (60%), pyrene (64%), and benzo(a)pyrene (82%). The strain's degradation rate was 75% (Mao & Guan 2016). By Birolli et al. (2018), the degradation of anthracene (50 mg mL^{-1} initial concentration) in a 2% malt medium over 14 days was tested with fungi like *A. sydowii*, *Cladosporium* sp., *Mucor racemosus, Penicillium citrinum*, and *Trichoderma harzianum*. Among them, *Cladosporium* sp. indicated the highest anthracene degradation at 16%. Additionally, when exposed to various PAHs (50 mg L^{-1}) in malt for 21 days, *Cladosporium* sp. degraded anthrone (100%), acenaphthene (78%), anthracene (71%), fluorene (70%), nitropyrene (64%), pyrene (62%), fluoranthene (52%), phenanthrene (47%), and anthraquinone (32%) (Birolli et al. 2018).

Unique Attributes of Mycoremediation

Fungi can exist in extreme environmental conditions, such as acidic (low pH) environments and areas with limited nutrients, where bacteria and plants frequently struggle, such as acidic soils, excessive salinity, or high concentrations of contaminants (Abdullah et al. 2020). Their ability to thrive in diverse conditions, such as low pH, high salinity, and high temperatures, further allows their resilience in contaminated environments (Ma et al. 2015, Deshmukh et al. 2016). Fungi has the ability to penetrate contaminated substrates, making them more efficient in bioremediation processes (Obukohwo et al. 2020).

Fungi degrade PAHs through multiple pathways and exhibit a rapid growth rate, contributing to their overall efficiency in bioremediation (Deshmukh et al. 2016, Ameh et al. 2023). The mycelium of fungi secretes extracellular enzymes and acids that can degrade complex organic materials such as lignin and cellulose, which are significant components of plant fibres. Bacteria depend on intracellular activities while plants do not have the enzymatic capacity to degrade complex compounds. They stabilize or absorb pollutants (Ma et al. 2015, Fayyad et al. 2020, Alao & Adebayo 2022, Dinakarkumar et al. 2024).

According to Godoy et al. (2016), 23 fungal species were collected from soil in Spain contaminated with pyrogenic PAHs as a result of fuel spills. Of these, 12 species exhibited a remarkable ability to oxidize anthracene into 9,10-anthraquinone, a less toxic compound. *Scopulariopsis brevicaulis* was notably able to degrade 100 μM of anthracene without utilizing extracellular enzymes, achieving 80% conversion to 9,10-anthraquinone within 21 days. In comparison, the ligninolytic fungus *Fomes* sp. used enzyme-dependent pathways to reach a higher conversion rate of 91% over the same period. *Pleosporales* sp. took 42 days to degrade anthracene. Additionally, *Aspergillus ustus*, *Penicillium pinophilum*, *Trichoderma* sp. and *Trichoderma harzianum* displayed degradation rates above 50% within 21 days, while *A. niger*, *Drechslera phlei*, *Lewia infectoria*, *Loratospora aestuarii*, and *Ophiosphaerella agrostidis* showed less than 50% degradation conversion in the same time frame (Godoy et al. 2016).

In another study, conducted by Arun et al. (2008), the basidiomycete fungi *Coriolus versicolor* degraded 42% of pyrene, *Daedalea elegans* degraded 35.8% of naphthalene, and *Fomitopsis palustris* degraded 31.7% of anthracene, and *Pleurotus ostreatus* degraded 20.6 % of acenaphthene within 28 days of incubation. According to Mohamed et al. (2012), *Aspergillus*

terreus, obtained from Orman Garden in Egypt, successfully degraded 98.5% of naphthalene and 91% of anthracene within four weeks in Gram-sterile soil samples treated with 150 ppm of either naphthalene or anthracene.

Table 2 demonstrates selected global case studies based on fungi on mycoremediation of PAH in diverse settings.

Even though all these studies confirmed that they are successful at the experimental level, they are still not being practised in the field. Further, many studies (Abo-State et al. 2021, Hamad et al. 2021, Vipotnik et al. 2022) suggested that after a successful pot and field trial, the organism they used could be applied in the field. Therefore, there should be a mechanism to apply those things to the field.

Fungal Adaptation for Mycoremediation for Different Environmental Factors

The degradation of substances by fungi is influenced by their growth characteristics and ability to withstand specific environmental conditions. Several abiotic factors, including temperature, pH, availability of carbon and nitrogen sources, light intensity, humidity, aeration, and the presence of trace elements, play a significant role in fungal growth and the biodegradation process (Bamforth & Singleton 2005, Gupte et al. 2016, Alao & Adebayo 2022). Moreover, fungi produce secondary metabolites that break down PAHs, enhancing their degradation capabilities.

Temperature

Temperature is another major factor in microbial metabolism and the degradation of hydrocarbons under both *in-situ* and *ex-situ* situations. It is a significant factor that stimulates fungal growth and influences biodegradation. As the temperature rises, the solubility of oxygen decreases, which decreases the metabolic activity of aerobic microorganisms. It limits their efficiency in hydrocarbon degradation (Gupte et al. 2016). This higher solubility enhances the bioavailability and diffusion rates of hydrocarbons, ultimately accelerating the biodegradation process (Alao & Adebayo 2022). At lower temperatures, PAH viscosity increases, and the flexibility of LMW hydrocarbons decreases, making biological degradation more difficult (Alao & Adebayo 2022). Studies have shown that as the temperature increases, the solubility of hydrocarbons in the medium is also higher in the medium, making hydrocarbons more readily available to microorganisms for consumption (Gupte et al. 2016).

According to Al-Dossary et al. (2020), *A. flavus* grew for 15 days at 25, 30, and 35 ºC. At 30 ºC, the total concentration of PAH contained compounds reduced from 5470.906 μg L^{-1} to 414.858 μg L^{-1}, representing 92.41% degradation (Al-Dossary et al. 2020). This might be due to the increased growth of the fungi because this can artificially enhance the efficiency of the energetics in the system through the coming together of the energy source and cell surfaces. At elevated temperatures, this might be slowed down, possibly due to distortion of the cell surfaces. Maximum growth of white rot fungi *P. chrysosporium* was recorded at 25 ºC incubation period for all tested PAHs (acenaphthene, anthracene, fluoranthene, naphthalene, phenanthrene, and pyrene) after 7 days incubation. The growth of fungi directly impacts the degradation process (Abo-State et al. 2021). Even though a 25 ºC temperature is appropriate for *P. chrysosporium*, it is not possible to supply it under tropical environmental conditions.

The optimal temperature range for microbial hydrocarbon degradation in aquatic environments is different. In freshwater systems, the highest degradation rates are observed between 20–30 ºC, while in marine environments, the optimal range is 15–20 ºC (Varjani & Upasani 2017, Xu & Yu 2018). These temperature conditions significantly impact the efficiency of microbial communities in breaking down hydrocarbons, indicating the importance of temperature in the overall biodegradation process.

Table 2 An overview of significant global case studies on the mycoremediation of PAH in diverse settings.

Climate zone	Samples taken from	Environmental setting	Fungi used	Degradation efficiency	Specific conditions	Duration	Reference
Tropical zone	China	Isolated from the China Forestry Culture Collection Center (Beijing, China)	*Peniophora incarnata*	Phenanthrene and anthracene were depleted by 91% and 71%, whereas the removal of benzo [a]pyrene was 35%	25 ºC 150 rpm	42 days	Tan (2023)
	India	Hardwood stump	*Ganoderma lucidum*	*Ganoderma lucidum* degraded 99.65% of phenanthrene and 99.58% of pyrene in mineral salt broth	27 ºC pH 6	30 days	Agrawal et al. (2018)
	India	Crude oil	*Fusarium* sp.	*Fusarium equiseti* and *Fusarium* sp. degraded 98.6% and 92.9% of total PAHs in crude oil, *F. equiseti* showed 97.8% maximum LMW PAH degradation, and *Penicillium citrinum* exhibited a 100% removal of HMW PAHs	80 rpm 28 ºC	23 days	Barnes et al. (2023)
	Nigeria	Effluent released by a petrochemical refinery	*Aspergillus* sp., *Talaromyces* sp., *Fusarium* sp., *Trichoderma* sp.	*Aspergillus* sp., *Talaromyces* sp., *Fusarium* sp. and *Trichoderma* sp. degraded 50 mg L^{-1} naphthalene (94.6, 96.9, 99.7 and 99.8%), phenanthrene (91.1, 92.2, 99.5 and 99.8%) and pyrene (89.4, 90.5, 92.6 and 94.2%), respectively	150 rpm	21 days	Ameh et al. (2023)
	Southwest Nigeria	Contaminated soil	*Trametes versicolor*	81.0% of chrysene degradation was achieved by *Trametes versicolor*, and by adding plantain peel support, this increased to 91.0%. Benzo[a]pyrene indicated a 38.0% rate and 49.1% with support, respectively	28 ºC	56 days	Vipotnik et al. (2022)
Temperate	North France	PAH-contaminated soils from brownfields	*Penicillium canescens*, *Cladosporium cladosporioides*, *Fusarium solani*, *Talaromyces helicus*	All 4 species degraded more than 30% of the initial 500 µg of benzo[a]pyrene	25 ºC	9 days	Fayeulle et al. (2019)

Table 2 Continued.

Climate zone	Samples taken from	Environmental setting	Fungi used	Degradation efficiency	Specific conditions	Duration	Reference
Arid and Temperate	Spain	Spill of petroleum-refined products	*Scopulariopsis brevicaulis*, *Fomes* sp.	*Scopulariopsis brevicaulis* and *Fomes* sp. converted 100 µM of anthracene into 9,10-anthraquinone	28 °C 120 rpm	21 days	Godoy et al. (2016)
			Pleosporales sp.	*Pleosporales* sp. was able to degrade 100 µM of anthracene into 9,10-anthraquinone		42 days	
Arid zone	Iraq	Soil contaminated with crude	*Penicillium* sp., *Aspergillus* sp.	*Penicillium* sp. degraded crude oil (5%, 10%) with 32.10% and 25.50% removal rates, respectively *Aspergillus* sp. degraded crude oil (5%, 10%) with 19.28% and 13.72% removal rates, respectively	28 °C	40 days	Abdullah et al. (2020)
	Iraq	Contaminated soil of the Rumaila oil field	*Ceriporia lacerata*	At 20mg L^{-1} pyrene, *Ceriporia lacerata* indicated a 55.5% removal rate after eight days. With co-substrate (glucose), pyrene degradation efficiency improved to 90.5%	30 °C pH 7	8 days	Al-Hawash et al. (2022)
	Egypt	Oil field with petroleum hydrocarbons	*Alternaria alternate Penicillium chrysogenum*	*Alternaria alternata* showed 72.07 % PAH degradation, and *P. chrysogenum* showed 59.51%. degradation rate	28 °C	30 days	Hamad et al. (2021)
Arid zone	Egypt	Previously isolated	*Phanerochaete chrysosporium*	Degraded 100% of acenaphthene, phenanthrene and pyrene under 100 mg L^{-1} of each PAH in BSM supplemented with 2000 µM $MnSO_4$	25 °C 150 rpm	7 days	Abo-State et al. (2021)
Polar zone	Antarctic	Antarctic soil	*Penicillium* sp.	At 100 mg L^{-1} concentration, *Penicillium* sp. degraded naphthalene (15.0%), acenaphthene (10.0%), and benzo[a]pyrene (2.00%)	20 °C 110 rpm	28 days	Govarthanan et al. (2017)

HMW-PAHs - High molecular weight polycyclic aromatic hydrocarbons, LMW-PAHs - Low molecular weight polycyclic aromatic hydrocarbons

Nutrient Availability

Nutrients play a major role in the successful biodegradation of PAHs. The optimal ratio of Carbon (C), Nitrogen (N), and Phosphorus (P) (C: N: P) is crucial for supporting microbial growth and metabolism during the degradation process (Gupte et al. 2016, Alao & Adebayo 2022). However, an excess of nutrients can negatively impact biodegradability, as it may cause nutrient imbalances. This imbalance can result in certain nutrients becoming limiting factors, which slows down the degradation process (Gupte et al. 2016). Therefore, maintaining an appropriate nutrient balance is essential to ensure efficient PAH biodegradation.

Al-Dossary et al. (2020) assessed the biodegradation ability of *A. flavus* under different conditions. The findings demonstrated that the degradation process was more significantly accelerated by the yeast extract, an organic nitrogen source, than by the inorganic source, $(NH_4)_2SO_4$. With 93.13% degradation, the medium with yeast extract as the nitrogen supply had the highest rate of degradation, while $NaNO_3$ showed 82.7% and $(NH_4)_2SO_4$ showed 74.34 % after 15 days (Al-Dossary et al. 2020). This might be a result of the yeast extract's high nitrogen content, which the fungi could easily employ to thrive. The growth of fungal mycelia was negatively impacted by the inorganic source $(NH_4)_2SO_4$, which may have slowed the breakdown process by lowering the pH of the medium.

The biodegradation of phenanthrene was studied by Omoni et al. (2023) in endophytic fungal strains, *Clonostachys rosea*, *Fusarium* sp., *Fusarium oxysporum*, and *Trichoderma harzianum*. This was performed under various carbon: nitrogen ratios- 10:1, 20:1, and 30:1 using different nitrogen sources, urea (CH_4N_2O), malt extract and ammonium nitrate (NH_4NO_3) for 30 days. Higher levels of mineralization were observed in fungal cultures, *Clonostachys rosea* and *Fusarium* sp., with a C: N ratio of 10:1, in which PAH degradation followed the order 10:1 > 20:1 > 30:1. Among these, fungal strains *Clonostachys rosea* and *Trichoderma harzianum* exhibited the highest phenanthrene mineralization after amendments of the N source with malt extract. Besides this, for *Clonostachys rosea*, phenanthrene mineralization was enhanced by higher C: N ratios alone (Omoni et al. 2023). This shows conclusively that most endophytic fungal strains degrade phenanthrene more readily in disturbed conditions with N amendment, particularly malt extract. More PAH solubility, oxygen availability, and mass transfer into the aqueous phase for fungal uptake could have all resulted from agitated circumstances.

pH

pH is another crucial factor that can either promote or hinder fungal growth and biodegradation (Gupte et al. 2016, Alao & Adebayo 2022). Many hydrocarbon-degrading microorganisms require a neutral pH for optimal activity, as demonstrated in studies by Alao & Adebayo (2022). Even slight variations in pH can significantly impact overall biodegradation efficiency.

According to the study by Hkiri et al. (2023) in Tunisia, the pH increased from 5 to 7 and an increase in biomass accumulation of *Chaetomium jodhpurense*, *Chaetomium maderasense*, *Emmia lacerata*, *Paraconiothyrium variabile*, and *Phoma betae*, five fungal strains observed without changing other factors such as glucose and temperatures. Within that range, removal of phenanthrene and benz[a]anthracene was higher for all five fungal strains during the first 9 days. The highest biodegradation rate for phenanthrene was 78.8% for *E. lacerate*, followed by 74.1% for *P. variabile*, 69.3% for *C. jodhpurense*, 48.4% for *C. maderasense*, and 46.3% for *P. betae*. The highest removal rate for benz[a]anthracene was shown by *P. variabile* (70.7%), which was followed by *P. betae* (58.7%), *E. acerate* (57.8%), *C. maderasense* (28.8%), and *C. jodhpurense* (26.9%) (Hkiri et al. 2023).

Two strains of *Trichoderma* were examined by Petrisor et al. (2016) in Romania under various pH circumstances. *Trichoderma viride* and *Trichoderma pseudokoningii* were tested with varying pH ranges such as 4.5, 5.5, 7.5, and 8.5. *Trichoderma viride* and *Trichoderma pseudokoningii* biomass production decreased between pH 7.5 and 8.5 on alkaline media (Petrisor et al. 2016).

Although these pH values may be appropriate in the laboratory for these fungi, in the field, these conditions may vary because of other environmental factors. In mycoremediation, the effectiveness of PAH degradation mainly relies on the metabolic activity and growth of the fungi involved. If the ecological pH is not suitable for the fungal species being used, their growth will be inhibited, leading to slower biodegradation rates (Petrisor et al. 2016, Gupta & Pathak 2020). The growth rates of fungi across different pH levels are influenced by their natural habitats and the environmental conditions at their site of isolation (Petrisor et al. 2016, Hkiri et al. 2023). These factors play a key role in determining how well fungi adapt to and grow in various pH ranges. Maintaining optimal pH conditions is, therefore, critical for enhancing microbial degradation processes.

Limitations and Challenges of Using Mycoremediation

Incomplete degradation during the remediation process can lead to toxic by-products, causing ecological impacts. Modern technologies are employed to explore the toxicological mechanisms of these filamentous fungi. Molecular biology methods provide insights into the basic structure and natural functions of compounds, both *in vivo* and *in vitro* (Ghosh et al. 2023). Filamentous bulking, toxicity, and dispersed growth impact bioreactor operation problems (Babu et al. 2019). This process requires a long remediation time and cost compared to conventional methods. Also, some fungi show slow growth, which is another limiting factor in the application of mycoremediation. There is a use of fungal consortia (mixed cultures) to enhance PAH degradation and improve remediation effectiveness (Balaji et al. 2014, Gupta et al. 2017).

Mycoremediation of PAHs

Fungal species are essential for breaking down hydrocarbons in polluted soils and water. They do this by producing ligninolytic enzymes, like peroxidase and laccase, which decompose complex hydrocarbons into simple and less harmful substances, aiding in the bioremediation process (Dinakarkumar et al. 2024). In a study by Al-Hawash et al. (2019), *Aspergillus* sp. demonstrated impressive removal efficiencies for various hydrocarbons. The fungus achieved 60.3% crude oil degradation, 97.4% for Naphthalene, 84.9% for Phenanthrene, and 90.7% for Pyrene after seven days of incubation. The study also found that, under similar conditions, biodegradation rates were lower for crude oil (51.8%), Naphthalene (84.6%), Phenanthrene (50.3%), and Pyrene (55.1%), showing that different hydrocarbons exhibit varying biodegradability depending on their molecular structure and complexity (Al-Hawash et al. 2019).

Another fungal strain, *Aspergillus sydowii*, exhibited optimal degradation efficiency (98.7%) of Anthracene (100 mg L^{-1}) under specific environmental conditions such as neutral pH, 30°C temperature, 2 g of biomass, and 0.2% (w/v) salinity within just 72 hours. These results emphasize the importance of optimizing environmental factors such as temperature, pH, biomass concentration, and salinity for enhancing fungal degradation efficiency (Bankole et al. 2020). Additionally, the white-rot fungus *Phlebia acerina* demonstrated greater benzo[a]Pyrene degradation ability than *P. chrysosporium* and *P. sordida*. After 32 days of incubation, *P. acerina* achieved a 57.7% degradation rate, showing the potential of certain fungal species for effectively remediating HMW PAHs (Zhang et al. 2023).

These findings demonstrate the effectiveness of fungi in hydrocarbon degradation, with enzymatic action and optimized environmental conditions being key to maximizing biodegradation rates.

Intermediate and final products during PAH mycoremediation

Various fungi have the ability to degrade different PAHs in various pathways, resulting in different intermediates and byproducts for the same PAH. Table 3 indicates biodegradation of different PAHs by various fungi and the isolated areas with the use of intermediate and final products.

Table 3 Biodegradation of different PAHs by various fungi and the isolated areas with uses of intermediate and end products.

PAH	Fungi	Isolated Location and Country	Metabolites	Reference	Applications in Industry
Anthracene	*Phanerochaete chrysosporium*	Microbiological Resources Center (MIRCEN), Faculty of Agriculture, Ain-Shams University, Cairo, Egypt	9, 10 anthracenedione	Abo-State et al. (2021)	For the production of textile (Sen et al. 2023)
	Aspergillus sydowii	Mangrove soil near Gbaramatu, Delta State, Nigeria	2-hydroxybenzoic acid	Bankole et al. (2020)	Pharmaceutical activity (Juurlink et al. 2014)
Phenanthrene	*Phanerochaete chrysosporium*	Microbiological Resources Center (MIRCEN), Faculty of Agriculture, Ain-Shams University, Cairo, Egypt	Naphthalene	Abo-State et al. (2021)	Antimicrobial, antiviral, antidepressant, antidiabetic, antipsychotic, anti-inflammatory, antihypertensive, antiprotozoal, anticancer, anticonvulsant, and anti-neurodegenerative actions are just a few of the many medicinal qualities of naphthalene derivatives (Elrayess et al. 2023)
			Benzene		Human carcinogen (Chiavarini et al. 2024)
	Coriolopsis byrsina	The fruiting body of the decayed wood surface (Department of Botany, Guru Ghasidas Vishwavidyalaya, India)	Acetic acid anhydride	Agrawal et al. (2021)	Mostly used in the production of cellulose acetate, an important component of the plastic, tobacco, chemical, textile, and health sectors (Zeidan et al. 2020)
Fluoranthene	*Phanerochaete chrysosporium*	Microbiological Resources Center (MIRCEN), Faculty of Agriculture, Ain-Shams University, Cairo, Egypt	Di-n-octyl phthalate	Abo-State et al. (2021)	Use as a plasticizer (Aminova et al. 2021)
			4-(N-Methylamino) benzoic acid		Use as a raw material in the chemical industry to make folate, a vital vitamin needed for DNA replication and synthesis. Because of its capacity to absorb UV rays, it is also utilized in the manufacturing of sunscreen and hair dyes (Haroon et al. 2023)
Fluoranthene	*Talaromyces pinopilus*	Isolated from activated sludge enriched with benzo(b)Fluoranthene	9-Fluorenone	Egbewale et al. (2023)	Its optical, photophysical, and biological characteristics, along with the presence of conjugated poly-(en)yens, make it a useful compound with intriguing photochemical and physicochemical capabilities (Kaur 2024)
Pyrene	*Trichoderma* sp.	Tropical forest of Johor Bahru, Malaysia	Benzoic Acid	Al Farraj et al. (2020)	Food preservatives (Turnbull et al. 2021) Antimicrobial properties (Yerlikaya et al. 2021)
			Acetic acid		Manufacturing of acetic anhydride and vinyl acetate (Zeidan et al. 2020)

Anthracene Degradation

Phanerochaete chrysosporium degrades anthracene by oxidizing it to 9,10-anthracene-dione, a quinone derivative, which undergoes oxidative cleavage and ring fission to produce 1,2-benzenedicarboxylic acid, bis(1-methylethyl) ester. This intermediate is further metabolized through oxidative steps, ultimately integrating into the tricarboxylic acid (TCA) cycle and being mineralized into carbon dioxide and water (Abo-State et al. 2021). In contrast, *A. sydowii* begins by converting anthracene to anthracene-1,8,9(2H,8aH,9aH)-trione, a triketone derivative. Further oxidation and rearrangement yield 2,4a-dihydronaphthalene-1,5-dione, which is cyclized and reduced to form 1,3,3a,7a-tetrahydro-2-benzofuran-4,7-dione. The pathway concludes with the production of 2-hydroxybenzoic acid (salicylic acid) as the terminal metabolite (Bankole et al. 2020). Fig. 3 shows pathways for anthracene degradation by *P. chrysosporium* and *A. sydowii.*

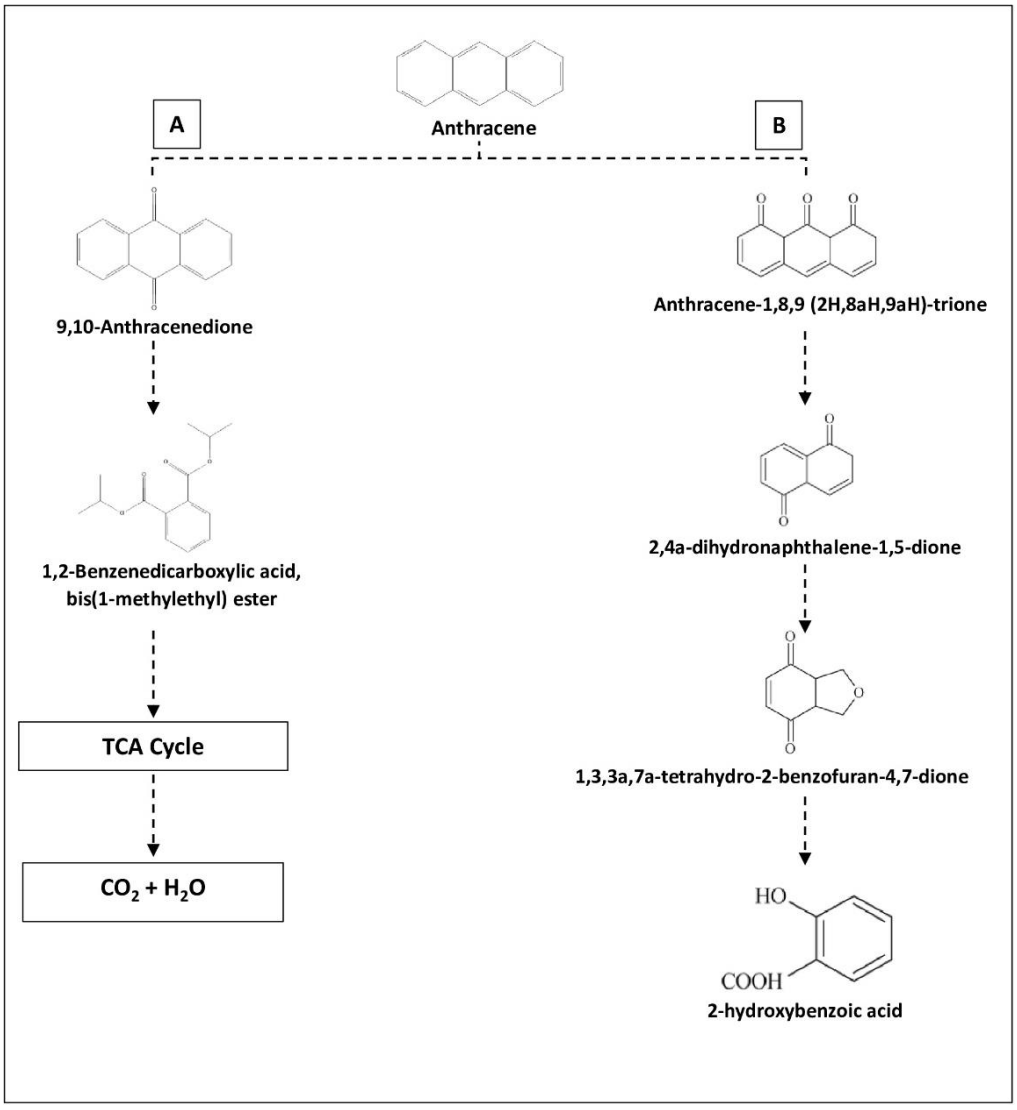


Fig. 3 – Pathways for anthracene degradation by *P. chrysosporium* and *A. sydowii*. A *P. chrysosporium*. B *A. sydowii*

Phenanthrene Degradation

Phanerochaete chrysosporium degrades phenanthrene by oxidizing it to naphthalene and cis-stilbene, which are processed separately. Naphthalene undergoes oxidation and ring fission, leading to benzene, while cis-stilbene converges on the same product. Benzene is further degraded into carbon dioxide or polymerized into phytane (Abo-State et al. 2021). *Coriolopsis byrsina* follows a pathway starting with the oxidation of phenanthrene to 9,10-dihydroxy phenanthrene, followed by

cleavage to 2,2-diphenic acid. These intermediate forms include phthalic acid, which can either be converted into benzene octyl, eventually mineralizing into carbon dioxide, or processed into acetic acid anhydride, leading to the same end product (Agrawal et al. 2021). Fig. 4 shows pathways for phenanthrene degradation by *P. chrysosporium* and *C. byrsina.*

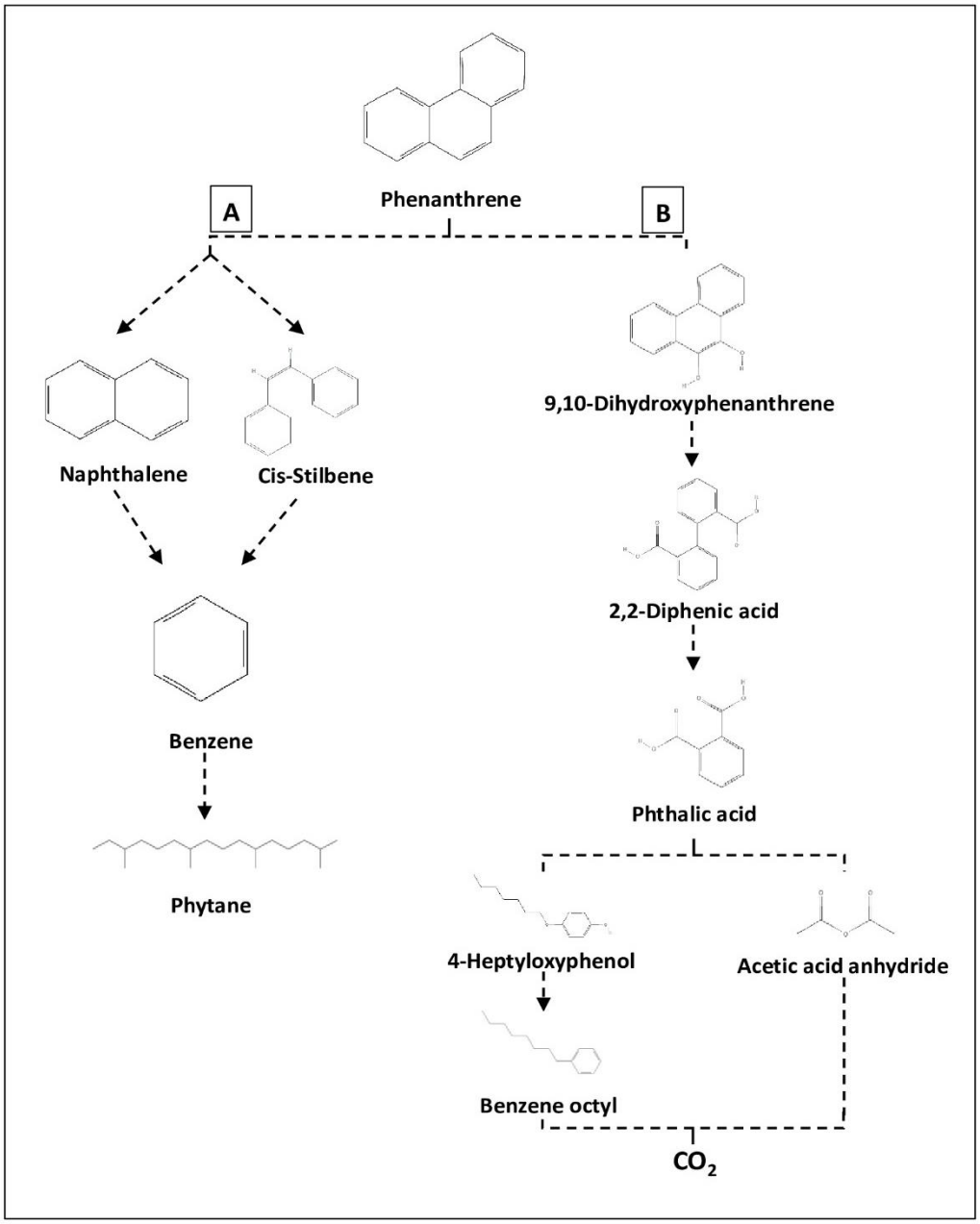


Fig. 4 – Pathways for Phenanthrene degradation by *P. chrysosporium* and *C. byrsina*. A *P. chrysosporium*. B *C. byrsina*

Fluoranthene Degradation

Phanerochaete chrysosporium begins fluoranthene degradation by oxidizing it to benzene-methanol, 3-phenoxy-, which undergoes ring cleavage to form di-n-octyl phthalate. This intermediate is further transformed into bis(3-methylbutan-2-yl) phthalate and eventually into 2-(neopentyloxycarbonyl) benzoic acid, concluding with the formation of benzoic acid, 4-(methylamino) (Abo-State et al. 2021). Alternatively, *Talaromyces pinopilus* initiates the process with the oxidation of fluoranthene to 9-fluorenone, which undergoes ring cleavage to yield 2,3-dihydro-1H-inden-1-one. This compound is converted into benzoic acid, which is mineralized into

carbon dioxide, completing the degradation (Egbewale et al. 2023). Fig. 5 shows the pathways for fluoranthene degradation by *P. chrysosporium* and *T. pinopilus.*

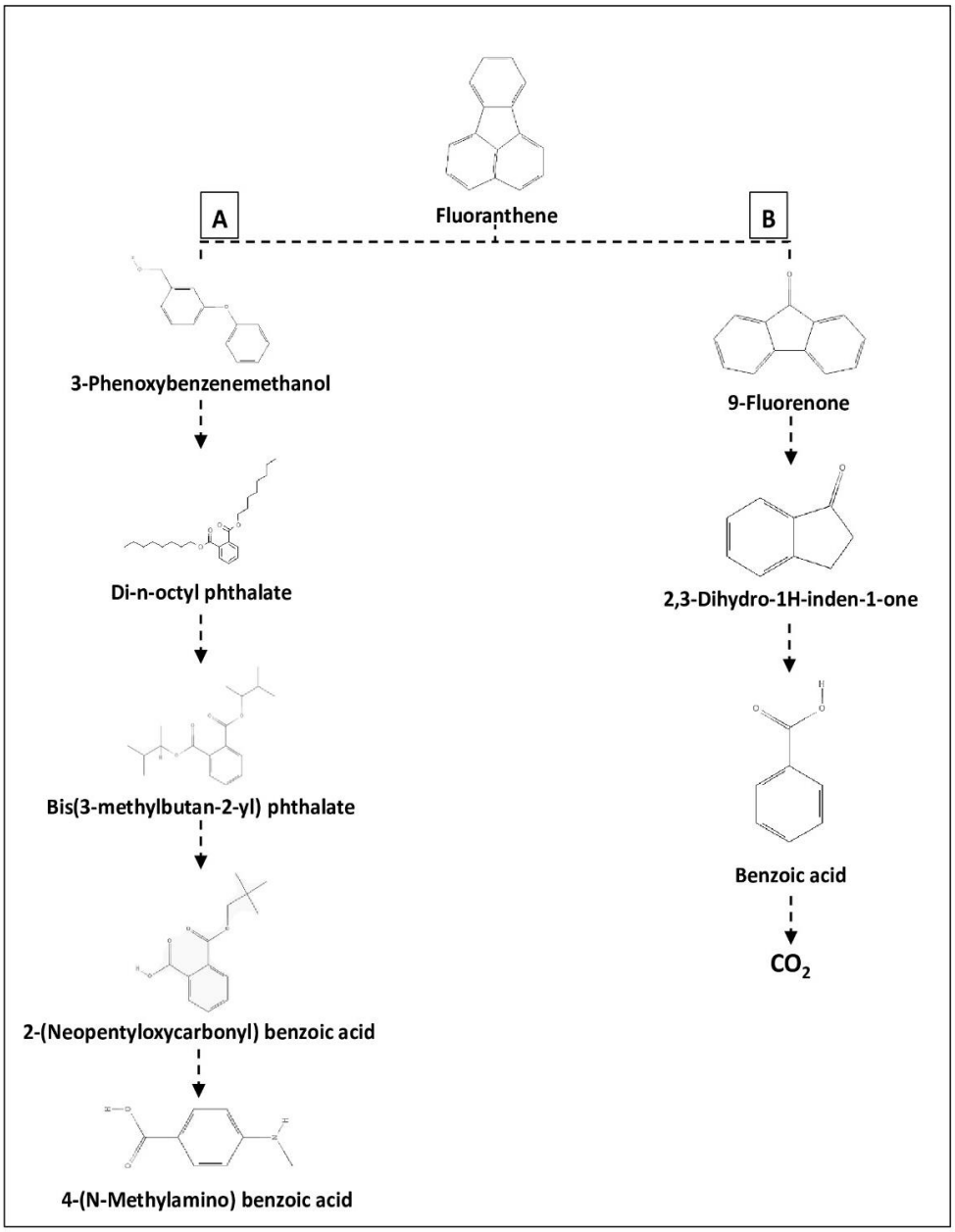


Fig 5 – Pathways for fluoranthene degradation by *P. chrysosporium* and *T. pinopilus*. A *P. chrysosporium*. B *T. pinopilus.*

Pyrene Degradation

Phanerochaete chrysosporium degrades pyrene through oxidation to 3,4-dihydroxy phenanthrene, which undergoes ring fission to produce phenanthrene. This intermediate is further processed into smaller aromatic compounds like biphenyl and naphthalene, which are then degraded into aliphatic hydrocarbons, including tetradecane, hexacosane, and octacosane (Abo-State et al. 2021). *Trichoderma* sp. oxidizes pyrene to pyrene-4,5-dione, followed by cleavage to form phenanthrene-4,5-dicarboxylic acid, which undergoes decarboxylation to phenanthrene-4-carboxylic acid and subsequently to benzoic acid. This compound is further broken down into acetic acid and carbon dioxide (Al Farraj et al. 2020). Fig. 6 indicates the pathways for Pyrene degradation by *P. chrysosporium* and *Trichoderma* sp.

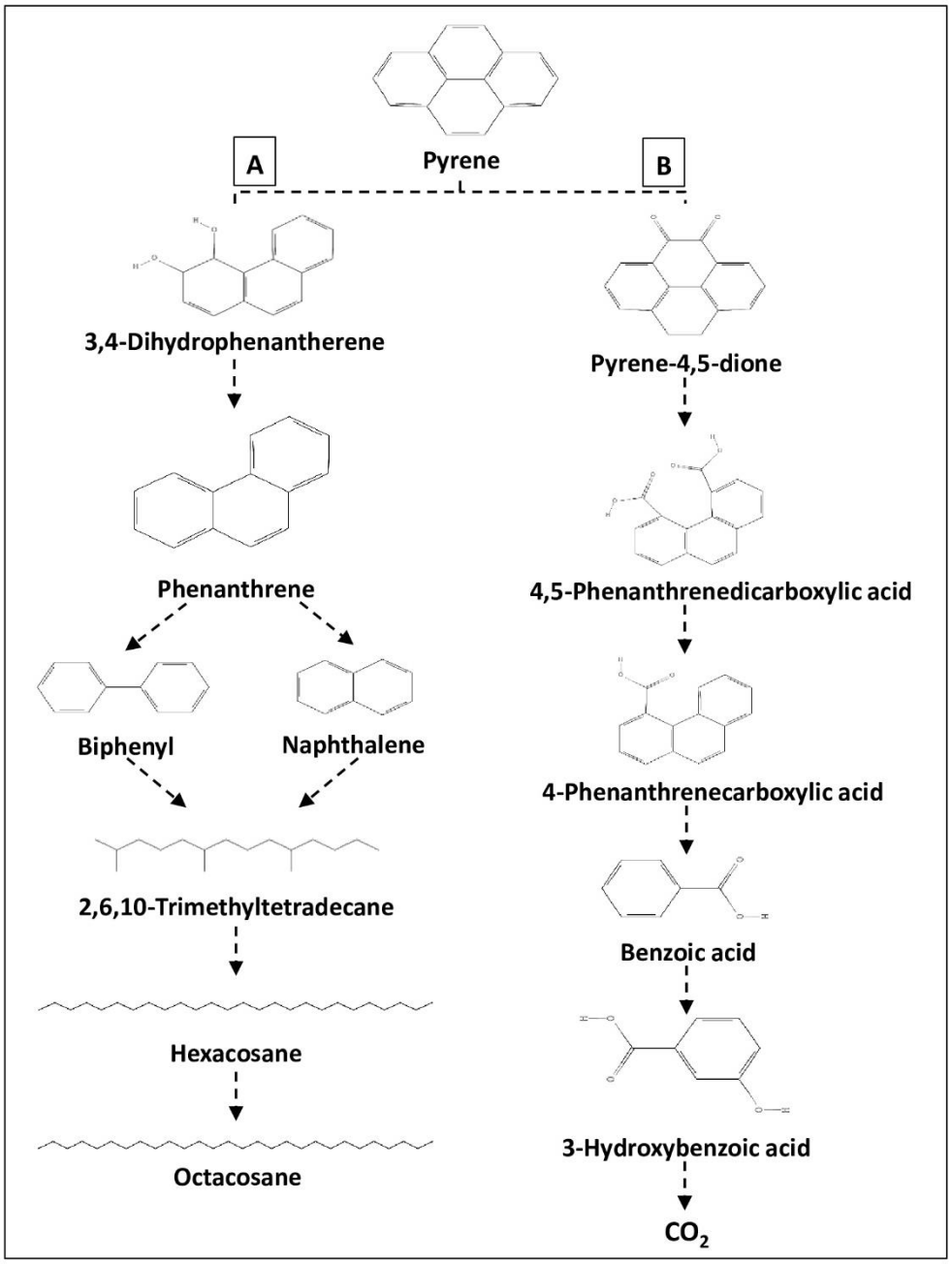


Fig. 6 – Pathways for pyrene degradation by *P. chrysosporium* and *Trichoderma* sp. A *P. chrysosporium*. B *Trichoderma* sp.

Mycoremediation of Diverse Contaminants

Other than the previous examples that have been discussed above, a wide range of extracellular enzymes found in fungi can degrade not only PAHs (Álvarez-Barragán et al. 2021), but also complex organic contaminants, such as dyes (Rajhans et al. 2021), heavy metals (Dell'Anno et al. 2022), and xenobiotics (Yadav et al. 2021). These enzymes can mineralize and oxidize pollutants, making them less harmful or innocuous to the environment. Laccases, manganese peroxidases, and lignin peroxidases (Benavides et al. 2024) are examples of these enzymes. Fig. 7 shows a schematic illustration of fungal growth and enzymatic pathways in mycoremediation applications.

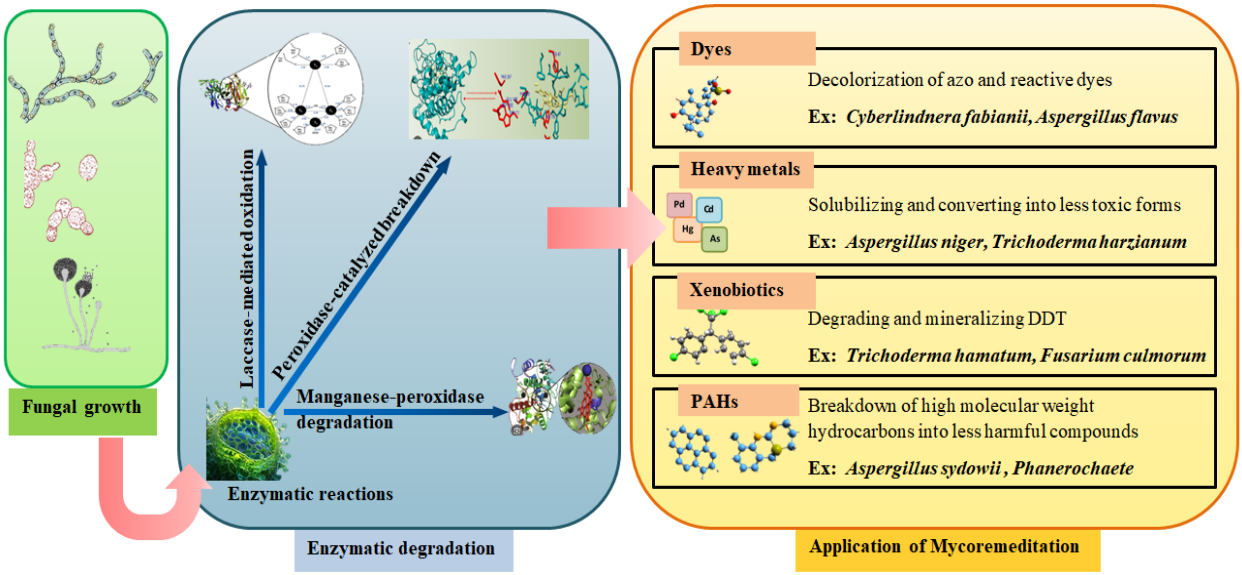


Fig. 7 – Schematic representation of fungal growth and enzymatic pathways in mycoremediation applications: a multifaceted approach to environmental pollutant degradation

Mycoremediation of Dyes

Multiple investigations have shown that some fungal species can effectively bioremediate synthetic dyes, highlighting the possibility of these organisms as green alternatives. According to Danouche et al. (2021), *Cyberlindnera fabianii* demonstrated impressive decolorization capabilities in less than 12 hours at a pH of 5.1. It removed more than 97% of the azo dye acid red 14 (50 mg L^{-1}). This efficiency proves that it works well for eliminating colours in acidic environments.

Similarly, *Trametes versicolor* decolorizes reactive and acid dyes with equal efficacy. This strain was shown to be able to remove more than 90% of dyes, including acid red 114 and reactive red 120, among others (up to 200 mg^{-1}), according to Yang et al. (2017), who documented this success in only 6 days. The efficacy varied among different dyes; for example, acid orange 7 removed 67% of the material after 9 days, whereas other dyes have shown superior performance. This variety exhibits a wide range of dye-degrading abilities and the impact of dye composition on the strain's efficacy.

Ranjusha et al. (2010) studied the biosorption capacity of *A. flavus* for the azo dye Remazol Black B. Increasing the dye concentration to 1000 mg L^{-1} resulted in a decrease in fungal biomass from 6.3 g L^{-1} to 1.44 g L^{-1}, despite an enhancement in dye absorption capabilities. Dye removal was reported to be complete at concentrations up to 250 mg L^{-1}, particularly at a pH of 4.5, indicating that biosorption is more efficacious in acidic conditions.

The results indicate that different fungal species exhibit varied levels of effective dye degradation, influenced by exposure length, pH, and dye concentration. Due to their diversity and ability to decolourize a range of colours, fungi such as *C. fabianii*, *A. versicolor*, and *A. flavus* hold significant potential as bioremediation agents for dye-polluted settings.

Mycoremediation of Heavy Metals

Traditional methods for removing heavy metals from the environment tend to be costly and are often not very effective at handling very low concentrations (Kour et al. 2022, Kumar & Dwivedi 2021). Microbial-based bioremediation has become a promising, affordable, and effective alternative for addressing environmental contamination. *Aspergillus niger* and *Trichoderma harzianum* are key players in this process because of their ability to release organic acids. These acids bind to heavy metals, aiding in their removal (Dinakarkumar et al. 2024). For example, *A. niger* produces citric acid, which can dissolve metals such as lead (Pb) and cadmium (Cd), transforming them into less harmful compounds that can be either immobilized or removed from the environment (Dinakarkumar et al. 2024).

Pleurotus sp. has shown the ability to reduce the concentration of cobalt (Co) more effectively than other heavy metals such as copper (Cu) and nickel (Ni). This reduction leads to increased fungal colony diameters, higher mycelial dry weights, and the production of antioxidative enzymes, further enhancing the accumulation and detoxification of heavy metals (Mohamadhasani & Rahimi 2022). In a study conducted by El-Bondkly & El-Gendy (2022), the ideal conditions for the bioremediation of Fe^{3+} and Co^{2+} from water solutions using *Aspergillus* sp. were determined. The best removal was achieved at temperatures between 45–55 ºC and pH levels of 4.5–5.0, with concentrations of 1000 mg L^{-1} for Fe^{3+} and 400 mg L^{-1} for Co^{2+}. These findings demonstrate the adaptability of fungal species to different environmental conditions and their ability for heavy metal bioremediation (El-Bondkly & El-Gendy 2022).

This highlights the promising role of fungi in the bioremediation of heavy metals, showcasing their ability to act under varied conditions and remove toxic heavy metals from contaminated environments.

Mycoremediation of Xenobiotics

Xenobiotics are chemicals not naturally found in the environment and created by humans. These include substances like agricultural chemicals and toxic wastes generated by industries such as paint, plastics, and textiles (Kour et al. 2022). In a study by Russo et al. (2019), the tolerance of *Trichoderma hamatum* and *Rhizopus arrhizus* to a 1 mg L^{-1} concentration of dichloro-diphenyl-trichloroethane (DDT) was assessed by measuring fungal biomass production and growth rate (Russo et al. 2019). The findings showed that both fungi demonstrated a precise tolerance to DDT. According to Ebsa et al. (2024), DDT and its major metabolites, DDE (2,2-bis(p-chlorophenyl)-1,1-dichloroethylene) and DDD (1,1-Dichloro-2,2-bis(p-chlorophenyl)ethane, were degraded in both media at the rate of 96–99% in 15 days incubated at 27 ± 2 ºC at 120 rpm at initial concentrations of DDT as 1750, 3500, 5250, and 7000 ppm by *A. niger* (Ebsa et al. 2024).

Fusarium culmorum was capable of degrading 96.9% of di(2-ethylhexyl) phthalate (initial concentration 3 g L^{-1}) within 13 days through the induction of esterase production in solid-state fermentation (Hernández-Sánchez et al. 2024). Puranik et al. (2023) indicated that *A. flavus* has the ability to mineralize 99.34% of dibutyl phthalate at a concentration of 0.1 g L^{-1} after 15 days of incubation in submerged fermentation (Puranik et al. 2023). The above examples indicated that some fungi, including *A. flavus*, *A. niger*, *F. culmorum*, *R. arrhizus*, and *T. hamatum*, have the ability to degrade different xenobiotics effectively.

Bioremediation Strategies Using Fungi

Microbially mediated bioremediation is a broadly preferred strategy for addressing global contamination issues (Kuppusamy et al. 2016, Babu et al. 2019, Kour et al. 2022, Dinakarkumar et al. 2024). According to the US Environmental Protection Agency, bioremediation can be categorized into two *in-situ* and *ex-situ* (Kour et al. 2022).

***Ex-situ* Remediation**

Ex-situ remediation involves the movement of pollutants from their initial polluted sites to another off-site for remediation (Babu et al. 2019, Kour et al. 2022). These treatments offer more precision and control in comparison to the *in-situ* method. When using *ex-situ* remediation, several factors are taken into account, such as the type of contaminant, pollutant, treatment costs, depth of contamination, and the location of the polluted site (Kour et al. 2022). However, additional costs arise from activities like soil excavation, transport, treatment, disposal, and refilling of the site (Babu et al. 2019).

The review covered by Kour et al. (2022) includes the bioremediation of various environmental contaminants, the procedures used in bioremediation, microbial diversity in bioremediation, and the monitoring of bioremediation processes. Bioreactors are an example of *ex-situ* treatment methods used for PAH remediation (Babu et al. 2019). Slurry-phase bioremediation, also known as bioreactors, involves a controlled process where contaminated soil is excavated,

mixed with water, and then placed into a bioreactor for treatment (Kour et al. 2022). Composting involves mixing contaminated soil with non-toxic organic materials such as hay, agricultural waste, manure, corncobs, and straw, while ensuring optimal air and water conditions for microbial activity (Godheja et al. 2019, Kour et al. 2022). The choice of amendments is dependent on the porosity of the soil and the necessary carbon and nitrogen balance to support microbial growth (Kour et al. 2022). Soil bio piles are used to remediate excavated soil contaminated with petroleum products, creating a suitable environment for both indigenous aerobic and anaerobic microbes (Kour et al. 2022).

In-situ Remediation

In-situ remediation is a treatment process conducted directly at the contaminated site. It eliminates the need for excavating and transporting soil, as well as the need to send the soil to off-site facilities for treatment (Gupta et al. 2017, Babu et al. 2019, Kour et al. 2022). Due to the above reasons, *in-situ*, remediation is considered a less expensive method in comparison to *ex-situ* remediation. For *in-situ* bioremediation to be effective, several environmental factors need to be considered, including weather conditions, soil permeability, nutrient availability, depth of contamination, the potential for moisture content, deep chemical leaching, pH levels, and temperature (Gupta et al. 2017). Land farming, biostimulation, bioaugmentation and bioventing are related to *in-situ* remediation (Babu et al. 2019).

Land Farming

The land farming process involves mixing contaminated soils with fertilizers, followed by tilling and irrigating. These steps are designed to encourage native microbes, enhancing the natural attenuation process. Additionally, they help to improve aeration, moisture levels, and soil uniformity, all of which support effective biodegradation (Wang et al. 2016).

Bioventing

Bioventing is a type of *in-situ* method of PAHs that combines biostimulation and bioaugmentation it has been effectively utilized to treat organic contaminants, including PAHs, in soil (Kuppusamy et al. 2016, Kour et al. 2022). This involves controlled stimulation of the airflow. It helps to provide oxygen to sustain the activities of microbes, increasing the bioremediation process (Kour et al. 2022).

Also, fungi interact with bacteria and other microbes to enhance PAH degradation through complementary metabolic activities. In a research investigated by Ma et al. (2015), bioaugmentation was applied in soil polluted with 50,000 mg kg^{-1} of total petroleum hydrocarbons (TPH) using two treatments: one with *Acremonium* sp. mycelia and the other with a consortium of *Acremonium* sp. and *Bacillus subtilis* mycelia. After 180 days, the TPH removal was 71.3 ± 5.2% for the first treatment and 74.2 ± 2.7% for the second (Ma et al. 2015). Reinoculated fungi treatments showed a remarkably high TPH removal ability compared to treatments that comprised bacterial augmentation. Several studies have shown that both pure fungal cultures and mixed cultures used for reinoculation have demonstrated effective and significant results in the removal of crude oil. Therefore, it is a cost-effective process (Ma et al. 2015, Kour et al. 2022).

Biopiling

In biopiling, dug soils are mixed with soil amendments. Then, it is kept in a treatment area and bioremediated using an aeration process. Here, pollutants are converted into carbon dioxide and water. Moisture, oxygen, nutrients, heat, and pH levels are regulated to increase biodegradation (Sharma 2012, Kour et al. 2022).

Bioaugmentation and Biostimulation

Biostimulation involves the addition of micro- and macronutrients, as well as electron acceptors, to enhance the biodegradation of the substrate (Hernández-Adame et al. 2021). In this

case, the removal efficiency or rate is higher in comparison to natural attenuation (Essabri et al. 2019, Hernández-Adame et al. 2021).

Bioaugmentation is related to the use of native or external microbial cultures to increase in-situ biodegradation, accelerating the process, especially in soils with high contamination levels (Ma et al. 2015, Abena et al. 2019, Hernández-Adame et al. 2021). Some bioaugmentation studies have shown fast degradation during the initial phase, followed by a significant decline in the degradation rate over time. The reason might be due to the low microbe numbers (Ma et al. 2015, Hernández-Adame et al. 2021). Through aerobic or anaerobic routes, hydrocarbons are removed, or microorganisms convert them into less hazardous metabolic products. The chemical makeup of the contaminants and the microbiological species found in the polluted soil have an impact on how well the remediation works (Ma et al. 2015, Prenafeta-Boldú et al. 2018).

For example, when bioaugmentation with *A. flavus*, *A. fumigatus*, *A. niger*, and *A. terreus* was used for the bioremediation of soils contaminated with 70,880 ± 975 mg kg^{-1} of TPH, the fungal consortium achieved the highest TPH removal at 57 ± 1.97%. Following this, the biostimulation process showed TPH removal rates of 49 ± 1.2% with *A. niger*, 44 ± 0.67% with *A. terreus*, 35 ± 0.98% with *A. fumigatus*, and 32 ± 0.38% with *Aspergillus flavus* (Hernández-Adame et al. 2021). A study by Essabri et al. (2019) investigated TPH removal through bioaugmentation and biostimulation using *Penicillium ochrochloron*, *A. niger*, and *Trichoderma viride* isolated from olive oil effluent. Within the first 30 to 40 days, *P. ochrochloron* removed 44% of the petroleum hydrocarbons, *A. niger* removed 49%, and *T. viride* removed 39%. *A. niger* degraded 71.19% of TPH among others (Essabri et al. 2019). García-Delgado et al. (2015) studied two bioaugmentation strategies for treating PAH-contaminated soil using *Agaricus bisporus*. Both augmentation approaches have been shown as highly effective in degrading more condensed PAHs (García-Delgado et al. 2015).

Conclusions

Fungi have the ability to bioremediate and biodegrade PAHs, which are the prominent major constituents of petroleum oil, as they have numerous capabilities, including the excretion of extracellular enzymes. It is also advantageous due to its environmentally friendly approach and cost-saving when compared to conventional methods. However, there are some drawbacks, including toxicity generation during the bioremediation and time consumption, which need to be considered.

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